

Removal of Waterborne Particles by Electrofiltration: Pilot-Scale Testing

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Abstract

Theoretical analysis using a trajectory approach indicated that in the presence of an external electric field, charged waterborne particles are subject to an additional migration velocity that increases their deposition on the surface of collectors (e.g., sand filter). Although researchers conducted bench-scale experiments to verify the effectiveness of electrofiltration, few studies have reported on the applications of electrofiltration in larger scale facilities. In this study, a prototype pilot-scale electrofiltration unit, consisting of an acrylic tank (0.3×0.3×1.2 m) with vertically placed stainless steel mesh electrodes embedded in a sand filter was tested at a local drinking water plant. Presedimentation basin water was used as the influent with a turbidity ranging from 12 to 37 NTU. At an approach velocity of 0.84 mm/s, an electrode voltage at 8 and 12 V increased the particle removal coefficient pC^* [defined as $-\log(C_{out}/C_{in})$] to 1.79 and 1.86, respectively, compared to 1.48 when there was no electric field. Reducing the approach velocity from 0.84 to 0.42 mm/s increased pC^* from 1.48 to 1.64, when the electrode velocity was 16 V. Repetitive experiments were conducted and the results were in agreement with those calculated by a theoretical trajectory analysis. The electrofiltration process was demonstrated to be more effective for removal of smaller particles ($<4\ \mu\text{m}$), the size range of many waterborne bacteria. A voltage of 8–12 V was shown to be the most cost-effective range, considering both the energy cost and filtration performance. The findings from this pilot-scale study are important for full-scale applications of the electrofiltration technology.

Key words: water treatment; drinking water; waterborne particles; pathogens; electrofiltration; trajectory analysis; collection efficiency

Introduction

EFFECTIVE REMOVAL OF WATERBORNE PARTICLES is an essential issue in the drinking water treatment process. The particle removal mechanisms can be classified as gravity separation (sedimentation or flotation) and filtration. To enhance the removal efficiency, chemical additives (e.g., iron or aluminum salts, polymers) are widely utilized to eliminate the negative charges carried by particles in nature. After charge neutralization, these particles are ready to attach to each other or to other surfaces to form larger settleable particles, called

flocs, which settle out or are removed during the water filtration process (AWWA, 1997). The filtration concept is quite simple. Influent water passes through a granular sand layer (or other filter media), particles are captured by the media, and the water exits as effluent with fewer particles. The prime removal of small particles is obtained in the filtration process, especially for parasites and other pathogens. However, outbreaks of some waterborne pathogens, such as *Cryptosporidium parvum* are well documented in the United States (Hayes *et al.*, 1989; Fox and Lytle, 1996; Harter *et al.*, 2000).

The chemical additives enhance particle removal efficiency; however, they also increase the volume of solids residuals that are formed as a byproduct of drinking water treatment. The landfill expense of residuals disposal amounts to millions of dollars per year for the water treatment industry. Advanced technologies are necessary to enhance the particle removal

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efficiency in the drinking water treatment process while minimizing the dosage of chemical additives. One potential alternative method is electrofiltration, which applies an external electric field inside the filtration layer to enhance particle removal. The application of electric field on particle removal is widely applied in air-particle systems, such as electrostatic precipitators (ESPs). However, in the liquid-particle system, it has received much less attention, probably because of the higher viscosity of water and limitation in applying high voltage in water media.

