# Role of filtration in managing the risk from *Cryptosporidium* in commercial swimming pools – a review

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## ABSTRACT

Most commercial swimming pools use pressurised filters, typically containing sand media, to remove suspended solids as part of the water treatment process designed to keep water attractive, clean and safe. The accidental release of faecal material by bathers presents a poorly quantified risk to the safety of swimmers using the pool. The water treatment process usually includes a combination of maintaining a residual concentration of an appropriate biocide in the pool together with filtration to physically remove particles, including microbial pathogens, from the water. However, there is uncertainty about the effectiveness of treatment processes in removing all pathogens, and there has been growing concern about the number of reported outbreaks of the gastrointestinal disease cryptosporidiosis, caused by the chlorine-resistant protozoan parasite *Cryptosporidium*. A number of interacting issues influence the effectiveness of filtration for the removal of *Cryptosporidium* oocysts from swimming pools. This review explains the mechanisms by which filters remove particles of different sizes (including oocyst-sized particles, typically 4–6 μm), factors that affect the efficiency of particle removal (such as filtration velocity), current recommended management practices, and identifies further work to support the development of a risk-based management approach for the management of waterborne disease outbreaks from swimming pools.

**Key words** | *Cryptosporidium* oocysts, filtration, particle counting, swimming pools, turbidity

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## INTRODUCTION

The source water which is used in swimming pools is usually of drinking water quality as it first enters the pool; thereafter it is sullied by bathers, either overtly as accidental releases of urine and faecal material, or more subtly due to ineffective showering practices, which can adversely affect the overall pool water quality (Ryan *et al.* 2017). This presents a risk to the health and safety of swimmers using the pool, which is mitigated through the implementation of pool water treatment processes that aim to return and continually maintain the water to a quality standard that is acceptable for bathing use. There has been growing concern about the increasing numbers of reported outbreaks of the gastrointestinal disease cryptosporidiosis, caused by the protozoan parasite *Cryptosporidium*. In 1988, the first reported outbreaks of cryptosporidiosis linked to a swimming pool occurred in the USA (Sorvillo *et al.* 1992) and in the UK (Joce *et al.* 1991). Between 1992 and 2011, there were 85 outbreaks of cryptosporidiosis linked to swimming pools in England and Wales (Chalmers 2012). However, this is likely to be an underestimate because the cryptosporidiosis



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outbreaks may not be identified or reported (Ryan *et al.* 2017).

The recirculating water treatment process in swimming pools typically includes a combination of filtration to physically remove particles from the water and disinfection by maintaining a residual concentration of an appropriate biocide, e.g. chlorine, in the pool water to kill or inhibit the growth of microorganisms. The pressurised filters remove suspended solids to maintain water clarity acceptable to both lifeguards and bathers, as well as microbiological safety of pool water. Pool filters normally contain sand, with typical particle size in the 600–800 μm diameter range, though alternatives are used, e.g. crushed recycled glass (Rutledge & Gagnon 2002). Some pools will have non-residual secondary disinfection in place such as ultraviolet (UV) light treatment or ozone dosing.

Most swimming pool treatment processes are not specifically designed and optimised for the removal of *Cryptosporidium*-sized particles; the main objective for filtration of pool water has traditionally been to clarify the water and to ensure that lifeguards can see across the whole of the bottom of the pool. Furthermore, due to the extended persistence and infectivity of the *Cryptosporidium* oocysts, it is thought that swimming pools could be a sink of *Cryptosporidium* and act as a vector for disease outbreaks.

*Cryptosporidium* is a single-celled protozoan parasite that infects the gut and can cause gastroenteritis, characterised by watery diarrhoea, abdominal pain, nausea and vomiting, and low-grade fever. It is transmitted between hosts (humans or animals) by the oocyst stage shed in faeces. The oocysts of human-infective species are typically 4–6 μm in diameter and are largely resistant to disinfection by conventional pool water biocides such as free chlorine (Chalmers & Davies 2010); the $C_t$ value (concentration of disinfectant multiplied by exposure time) for a 3 $\log_{10}$ reduction in oocyst viability reported for free chlorine at pH 7.5 and at 25 °C is up to 15,300 mg/L min (Shields *et al.* 2008). This corresponds to a disinfection time of 10.6 days in pool water containing 1 mg $L^{-1}$ free chlorine (Chalmers *et al.* 2016) and confirms that normal free chlorine dosage provides no practical residual disinfection for *Cryptosporidium* oocysts in a swimming pool.

A single accidental faecal release (AFR) in a 450 $m^3$ municipal pool, if well-mixed, could result in an average concentration of around 20,000 oocysts $L^{-1}$ (20 $mL^{-1}$) (Gregory 2002). Dufour *et al.* (2006) estimated that the average amount of water swallowed during a 45 min pool session was 37 mL for non-adults and 16 mL for adults. If a pool user swallowed only 10 mL water containing 20 oocysts $mL^{-1}$, this would lead to ingestion of 200 oocysts. This is well above the reported infective dose (<100 oocysts) (Ryan *et al.* 2017), and dose–response modelling has shown a chance of infection from ingestion of just a single oocyst (Messner *et al.* 2001). This could, therefore, pose a major risk to the public health of swimmers using a pool.

In most swimming pools, the principal protection against *Cryptosporidium* is removal through filtration (WHO 2006). However, conventional pool filters are not designed and commissioned for the purpose of *Cryptosporidium* removal (Amburgey *et al.* 2012). In some pools, there are other non-residual disinfectant processes that can inactivate *Cryptosporidium*. These include UV light or ozone (both of which are applied within the plant room). However, most pools rely on filtration alone for the removal of oocysts and on good practices to prevent contamination in the first place, such as insisting on pre-swim showering and dissuading customers with gastroenteritis from using the pool (Ryan *et al.* 2017; Chalmers & Johnston 2018).