In the electrofiltration process, the primary mechanisms of particle deposition on collector surfaces include interception, inertial impaction, electrostatic attraction, and gravitational settling (Tien, 1989). In the absence of any external field, the particles entrained by the fluid pass around the collector. Only particles that are close enough to the collector surface can be captured due to short range van der Waals forces. In the aqueous phase, most of the natural particles, including biological colloids, carry some negative surface charge. Similar to the effect of gravity field, applying an external electric field in the filter media increases the probability of particle deposition on the collector surface by the additional migration velocity of these charged particles, as shown in Fig. 1. Unlike in the conventional flocculation process, charge neutralization is not necessary to enhance the particle removal efficiency in the electrofiltration process. Therefore, the dosage of chemical additives can be minimized, reducing residuals generation and thereby decreasing the cost of chemicals and waste disposal. Electrofiltration can potentially benefit drinking water treatment plants under one of the following scenarios: (1) for direct filtration (rapid mix and flocculation with no sedimentation) and in-line filtration (rapid mix with no flocculation and sedimentation) plants, the electrofiltration process may reduce or eliminate the need for coagulants; (2) for complete treatment plants, electrofiltration can reduce the amount of coagulants needed for treatment and thereby reduce residuals formation.

Electrofiltration has been utilized in the process of cake filtration, residuals dewatering, and removal of particles in

low ionic strength aqueous suspensions (Lockhart, 1983; Ptasiński and Kerkhof, 1992; Zhang *et al.*, 2000). Theoretical analysis regarding the beneficial effect of electrofiltration has been reported in the literature (Judd and Solt, 1989; Kulkarni *et al.*, 2005). Bench-scale experiments conducted by Kulkarni *et al.* (2005) using an external direct current electric field applied on a cylindrical filter column resulted in significant improvement in the removal of test particles (Kaoline and polystyrene latex beads) and *Cryptosporidium* oocysts in three different types of media. Zhang *et al.* (2000) studied electrofiltration of hydrosols in lab-scale fixed-bed filters and found enhanced particle removal due to the electric field. Genc and Tosun (2002) conducted experimental studies on the effect of electrode configuration in electrofiltration and found that the mesh electrode configuration performed better than the spot and foil configurations in terms of the percent gain in volume filtrate alone. A limited number of literature reported pilot-scale electrofiltration through membranes (Hofmann *et al.*, 2006; Weigert *et al.*, 1999). Little literature, however, has been reported on the application of electrofiltration for drinking water treatment in pilot-scale or full-scale facilities.

The objective of this study was to investigate the effectiveness of electrofiltration technology for waterborne particle removal in a pilot-scale unit. The experiments were conducted at the site of a local water treatment plant in St. Louis, Missouri. The effects of electrode voltage and filtration flux were examined. The experimental particle removal efficiencies were obtained and compared with the theoretical predictions based on trajectory approach analysis (Kulkarni *et al.*, 2005). The findings from this pilot-scale study will be of great importance for full-scale applications of electrofiltration technology.

Experimental Setup and Methods

A pilot-scale electrofiltration unit (E-filter) was designed and constructed to simulate the filtration process at a local drinking water treatment plant. A schematic of the pilot-scale unit is shown in Fig. 2a. The influent water was pumped to the water tank from the presedimentation basin of the water treatment plant. The presedimentation basin was treated with cationic polymers and ferric sulfate, followed by a softening basin that was treated with additional cationic polymer and ferric sulfate along with lime, chlorine, fluoride, and ammonia. Activated carbon was added seasonally to the softening basin and also occasionally to the presedimentation basin. The reason to choose influent water from presedimentation basin in this work was because it contained less chemical additives and a suitable zeta potential of the waterborne particles, as explained later in the Results section of this article.

The E-filter consisted of an acrylic filtration column that was 1.2 m high, with a cross-section area of 0.09 m², as shown in Fig. 2a. From the bottom to the top, the media layers included a coarse gravel layer (10-cm deep with an average grain size of 10 mm), a fine gravel layer (25-cm deep with an average grain size of 4 mm), and a sand layer (50-cm deep with an average grain size of 0.51 mm), respectively. All of the sand and gravel were the same media currently utilized in the local drinking water treatment plant. Twelve stainless steel mesh electrodes (0.3×0.6 m) were embedded in the E-filter with 2.5 cm spacing (as shown in Fig. 2b). The electrodes were alternatively connected to the positive and negative output of

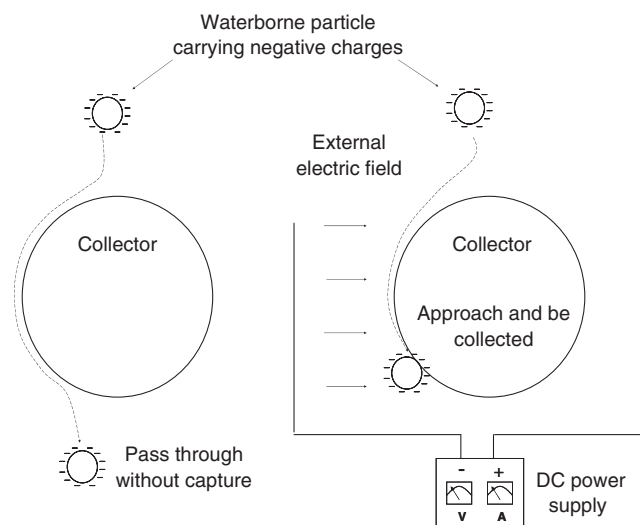


FIG. 1. Schematic diagram of particle motion with and without external electric field.

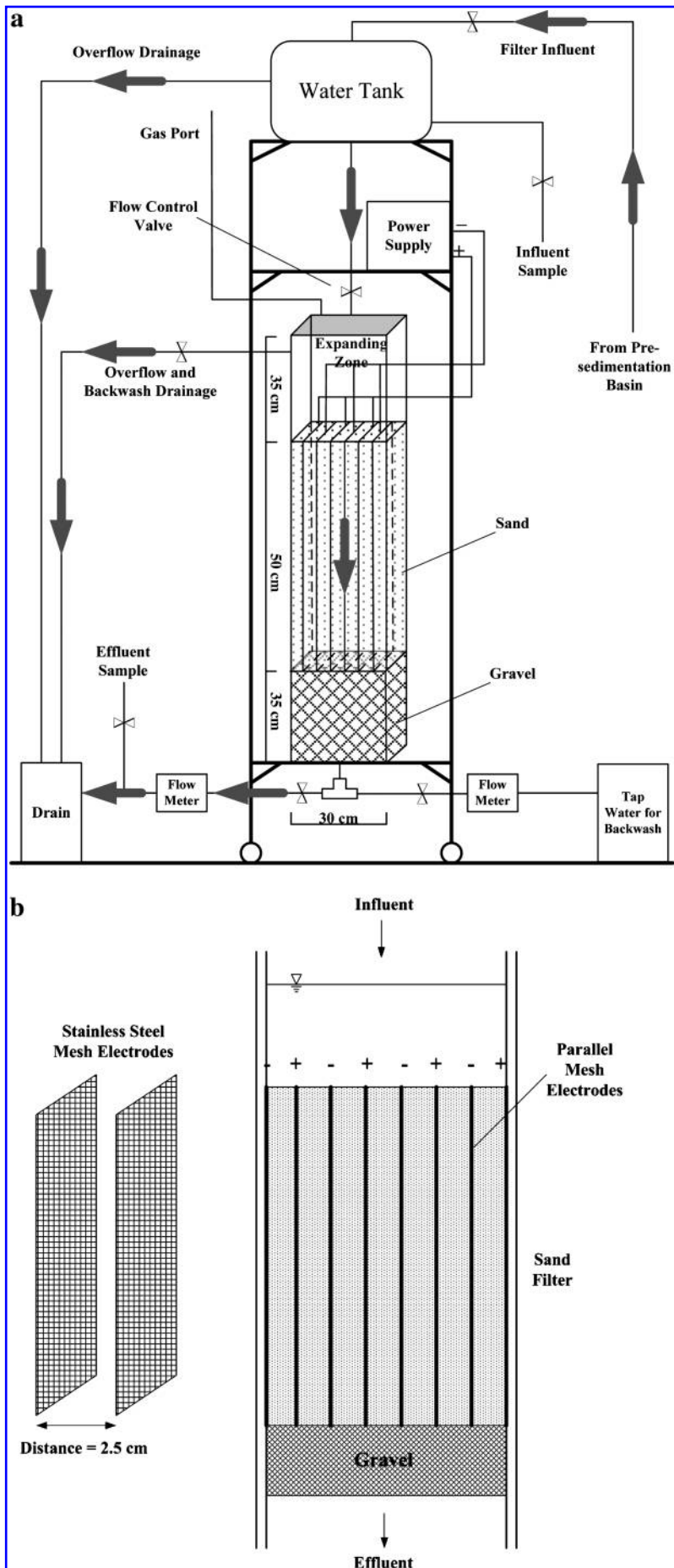


TABLE 1. EXPERIMENTAL PLAN FOR ELECTROFILTRATION TESTS

Test	Approach velocity V_w (mm/s)	Applied voltage U (V)	Description
1	0.42	$0 \leftrightarrow 8$	Power on and off alternatively
2	0.42	$0 \leftrightarrow 16$	Power on and off alternatively
3	0.84	$0 \leftrightarrow 24$	Power on and off alternatively
4	$0.42 \rightarrow 0.63 \rightarrow 0.84$	16	Consecutively increasing the approach velocity
5	0.84	$0 \rightarrow 8 \rightarrow 12 \rightarrow 16$	Consecutively increasing the voltage

During each test the filter was backwashed for 15 min between the changes in operating conditions.

a power supply (BK Precision, Yorba Linda, CA; VSP6020). A water tank located on the top of the frame was utilized as a reservoir to maintain a constant hydraulic head, approximately 0.9 m of water, above the sand filter. The influent flow rate was controlled by a valve between the water tank and the filter. The filtered water (E-filter effluent) passes through a flow meter, which monitored both the flow rate and the total flow, and was then discharged to drain. During the interval of each run, the E-filter was backwashed for 15 min by tap water to remove the captured particles.

The experimental plan is summarized in Table 1. Tests 1 to 3 were designed to prove the concept of electrofiltration, that is, to verify that the electric field enhances particle capture in a large-scale unit. For each of these tests, while keeping the influent approach velocity constant, the power was turned on and off alternatively every 2 h so that the enhancement due to the electric field can be determined. The hydraulic residence time in the filter was approximately 10 