Data from the drinking water treatment industry indicate that, in combination with effective coagulation/flocculation and sedimentation, sand filters can achieve a 1.5–3 $\log_{10}$ removal of oocysts (LeChevallier *et al.* 1991; Gregory 2002; Betancourt & Rose 2004). This equates to 97.2% and 99.9% removal of oocysts from water in a single pass through the filter (hereafter referred to as the filter efficiency, $E$). However, less is known about the removal of oocysts from swimming pool water, though it is suggested that swimming pool filtration systems may be less effective than drinking water treatment due to the frequent absence of effective coagulation/flocculation and sedimentation and often sub-optimal backwashing procedures. Furthermore, the maximum recommended water velocity through the filter in public pools with medium-rate filters is 25 m $h^{-1}$ (PWTAG 2017b), though this may be exceeded in practice. These are substantially greater than velocities of no more than 10 m $h^{-1}$ used in drinking water treatment (Gregory 2002). Studies on removal of oocysts by pool filters have been restricted mainly to the use of surrogates,

e.g. polystyrene microspheres in pilot-scale studies (e.g. Croll *et al.* 2007; Lu & Amburgey 2016) and in one case measurement of oocyst-sized particles in an operational pool (e.g. Stauder & Rodelsperger 2011).

The removal of oocysts from water through filter retention is a complex process in packed-bed sand filters. This includes factors that affect the delivery of oocysts from the pool to the filter (e.g. the location and number of filter inlets and outlets, and how this influences the mixing characteristics of the pool) through to processes within the filter itself. The aim of this paper is to focus on the latter and to provide a review of the current knowledge of the removal of *Cryptosporidium* oocysts by filters in commercial swimming pools and to identify the major risks and the gaps in our current understanding. The review considers the mechanisms by which filters remove particles of different sizes (including oocyst-sized particles), factors that affect the efficiency of particle removal (such as filtration velocity and use of coagulant), current recommended management practices, and suggestions for further work to support the development of a risk-based management approach. The main factors to be considered are listed in Table 1.

**Table 1** | Factors that could influence the removal of *Cryptosporidium* oocysts by swimming pool filters and their operational monitoring

| |
|---|
| **Processes within the filter bed** |
| Straining |
| Sedimentation |
| Interception (impaction, Brownian motion) |
| Surface retention/attachment (van der Waals forces) |
| Detachment |
| Ripening |
| **Design/operation** |
| Filter media choice |
| Filter media condition |
| Filter bed depth |
| Coagulation/flocculation |
| Flow rate/filtration velocity |
| Backwashing procedure |
| **Monitoring** |
| Visual inspection |
| Pressure differential |
| Turbidity measurement |
| Particle counting |

## HOW DO SAND FILTERS REMOVE PARTICULATE MATERIAL?

A swimming pool sand filter bed consists of packed solid particles, which, in the case of 16/30 sand (sand which passes through a No. 16 sieve but is retained by a No. 30 sieve), range in size from 0.6 mm to 1.2 mm (600–1,200 μm), with the majority normally being in the range 600–800 μm. The spaces between the packed sand particles (the porosity) make up 35–50% of the total volume occupied by the particles depending on how rounded or irregular the shape of the grains. If the particles are assumed to be spherical, then the effective diameter of the spaces between the particles is equivalent to one-seventh of the diameter of the sand grain (Huisman & Wood 1974). For 16/30 sand, the smallest pore size will, therefore, be about 0.1 mm (100 μm), so particles smaller than this (about the smallest size that can be resolved by the naked eye) will not be removed by a simple straining mechanism, but will move into the body of the media bed rather than being retained at the surface. *Cryptosporidium* oocysts (4–6 μm in size) will, therefore, not be retained simply by the sand particles acting as a screen. However, as larger particles become trapped within the pores, the space sizes are reduced further, or where localised restrictions occur due to irregular-shaped sand particles coming into close contact, increased straining could occur which may trap some oocysts. Straining is not, therefore, likely to be the major mechanism for removal of *Cryptosporidium* oocysts from pool water, unless the oocysts are present within a much larger floc which could occur where there is effective coagulation/flocculation or if the oocysts are attached to faecal material.

As water travels through the pores between the sand particles, it will pass by the extensive surfaces of the sand grains. For example, 1 $m^3$ volume of 0.6 mm diameter spheres will have an estimated total surface area of 6,252 $m^2$ (Huisman & Wood 1974). Particles which are too small to be screened could be retained by the filter media as a result of weak intermolecular binding forces that come into play if the particles can get very close (i.e. within nanometres) to the surface of the sand grains. This

surface adsorption is likely to be the principal mechanism for the removal of *Cryptosporidium* oocysts.

One approach to quantifying the effectiveness of particle removal by filters is to compare the measurements of the particle contents of the influent and effluent. The simplest index of the effectiveness of filtration is the filter efficiency ($E$), defined as the fraction (often expressed as a percentage) of particles removed from water as it passes through the filter:

$$E = \frac{C_0 - C_L}{C_0} \tag{1}$$

where $C_0$ and $C_L$ are the influent and effluent solid concentrations (or the particle counts, or turbidity, depending on the measurements made).

However, $E$ is not just a measure of the effectiveness of the media in removing particles, as this depends also on the depth of the media bed. The impact of the depth of the bed can be accounted for by considering a sand filter as a deep packed bed comprising layers of single collectors (sand grains), where each layer removes a fixed proportion of the particles suspended in the water approaching the layer. This gives rise to an exponential decrease in the particle content of the water as it moves down through the filter:

$$C_L = C_0 \exp(-\lambda L) \tag{2}$$

where $L$ is the filter depth in m and $\lambda$ is the filter coefficient in units of $m^{-1}$. $1/\lambda$ is known as the characteristic length of the filter (in m) which is sometimes used as a measure of the intrinsic effectiveness of the media in removing particles (Lawler & Nason 2006). Equation (2) can be used to derive an empirical value for the filter coefficient based on the measurement of the particle content of the influent and effluent. For example, if a media bed 0.8 m deep removes 70% of particles, then the value of $\lambda$ will be 1.50. This value will vary during the backwash cycle, dependent on the degree to which the filter media are loaded with finer material removed from the water (Amburgey 2005), and may also change if the filter media degrade in some way, e.g. if balling or channelling occurs (PWTAG 2017b). However, it is an oversimplification of a real filter because the particle removal capability of the layers will not be the same throughout the filter once the upper parts of the filter become loaded.