and 20 min when the approach velocity was 0.84 and 0.42 mm/s, respectively. The testing duration (2 h) was appropriate because it was much longer than the maximum hydraulic residence time (20 min). The filter was backwashed for 15 min (approach velocity was approximately 5 mm/s) between the switch of each power on/off condition. Each test (i.e., the power on/off cycle) was repeated for three times. Test 4 investigated the effect of approach velocity on electrofiltration while applying a constant voltage across the electrodes. Test 5 explored the effect of electrode voltage while maintaining a constant approach velocity. For all the tests, both influent and effluent samples (sample size = 3) were collected and analyzed in terms of turbidity, zeta potential, and particle counts. The turbidity was measured using a Hach 2100AN turbidimeter. The zeta potential was measured using a Zetasizer Nano ZS (Malvern Corp., Malvern, PA). The number of waterborne particles in the size range of 2–100 μm was measured using an optical particle counter (Particle Measuring Systems, Boulder, CO).

Theoretical Analysis

A theoretical analysis using a trajectory approach was previously reported by Kulkarni *et al.* (2005) to calculate the critical trajectory around a spherical collector in the presence of an electric field from the solution of a trajectory equation. The equation combines the effects of the external electric force, the gravity force, the viscosity drag force, the electric double layer force, and the London–van der Waals force. In this study, the theoretical approach developed by Kulkarni *et al.* (2005) was adopted to calculate the external electric force, the particle migration velocity due to electric force, the single collector efficiency, and the overall particle collection efficiency of electrofiltration.

A charged colloidal particle in the presence of an electric field experiences a force proportional to its charge and the electric field. As the electrodes are placed vertically in this study, the strength of the electric field E can be calculated from the electric potential, U , and the spacing of the electrodes, L_s .

Subsequently, the external electric field force that a particle experiences is expressed as

$$F_E = qE \quad (1)$$

where q is the particle surface charge that can be derived from the measured zeta potential of the colloidal particles:

$$q = 3\pi d_p \varepsilon_D \zeta \quad (2)$$

where d_p is the particle size, ε_D is the dielectric permittivity of water ($\sim 80 \varepsilon_0$), and ζ is zeta potential. The migration velocity of a particle due to the electric field can be calculated as

$$V_m = \frac{F_E}{3\pi\mu d_p} \quad (3)$$

where μ is the viscosity of water. The nondimensional external electric field force, the nondimensional gravity force, and the nondimensional van der Waals force can be calculated, respectively, as follows:

$$N_{EF} = \frac{V_m}{V_w} = \frac{qE}{3\pi\mu d_p V_w} \quad (4)$$

$$N_G = \frac{d_p^2 (\rho_p - \rho_w) g}{18\mu V_w} \quad (5)$$

$$N_{LO} = \frac{4H}{9\pi\mu d_p^2 V_w} \quad (6)$$

where V_w is the approach velocity of water, and ρ_p and ρ_w are particle and water density, respectively. Once the above nondimensional forces are calculated, the single collector efficiency, η_{single} , can be solved from the trajectory equation (Kulkarni *et al.*, 2005).

Finally, the overall particle removal performance can be expressed using a log-removal coefficient, pC^* :

$$pC^* = -\log\left(\frac{C_{\text{out}}}{C_{\text{in}}}\right) \quad (7)$$

where C_{in} and C_{out} are particle concentrations at the filter inlet and outlet, respectively. The value of $-\log(C_{\text{out}}/C_{\text{in}})$ can be

TABLE 2. MEASURED TURBIDITY AND ZETA POTENTIAL FOR RIVER WATER AND PRESEDIMENTATION BASIN WATER

Water sample	Turbidity (NTU)				Zeta potential (mV)		
	Test 1	Test 2	Test 3	Test 4	Test 1	Test 2	Test 4
River water	53 ± 6	233 ± 22	115 ± 13	481 ± 45	−16.5 ± 1.2	−18.3 ± 1.1	—
Presedimentation basin water (E-filter influent)	12.4 ± 0.8	32.2 ± 4.2	12.0 ± 1.1	37 ± 5.5	−15.9 ± 0.3	−17.1 ± 0.7	−18.6 ± 0.5

estimated from the following relation (Yao *et al.*, 1971; Kulcarni *et al.*, 2005):

$$-\log\left(\frac{C_{out}}{C_{in}}\right) = \frac{\eta_{single}}{\frac{4}{3} \frac{a_c}{(1-\varepsilon)L_H}} \quad (8)$$

where η_{single} is the single collector efficiency, a_c is the radius of sand granule, ε is the filter media porosity, and L_H is the depth of sand filter.

Results and Discussion

Turbidity and zeta potential

Table 2 lists turbidity and zeta potential results for samples of river water and presedimentation basin water. The turbidity of river water varied from test to test, mainly due to seasonal changes. The turbidity of presedimentation basin water also changed with corresponding fluctuations in river water, but generally was one order of magnitude lower than that of the river water. The zeta potential, however, did not vary much from test to test, and it only decreased slightly from the river water to the presedimentation basin water. Because zeta potential is a critical parameter representing the surface charge of waterborne particles, a higher absolute value of zeta potential normally corresponds to a higher efficiency of electrofiltration. Hence, the presedimentation basin water, which generally has a zeta potential greater than −15 mV (Table 2), was selected as influent water to the E-filter.