There is a wide range of factors that affect the filter coefficient (Tufenkji *et al.* 2006) and it is, therefore, unrealistic to expect to model the complexity of a real swimming pool filter from first principles. However, attempts to produce a mechanistic model of particle removal provide a valuable insight into some of the key processes (Ncube *et al.* 2018a). A starting point is to model the removal of particles from water as it approaches and passes a single spherical collector (Yao *et al.* 1971). Only a small proportion of the particles approaching a collector will get close enough to the collector surface for attachment to be possible (this dimensionless fraction being termed the transport coefficient, $\eta$), and only a proportion of those particles that make contact with the collector surface will attach (this proportion being the attachment coefficient, $\alpha$). The overall proportion of particles approaching the collector that actually attach is, therefore, the product of these two terms ($\eta\alpha$).

The model of Tufenkji & Elimelech (2004) scales up the single collector model of particle removal to predict particle removal by a filter bed. This model relates the filter coefficient to the media geometry (the collector diameter, $d_m$ and the filter bed porosity, $\epsilon$) and also to the single collector transport and attachment coefficients discussed by Yao *et al.* (1971):

$$\lambda = \frac{3\,(1-\varepsilon)}{2\,d_m}\,\eta\alpha \tag{3}$$

Particles attach to the collector surface by van der Waals forces, which is a universal but short-range phenomenon which holds particles at the surface once contact has been made. This is an important mechanism for removal of microscopic particles from pool water (Huisman & Wood 1974; Tufenkji & Elimelech 2004). In this context, van der Waals forces are the result of the sum of all the individual intermolecular forces between the two interacting surfaces. These forces will only be effective over very short distances (nanometres); hence particles, to all intents and purposes, have to be in contact with the surface of the media before adsorption can occur.

The starting point to modelling the transport coefficient is to recognise that contact between sand grain surfaces and

suspended particles is compromised because the water flowlines typically divert around the edges of the collectors (sand grains). If suspended particles are to make close contact with the collector surface, then they need to divert (break out) from the water flowlines. There are three principal mechanisms that can achieve this (Huisman & Wood 1974):

1. Sedimentation by gravity can cause suspended particles to approach the up-facing surfaces of collectors. The settling velocity depends strongly on particle size and density, and the value for *Cryptosporidium* oocysts at 23 °C has been estimated to be 0.35 μm $s^{-1}$ for isolated oocysts (Medema *et al.* 1998), but is affected when oocysts attach to other particles such as organic matter (Medema *et al.* 1998). To put this velocity into the context, a typical average downward velocity of water within the void space of a swimming pool filter is 60 m $h^{-1}$ (17 mm $s^{-1}$), providing an average time of about 0.03 s for water to pass a single sand grain. In this time, an oocyst would settle just 0.01 μm, making sedimentation a most ineffective transport mechanism for individual *Cryptosporidium* oocysts.
2. Particles can impact on the surface of the interceptor if they have sufficient momentum to break out of the diverting water flow line (impaction). Particles as small as *Cryptosporidium* oocysts will have very little momentum because of their small size rendering impaction also to be ineffective.
3. Random Brownian motion (resulting mainly from collisions between suspended particles), sometimes referred to as diffusion, can bring suspended particles into close contact with interceptor surfaces. With particles smaller than 1 μm (particles responsible for causing turbidity in water), diffusion will be the dominant process by which suspended particles make close contact with sand grains (Yao *et al.* 1971; Tufenkji & Elimelech 2004).

The overall value for the transport coefficient is the sum of the coefficients due to sedimentation, impaction and diffusion, each of which depends on a number of factors which include particle size and water velocity. Tufenkji & Elimelech (2004) describe one such attempt to model these three processes, and that has been incorporated into a proposed technique for designing filter systems (Lawler & Nason 2006).

The impact of particle size and filtration velocity on the filter efficiency was predicted using the Tufenkji & Elimelech (2004) set of equations presented within Lawler & Nason (2006) (Figure 1). The values used for the parameters were similar to Figure 2 in Lawler & Nason (2006), but also show the cases of filtration velocities of 10, 20 and 40 m $h^{-1}$ in addition to the 5 m $h^{-1}$ case presented originally. This

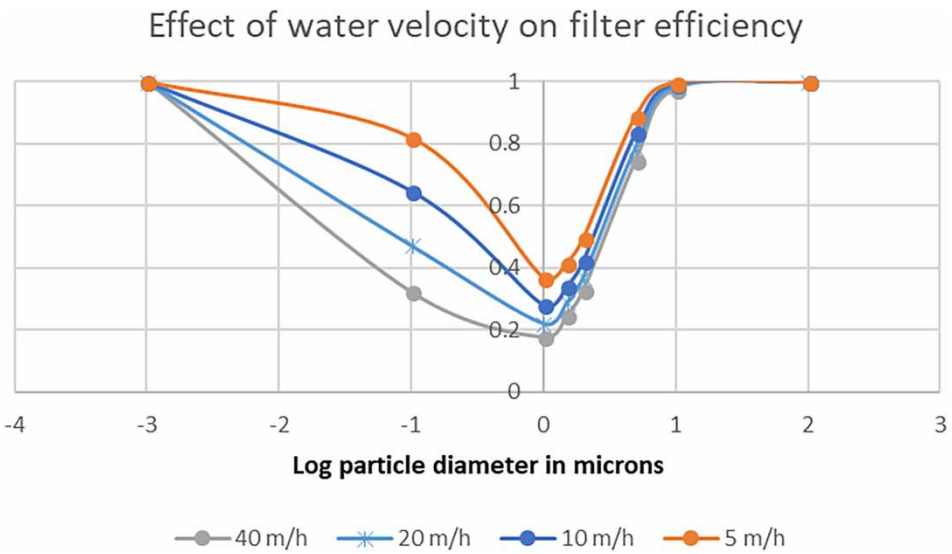


**Figure 1** | Predicted removal efficiency (*E*) of different size particles in packed-bed sand filters with different filtration velocities using the Tufenkji & Elimelech (2004) equations presented by Lawler & Nason (2006).