Proof of concept tests

The turbidity and particle count results from Tests 1 to 3 are shown in Table 3. Due to the filtration characteristics of the sand filter (zero applied voltage), the turbidity and particle count of the effluent samples were lower than those of the

influent. Applying a voltage to the electrodes further reduced the turbidity and particle count of the effluent. The voltage applied in Tests 1, 2, and 3 reduced the particle count to ~59, 63, and 55%, respectively, of the level without power. Because each of the tests was repeated three times, the results confirmed that the electrofiltration process is effective in removal of waterborne particles. Table 3 also shows that particles in the size range of 2–4 μm , accounted for approximately two-thirds of the total influent particle count. In addition, compared to the no-power condition, the major reduction in particle count due to the electric field was also in the size range of 2–4 μm . Thus, the electrofiltration process shows promise for removal of waterborne pathogens (e.g., bacteria) because most of bacteria are in the micrometer size range. The performance of electrofiltration as a function of particle size can be more clearly observed by plotting pC^* , as shown in Fig. 3. It demonstrates that the electric field helped particle removal for all sizes of particles. The fluctuations in pC^* for the case of “no power” in the range of 20–30 μm in Test 2 and in the range of 10–30 μm in Test 3 may be due to the system error of the optical particle counter. The raw data on effluent particle counts showed extremely small particle concentrations in this size range, and thus, a small measurement error in effluent particle count would cause a relatively large variation in pC^* as it is calculated by a log-function. However, there was no obvious fluctuation in pC^* for the two “with-power” cases. This may be because the enhanced particle removal minimized the error induced from the effluent particle count measurement.

Effect of approach velocity

Figure 4 demonstrates the effect of approach velocity of water on effluent turbidity and particle removal performance (Test 4). The approach velocity of water was increased from

TABLE 3. TURBIDITY AND PARTICLE COUNT RESULTS FOR TESTS 1 TO 3

Test	Water sample	Turbidity (NTU)	Particle count ($\times 10^3/\text{mL}$)		
			Total	2–4 μm	>4 μm
Test 1	Influent	12.4 ± 0.8	41.6 ± 2.7	34.3 ± 2.6	7.34 ± 0.44
	Effluent (0 V)	0.66 ± 0.07	2.79 ± 0.25	2.39 ± 0.23	0.40 ± 0.03
	Effluent (8 V)	0.47 ± 0.10	1.65 ± 0.17	1.35 ± 0.12	0.30 ± 0.03
Test 2	Influent	32.2 ± 4.2	55.0 ± 4.9	37.1 ± 4.3	17.8 ± 1.3
	Effluent (0 V)	8.25 ± 0.93	2.47 ± 0.29	2.00 ± 0.16	0.46 ± 0.05
	Effluent (16 V)	5.05 ± 0.33	1.54 ± 0.17	1.25 ± 0.13	0.29 ± 0.04
Test 3	Influent	12.0 ± 1.1	31.8 ± 2.4	22.1 ± 2.1	9.69 ± 0.47
	Effluent (0 V)	0.97 ± 0.15	2.02 ± 0.31	1.44 ± 0.27	0.59 ± 0.12
	Effluent (24 V)	0.95 ± 0.08	1.12 ± 0.13	0.85 ± 0.11	0.27 ± 0.05

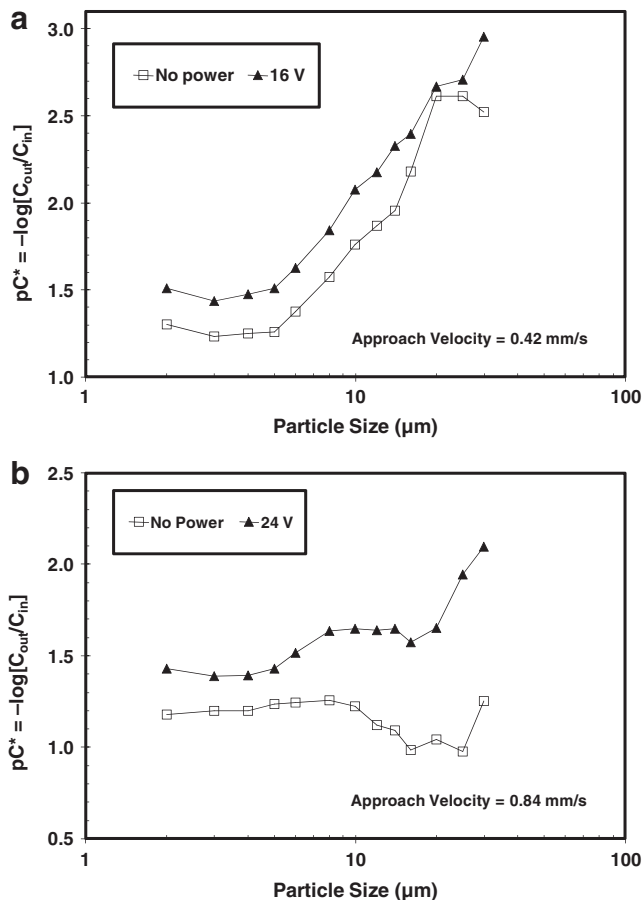


FIG. 3. Comparison of particle removal performance with or without the electric field under different operating conditions: (a) $V_w = 0.42$ mm/s, $U = 16$ V (Test 2) and (b) $V_w = 0.84$ mm/s, $U = 24$ V (Test 3).

0.42 to 0.63 and to 0.84 mm/s consecutively, whereas the influent turbidity was relatively stable. At each velocity level, applying a voltage at 16 V decreased the effluent compared to the no-power condition. At 0.42 and 0.84 mm/s, the electric field reduced the effluent turbidity from 10 to 4 and from 15 to

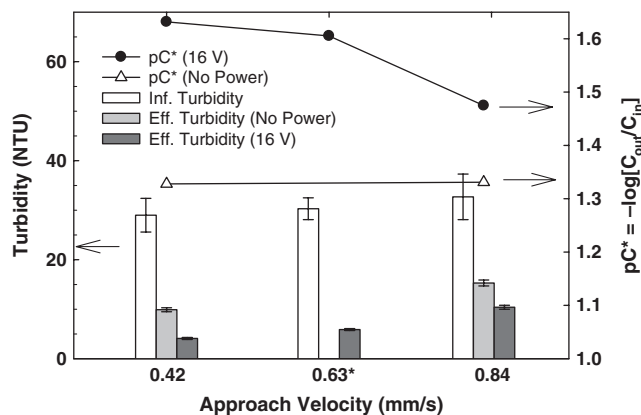


FIG. 4. Effect of filtration approach velocity on E-filter effluent turbidity and particle removal performance (Test 4). The error bars represent one standard deviation. *Effluent turbidity was not measured at the condition of no-power and 0.63 mm/s.

10 NTU, respectively. In addition, the effluent turbidity decreased as the velocity decreased. With no power, pC^* was at a constant level at 1.33. With a 16 V voltage, pC^* was improved to 1.64, 1.61, and 1.48 at the velocity of 0.42, 0.63, and 0.84 mm/s, respectively. It is obvious that a lower approach velocity decreased the effluent turbidity and increased the particle removal. This is because a lower velocity corresponds to a longer residence time inside the E-filter so that the charged particles can migrate a longer distance horizontally, and hence, are more likely to be collected. This set of tests again verified the effectiveness of the applied electric field.