example filter comprised a media bed 0.6 m deep, 40% porosity and with 600 μm diameter spherical collectors (typical of a swimming pool filter).

Figure 1 shows the characteristic relationship between filter efficiency ($E$) and particle diameter, with a local minimum that modelling studies predict to be in the 1–5 μm range (Lawler & Nason 2006). It is important to note that with respect to *Cryptosporidium* oocyst removal by filtration, it is the particles within the 4–6 μm size range that have the least chance of making sufficiently close contact with sand surfaces to enable their adsorption. Particles of this size tend to be too small for sedimentation/impaction to be effective and too large for diffusion to be effective. Together, these factors render the removal of *Cryptosporidium* oocysts from pool water by filtration alone a challenge.

Though the mechanistic modelling studies above give an insight into the issues affecting the effectiveness of swimming pool filters, there will be need for at least a semi-empirical approach to assessing quantitatively the effectiveness of swimming pool filtration. Such assessments would be based on measurements of particle removals, which might be achieved by monitoring the changes in either turbidity or else be monitoring the changes in the number of particles present within the particle size band of interest as water passes through a filter.

## WHAT DO TURBIDITY MEASUREMENTS AND PARTICLE COUNTING TELL US?

Turbidity is caused by particles in suspension, primarily <1 μm diameter (smaller than *Cryptosporidium* oocysts) subject to Brownian motion, resulting in lack of water clarity. Measurement is based on the scattering of light by the particles and expressed as nephelometric turbidity units (NTU), although other units are used. Turbidity impacts on water clarity, and hence safety of bathers, and can only be detected by the naked eye at approximately 4 NTU and above in the depth of water typical of pools in which lifeguards are viewing bathers (WHO 2011). Turbidity can also be associated with reduced disinfection potential either by the presence of particles protecting microorganisms from the action of disinfectants or by consuming disinfectant when organic particulates are oxidised (WHO 2006).

The removal of particles in the sub-micron size range (<1 μm diameter) by filtration can be assessed by measuring the turbidity of water. For larger-sized particles, light obscuration particle counting can be used (Hargesheimer *et al.* 1998). Although both methods have been used to assess drinking water treatment processes (Gregory 2002; Emelko *et al.* 2005), it is important to recognise that they are measuring very different aspects of water quality. In their review of the literature, Emelko *et al.* (2005) concluded that turbidity is not a good surrogate for predicting *Cryptosporidium* removal by filters. For example, LeChevallier *et al.* (1991) reported that turbidity reduction only accounted for 17% of the variation in *Cryptosporidium* removal (expressed as log removals) in 66 surface water treatment plants in North America. However, turbidity monitoring is widely used to measure water treatment performance, and turbidity data sets have been used to develop an index of the robustness of granular filtration, specifically in relation to managing the risk from *Cryptosporidium* (Huck & Coffey 2004). The attraction of turbidity measurements over particle counting is that it is much simpler and cheaper to implement and more amenable to online measurements (Upton *et al.* 2017). Hartshorn *et al.* (2015) used 15 min turbidity data from five surface water drinking treatment works in the UK to quantify turbidity robustness and risk scoring in relation to filtration performance. Stauder & Rodelsperger (2011) carried out a study of swimming pool filtration performance *in situ* in which both turbidity and particle counting measurements were used to monitor the inlet water and filtrate during four weeks of a busy outdoor paddling pool (up to 12,000 visitors per day). Water in this pool was treated using a sand/anthracite dual media filter (filtration velocity 35 m $h^{-1}$), with an aluminium-based coagulant dosed pre-filter. The particles in the 1–100 μm detection range were dominated by the smaller-sized 1–10 μm particles (89% of total particle counts). Removal efficiencies (number of particles removed by the filter relative to the number of particles entering the filter) for the 1–10 μm size class were >98%.

The maximum allowable turbidity for drinking water is 4 NTU (DWI 2017), although municipal supplies should normally achieve <0.5 NTU prior to disinfection (WHO 2011); for pool water, the turbidity should be <0.5 NTU (PWTAG 2017b), which is almost an order of magnitude lower than the limit of detection by the naked eye in

relatively shallow water (WHO 2011). So, while visual assessments of water clarity are useful for assessing gross failures of the filtration system, they cannot be relied upon for detecting changes in turbidity that could be critical for microbiological risk to pool users. The advantages of particle counting compared to the measurement of bulk turbidity in relation to the risk from *Cryptosporidium* oocyst in pools require further investigation.

One important difference between swimming pool filtration and industrial/potable water treatment is that the pool water is constantly recycled through the filtration system. Pool water is also normally cleaner than that treated for drinking water and is subject to large fluctuation in particle content and turbidity during diurnal cycles due to variations in bathing load. During prolonged periods where there are no bathers (and so very little particle input to the pool), such as at night or during a closure period to clean up after an AFR, most of the water will pass through the filter several times with little further particle input. One study reported turbidity of <0.1 NTU during nighttime monitoring of a very busy outdoor paddling pool (Stauder & Rodelsperger 2011). However, there is a dearth of data on how filter removal efficiencies change with variation in bathing load during opening hours and overnight and during prolonged closure periods. It is thought that the recovery which is believed to occur overnight is important in the cleanup of pool water following faecal contamination events, but this requires further investigation. For example, the particle content of filtrate will be contributed to by particles that are detaching from a filter as well by particles that are simply passing through the filter without attaching. As the detachment process will continue throughout the day and night, this might result in reduced apparent filter efficiencies at night when the water being delivered to the filter is very clean. Further detailed work is needed to understand the relationship between the particle content of water and particle removal by packed-bed sand filters.

## WHAT IS THE EFFECT OF FLOW RATE ON FILTRATION?

The flow rate will have no impact on the screening properties of sand filters which are responsible for the removal of particles >100 μm in size. However, the flow rate will affect the efficiency of removal of particles <100 μm in size (and invisible to the naked eye) where the mechanism of removal depends on sedimentation, impaction and Brownian motion (Huisman & Wood 1974). In general, the faster the rate of flow of water through the filter the lower the filtration efficiency. Gregory (2002) concluded that pool filters if operated with good coagulation and at low flow rates (around 10 m $h^{-1}$) could be expected to give 3 $\log_{10}$ reduction (99.9% removal) of oocyst-sized particles. Medium-rate sand filters used for treating pool water normally operate at faster flow rates, typically 25–30 m $h^{-1}$. Gregory (2002) proposed that log removal would be halved with each doubling of flow rate. If true, then increasing the filtration velocity from 10–14 m $h^{-1}$ to 25–29 m $h^{-1}$ would be predicted to decrease the log removal from 3 $\log_{10}$ removal to 1.5 $\log_{10}$ removal (i.e. from 99.9% to 97% removal). However, this hypothesis, which was based on observations of turbidity removal, has not been investigated for the removal of oocyst-sized particles by pool filtration systems under standard operating conditions. Ideally, the aforementioned hypothesis should be tested using a combination of modelling and measurement on full-scale operational pools. The modelled impact of flow rate on filtration efficiency (Figure 1) showed that it is the removal of particles within the 0.1–5 μm size range which was strongly reduced by increasing filtration velocity. Hence, filtration velocity is likely to be a major determinant of filtration efficiency in swimming pool filters with respect to particles sizes relevant to both turbidity and *Cryptosporidium* oocysts removal.

A pilot-scale study reported by Lu & Amburgey (2016) tested the removal of *Cryptosporidium*-size microspheres by sand filtration with a polyaluminium chloride (PAC) coagulant. Microsphere removal was 90% at 30 m $h^{-1}$, but only 50% at the faster flow rate of 37 m $h^{-1}$. This suggests that the sensitivity of *Cryptosporidium* oocyst removal to filtration velocity can be much greater than would be predicted by Gregory (2002).

## WHAT IS THE EFFECT OF FILTER RIPENING AND BACKWASHING?

Cleaning of a filter is achieved by backwashing; this involves reversing the flow through the media bed, fluidising the filter

material and passing the backwash water to waste (WHO 2006). This is normally triggered by one or more of the following: the maximum allowable turbidity value has been exceeded (particularly if the filtrate is being monitored and elevated turbidity is indicative of filter breakthrough occurring), or a specified period of time has elapsed since the previous backwash, or a specified differential is observed between the water pressures at the inlet and outlet of the filter (WHO 2006). The retention of particles following interception and adsorption to the surface of a sand grain is a reversible process, and the nature and performance of the filter bed in terms of both the attachment and detachment of particles changes during the backwash cycle, resulting in changes of filter efficiency (Amburgey 2005). The duration of the backwash and the flow rate during backwashing are critical in determining the effectiveness of the backwash process. The backwashing procedure may include a period of scouring with air to assist in fluidising the filter bed (PWTAG 2017b).

Immediately following backwashing of a filter, a ripening sequence is observed during which the quality of the filter effluent initially deteriorates, then recovers (Amburgey 2005). In general, there will be an initial flush of particles in the filtrate immediately following a backwash, which can be diverted to waste in a rinse procedure (PWTAG 2017b), though many existing pool filtration systems do not have the pipework required for rinsing. This is followed by a more gradual improvement in particle removal as the filter ripens. Once the filter is ripened, there will be a period of near-optimum filtration. As the filter becomes loaded with trapped particles, there will be a further phase of decreased removal and a point is reached where a surge of particles breaks through the filter (Amirtharajah 1988). The duration and extent of each of these stages depend on many factors such as the filter media, filter loading rate, backwash flow rate and duration (Amburgey 2005).

There has been extensive research into the factors affecting filter ripening and in developing backwashing techniques that promote ripening and optimise the amounts of water used in the backwashing and rinsing procedure (e.g. Amburgey 2005). However, this work has been directed primarily at the drinking water industry where the source water being filtered is normally dirtier than in swimming pools, and backwashing is carried out more frequently than is the case for swimming pools. In industrial applications of sand filters, the backwash is normally optimised due to high operational costs of backwashing. This is not generally the case in swimming pool operations where the backwash process is often sub-optimal in terms of factors such as the timing, water velocities used, and whether or not the procedures include air-scouring or rinsing.

Stauder & Rodelsperger (2011) measured peaks in both turbidity and particle counts in the water emerging from the filter (filtrate) in the hour following backwashing. The breakthrough of particles following backwashing could allow *Cryptosporidium* oocysts to return to the pool and pose a risk to swimmers (Ryan *et al.* 2017). More information is needed on the impact of filter ripening and backwash procedures on the performance of granular filters in removing (and retaining) oocyst-sized particles from pool water. For example, because pool water is generally very clean, it is likely that the ripening process will be much slower than in drinking water treatment due to the slow rate of loading of the filter with particles. Hence, there is likely to be a longer period following backwashing before the filter achieves optimal filter performance, but there is very little evidence to show how long this period should be, and the extent to which filter performance is reduced following backwashing. There is also a paucity of examples demonstrating how changes in filtration performance during the backwash cycle vary from pool to pool depending on factors such as the bathing load, the area of filter, and the effectiveness of the scouring/backwashing/rinsing procedures. A further consideration is that if backwashing causes a substantial reduction in filter efficiency for an extended period, what is the optimum scheduling of backwashing when there are multiple filters on a pool in order to achieve the most effective filtration at all times?

## WHAT IS THE EFFECT OF COAGULANTS AND FILTER AIDS?

Water treatment also depends on processes which convert fine particulate material into a form which allows physical separation (destabilisation). This could involve a change in the surface properties, increasing the absorptivity of particles to a filter medium (a filter aid), or causing

aggregation of smaller particles into larger units (a coagulant). The treatment of water with coagulants destabilises particles by compressing the surface electrical double layer (Bratby 2016).

PAC is widely used in commercial swimming pools to enable coagulation of particles prior to filtration (WHO 2006). However, in swimming pool water treatment, if added, PAC is generally dosed without agitation (relying on turbulence to ensure uniform dispersion) and with often only a few seconds residence time between the point of injection and the water arriving at the filter bed. This is in contrast to drinking water treatment where, in general, greater emphasis is placed on providing for effective coagulation pre-filter. This involves a combination of agitation to disperse coagulants uniformly within the water at the appropriate concentration, followed by a prolonged period for coagulation to occur before the water reaches the filter media (Bratby 2016). In a pilot-scale pool study reported by Lu & Amburgey (2016), the removal of *Cryptosporidium*-size microspheres by sand filtration was tested with different coagulants. In the absence of coagulant, the microsphere removal was only 20–63%, but the removal was increased to 99% by the use of aluminium-based coagulants.

In addition to a potential role as a coagulant, materials such as aluminium oxide/hydroxides (as well as other filter aids such as cationic polymers) can act as filter aids which use electrostatic attraction to adsorb particles onto the filter bed. The surfaces of *Cryptosporidium* oocysts and filter media such as sand grains usually carry a negative charge at the near-neutral pH range of pool water (Kim *et al.* 2010) resulting in electrostatic repulsion. Shaw *et al.* (2000) showed that the use of a surface coating of hydrous iron aluminium oxide on filter sand altered the zeta potential from −40 to +45 mV at pH 7.0, reversing the electrostatic repulsion and resulting in a 2.9-fold increase in particle recovery. In view of the short contact time between coagulants and particles in the zone between the point of coagulation injection and the filter bed in swimming pool filtration systems, this alternative mechanism may, in principle, be a more effective approach than pre-filter coagulation. Though there has been extensive research into the use of coagulants and flocculants in industrial applications, there is a dearth of *in situ* studies of swimming pool applications.

The modelling work of Tufenkji & Elimelech (2004) assumed that once particles collide with the collector surface, then attachment takes place (i.e. $\alpha = 1$), which might be reasonable in the case of deep bed filtration where chemical coagulants are used such that the particles are fully destabilised. However, if there is electrical repulsion between the collector surfaces and particles, then attachment will inevitably be reduced. The zeta potential of *Cryptosporidium parvum* oocysts will vary as a function of the ionic strength of the solution, becoming less negative as ionic strength increases (e.g. from 0.1 to 100 mM) due to the expansion of the electrical double layer (Kim *et al.* 2010). The zeta potential of oocysts was near-neutral (−5.9 mV) at ionic strength of 1 mM, whereas the zeta potential of quartz sand media was −51.6 mV at this ionic strength, indicating that there is a significant energy barrier between *C. parvum* oocysts and sand media under normal ionic strength in pools (Kim *et al.* 2010). Despite this physical repulsion as an oocyst approaches a sand particle, oocysts do adhere to silica surfaces, possibly mediated by biomolecules on the oocyst surface (Tufenkji *et al.* 2006). Water treatment with coagulants plays a role by compressing the surface electrical double layer, thereby reducing electrostatic repulsion effects. Electrostatic repulsion effects on particle transport and attachment have proved extremely difficult to model, though it is evident that particle removal is reduced when there is significant repulsion (Ncube *et al.* 2018b).

## WHAT IS THE EFFECT OF DIFFERENT FILTER MEDIA?

There are a number of alternative media available for use in swimming pool filtration, with a trend for traditional 16/30 sand being replaced by a range of artificial glass media and non-glass media such as extruded high-density polyethylene (e.g. OC-1 media). These media can differ widely in a number of characteristics that will affect the filter performance in terms of removal of oocyst-sized particles. These include the porosity (which will affect the average water velocity), the sizes of water channels which will affect the water velocity profiles adjacent to collector surfaces (Bradford *et al.* 2009), and the micro-topography of the interceptor

surfaces will affect the strength of short-range forces. Surfaces with irregularities will tend to provide greater opportunities for contact to occur (Jin *et al.* 2015), and collector shape irregularity may also provide opportunities for straining of oocyst-sized particles at pore necks (Tufenkji *et al.* 2004).

This raises the question of the extent to which the smoothness of the filter media affects its performance in terms of removal of *Cryptosporidium* oocysts. For example, there is evidence that sand grains are more effective than artificial glass in terms of the removal of *Cryptosporidium*-sized particles (Rutledge & Gagnon 2002). Laboratory tests and media comparisons have been carried out under carefully controlled conditions (e.g. Jin *et al.* 2015), but there is no information from *in situ* monitoring in operational swimming pool plant rooms.

## WHAT IS THE MINIMUM DEPTH OF MEDIA BED REQUIRED?

Lawler & Nason (2006) developed an approach to designing filters which combined Equations (1)–(3) above to determine the minimum depth of media bed required to achieve satisfactory filtration. The starting assumption was to use the Tufenkji & Elimelech (2004) model to predict the particle size giving the lowest removal efficiency (i.e. the smallest value for $\eta$, for the filter in question, based on the filtration velocity, temperature and grain size, which in their case was a particle size of 1.5 μm). Lawler & Nason (2006) then identified eight wastewater/potable water treatment filters known to give good filtration and estimated in each case the value of $\lambda$ (Equation (3)), and hence, the filtration efficiency (combining Equations (1) and (2)) predicted for the particle size with least efficient removal.

The calculated filtration efficiencies for these filters known to perform well were found to fall within a narrow band around 25% for the 1.5 μm size particles that were most difficult to remove. Lawler & Nason (2006) proposed that a minimum removal efficiency of 25% for 1.5 μm particles should be the value to use to design the bed depth required for good filtration. Applying the Lawler & Nason (2006) design proposal to swimming pool filters with 700 μm sand grains suggests that the minimum bed depth required for good filtration at the maximum filtration velocity (25 m $h^{-1}$) is about 800 mm, which corresponds to the minimum bed depth recommended for swimming pool filters in the UK (PWTAG 2017b).

The analysis of Lawler & Nason (2006) indicated that filters reputed to perform well had only 20–33% estimated filtration efficiency with respect to the particle size most difficult to remove. This implies that filters that apparently perform well with respect to, say, the removal of turbidity may have low efficiency in the removal of *Cryptosporidium* oocysts. However, the following should be noted:

- This is for a clean filter bed, and the filtration efficiency is likely to improve as the filter ripens. However, in the context of swimming pool filters that are filtering very clean water, it is unclear how long it takes for filters to ripen to the point where there is more effective removal of 4–6 μm size particles.
- This assumes that there is an appropriate addition of coagulants to ensure particle destabilisation (and hence an attachment coefficient $\alpha \approx 1$). If electrostatic repulsion is preventing attachment, then the filter efficiency will be much lower. On the other hand, if the *Cryptosporidium* oocysts are coagulating into larger flocs before arriving at the sand bed, then filtration efficiencies will be higher.

## INFLUENCE OF POOL WATER CIRCULATION ON RISK MANAGEMENT

The conventional indicator used to assess whether water circulation is adequate to maintain good water quality is turnover time. This is described as the time it takes a volume of water equivalent to the volume of the pool to pass through the filtration and circulation system (conventionally expressed in hours). Turnover time is calculated from the ratio of the volume of water in the pool to the circulation rate (conventionally expressed in hours). A pool of 450 $m^3$ volume and a circulation rate of 150 $m^3$ $h^{-1}$ will have a turnover time of 3 h. Pool operational guidelines, e.g. those used in the UK (PWTAG 2017a), commonly use turnover time as the basis for flow rate recommendations. For example, PWTAG (2017a) recommends that a 25 m leisure pool should have a turnover time of no more than

3 h. This does not mean that in a single 3 h period, all of the water in the pool will have passed through the filtration system. Even if the water in the pool is perfectly mixed at all times, theory indicates that only 63% of the water in the pool will pass through the filtration system in one turnover time (Gage *et al.* 1926). In other words, 37% of the water in the pool tank will not be filtered in each turnover.

The current guideline for cleaning up a pool after a suspected *Cryptosporidium* contamination is that the pool is closed and the water allowed to circulate for six turnovers, after which the filters are backwashed and the pool can then be returned to use if the pool operator is confident in their backwashing procedure (PWTAG 2017a). If the pool is perfectly mixed, then after each turnover, 37% of the water in the pool at the start of each turnover period will remain unfiltered at the end of each turnover; so, after six turnovers, the amount of untreated water remaining in the pool will be equivalent to 0.25% of the water originally present in the pool.

If we also take into account the efficiency of the filters, we can estimate the removal of oocysts in each turnover period. For pools with medium-rate filtration (i.e. filtration velocities up to 25 m $h^{-1}$), UK guidelines (PWTAG 2017a) are based on the understanding that effective pool filters will remove 90% of oocysts in a single passage of water through the filter. If so, then a well-mixed pool will lose $63\% \times 90\% = 56.7\%$ of oocysts from the pool in each turnover (in other words, 43.3% of oocysts in the pool tank at the start of a turnover period will remain in the pool after each turnover). As a result, after six turnovers, the number of oocysts remaining in the pool will be 0.66% of those present in the pool at the start of the six-turnover period (i.e. >2 log reduction).

However, even pools designed specifically to have rapid mixing can have poorly mixed dead zones (Lewis *et al.* 2015; Chalmers *et al.* 2016), and there is, therefore, considerable uncertainty about the proportion of pool water that will pass through the filtration and circulation system in any one turnover time. For any particular pool, the proportion of filtered and unfiltered water at a particular point in time is unknown. For example, if the impact of poorly mixed dead zones, and any short-circuiting of flows between inlets and outlets, was to increase the percentage of water that is untreated per turnover from 37% to 47%, then the number of turnovers would have to increase to eight to achieve the same result as a pool assumed to be perfectly mixed (Table 2).

A further complication is that after an AFR, any oocysts present in the faecal material will not be uniformly distributed around the pool. If the oocysts are located within a dead zone, this could have the effect of substantially reducing the percentage of oocysts that pass through the filtration system per turnover. Indeed, dead zones are usually located where the wall of the pool meets the floor, which is precisely where AFRs are most likely if the bather responsible is holding the handrail at the time. These considerations represent a major uncertainty and challenge in managing risk from *Cryptosporidium* in swimming pools.

Furthermore, the filtration system might be less effective than is presumed. For example, if the filters are only removing 50% of oocysts in a single pass of water rather than 90% (as in the case reported by Lu & Amburgey 2016), then this

**Table 2** | Calculated removal of oocysts from pool water after six turnovers assuming either a perfectly mixed pool with 63% of water treated per turnover (best mixing), or a less well-mixed pool with only 53% of water treated per turnover (poor mixing), and filters operating at either 90% or 50% efficiency

| Mixing/filtration scenario | % of pool water treated (untreated) per turnover | Filter efficiency[a] | % of oocysts removed (remaining) in pool water after six turnovers | Number of turnovers required[b] | Number of oocysts per 37 mL after six turnovers[c] |
|---|---|---|---|---|---|
| Best mixing/best filtration | 63 (37) | 90 | 99.3 (0.7) | 6 | 5 |
| Poor mixing/best filtration | 53 (47) | 90 | 98 (2.0) | 8 | 15 |
| Best mixing/poor filtration | 63 (37) | 50 | 89.7 (10.3) | 13 | 98 |
| Poor mixing/poor filtration | 53 (47) | 50 | 84.2 (15.8) | 16 | 121 |

[a]% of oocysts removed by filters from treated pool water.
[b]Required to achieve same removal as a best-case scenario, i.e. 63% of water treated per turnover and filters operating at 90% efficiency.
[c]Based on the initial contamination of 20,000 oocysts $L^{-1}$ estimated for a typical AFR into a 450 $m_3$ pool (Gregory 2002), equivalent to 740/37 mL, using the value of 37 mL as the average volume of water swallowed by non-adults during a 45 min pool session of swimming (Dufour *et al.* 2006).

would increase the number of turnovers required to achieve similar removal of oocysts from six to 13 turnovers (Table 2). A combination of poor mixing (47% of water untreated per turnover) and poor filtration (50% oocyst removal efficiency) would increase the number of turnovers required to achieve a similar removal of oocysts from six to 16 (Table 2).

The first data row in Table 2 show that the assumed 'best practice case' (a pool with perfect mixing and 90% filtration) removes 99.3% of oocysts during the course of six turnovers (0.7% of the original oocysts remain in the pool water). Increasing the percentage of untreated water remaining in the pool per turnover from 37% to 47% (to take account of factors such as dead zones, uneven oocyst distribution and any short-circuiting of flows between inlets and outlets) approximately trebles the number of oocysts remaining in the pool. Reducing the filter removal efficiency from 90% to 50% has a much more dramatic effect, resulting in over 10% of the original oocysts remaining in the pool after six turnovers. Under these circumstances, the residual oocyst concentration in the pool water could result in pool users ingesting considerably more oocysts than the dose (1–100 oocysts) capable of causing infection (Ryan *et al.* 2017).

## KNOWLEDGE GAPS AND CONCLUSIONS

This review of the processes by which filters remove particles of different sizes from pool water has highlighted a number of areas where significant gaps exist in our understanding of the factors controlling the risk to pool users from particulate material, and specifically for *Cryptosporidium* oocysts. Though we have used the removal of oocysts following an AFR to illustrate conclusions, the discussion about the factors affecting filtration apply equally to maintaining water in good condition during normal operation. We have shown that for a 'best-case scenario' pool, the UK guidelines on dealing with faecal contamination (PWTAG 2018) will significantly reduce the numbers of oocysts in the pool and minimise the risk of infection while still providing operators with a realistic course of action. However, if there is less than optimal filtration, the absence of coagulation in pool water treatment or a poorly mixed pool current guidelines may fail to mitigate the risk.

In view of the fact that commercial swimming pools do not lend themselves to experimentation, advances in our understanding are most likely to be gained from a combination of modelling and *in situ* measurement. Mass balance models such as that used by Stauder & Rodelsperger (2011) can underpin our understanding of the behaviour of particulate material (e.g. *Cryptosporidium* oocysts) in pool plant systems and, when combined with the measurement of turbidity and particle counting, offer opportunities for the development of tools which integrate our understanding of both the hydraulics of the pool and the filtration efficiency to assist in assessing risk, e.g. from *Cryptosporidium* oocysts in pool water.

A number of specific areas that require further investigation have been identified in this review:

1. Validation of existing models with full-scale swimming pool studies will also enable quantification of filtration effectiveness in relation to different particle size fractions. Full-scale studies on operational pools are also necessary to investigate factors affecting the filtration, including the use of coagulants and filter aids, as well as the employment of different filtration media types.
2. The understanding of the removals of different particle size fractions needs to be improved via a detailed study of the nature and behaviour of these particles, together with better understanding of the value and limitations of turbidity measurements with respect to the removal of *Cryptosporidium* oocyst-sized particles.
3. Impact of circulation rate, particle loading, backwash frequency, backwash flow rate and time following backwash on the filtration of different size particles requires a detailed investigation that should include continuous monitoring of filter performance through a backwash cycle at a range of pool sites.
4. The effect of the many factors that affect the delivery of oocysts/turbidity from the pool to the filtration system needs to be quantified. These include:
   - the location and number of filter inlets and outlets, and how these impact on the mixing characteristics of the pool;
   - moveable floors;
   - the ratio of sump flow to surface draw-off;
   - bathing load and distribution within the pool;

- the likely input of particulate material from bathers, and how this is affected by factors such as age, pre-swim hygiene, and whether pools are indoor or outdoor.

Once this information starts to become available, it then becomes possible to develop a risk-based approach to managing swimming pool water, particularly for the management of waterborne disease outbreaks, along the lines proposed for drinking water (e.g. Havelaar 1994; Petterson & Ashbolt 2016). For example, HACCP (Hazard Analysis and Critical Control Points) provides a framework for integrating current scientific knowledge of microbiological hazards into a quality management system based on monitoring of critical control points in the water treatment process and has been applied to the production of drinking water in Belgium (Dewettinck *et al.* 2001). Such an approach would enable the outputs of the research identified in this review to be used in ways that will be of direct benefit and reassure operators, regulators and users of the safety of commercial swimming pools.

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