Effect of electrode voltage

Figure 5 shows the effect of electrode voltage on electrofiltration, whereas the approach velocity was maintained at 0.84 mm/s (Test 5). The power was off at the beginning and then the electrode voltage was increased from 8 to 12 V and to 16 V consecutively. It is observable that the effluent turbidities for all the power-on conditions were lower than that for the power-off condition. Because of the fluctuations in influent turbidity during the testing period, it is hard to observe the improvement in filtration due to the increased voltage by just comparing the absolute values of the effluent turbidity. However, the trend is much clearer from the result of pC^* . When the power was off (0 V), pC^* was equal to 1.48. The value of pC^* increased to 1.79, 1.86, and 1.88 as the electrode voltage increased to 8, 12, and 16 V, respectively.

To better quantify the effectiveness of electrofiltration as a function of voltage, the data of pC^* for 2- μm particles in both theoretical predictions and experimental trials are plotted in Fig. 6. When there was no applied voltage, the experimental pC^* was 1.18. As the electrode voltage increased to 8, 12, and 16 V, pC^* increased to 1.65, 1.88, and 1.89, respectively. The value of pC^* predicted by the theoretical model [calculated by Eqs. (7) and (8)] followed a similar trend but with a generally higher value compared to the experimental results. As the voltage increased from 0 to 8, 12, and 16 V, the theoretical pC^* increased from 1.72 to 2.12, 2.58, and 4.26, respectively. There is a significant difference between the experimental and theoretical results for the condition at 16 V. The experimental pC^* at 16 V was almost the same as that at 12 V, but the theoretical

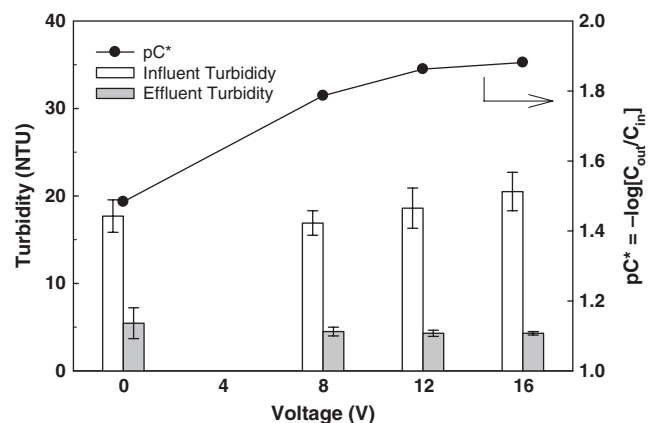


FIG. 5. Effect of electrode voltage on E-filter effluent turbidity and particle removal performance (Test 5). The error bars represent one standard deviation. Approach velocity = 0.84 mm/s.

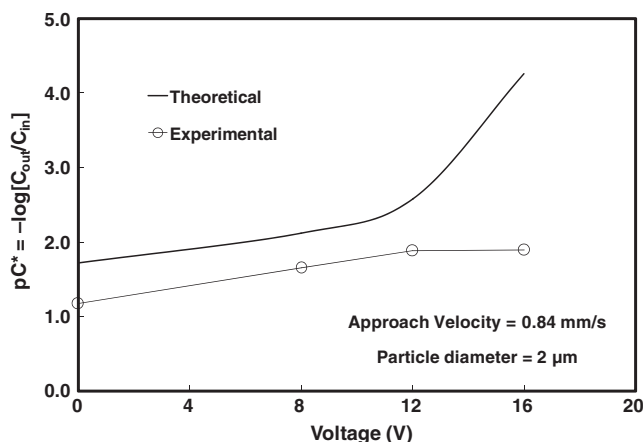


FIG. 6. Comparison of theoretical and experimental removal performance (for 2- μm particles) as a function of electrode voltage.

model predicted a significantly higher pC^* at 16 V than that at 12 V. It may be because the theoretical model assumed that the spherical collector was placed in an electric field of uniform strength; however, the practical electric field distribution could be complicated due to the presence of granular media and the electric double layer around the electrode surface, which leads to weaker electric fields in the interior of granular media (Kulkarni *et al.*, 2005). Another possible reason is that the theoretical model assumed 100% attachment efficiency, that is, a particle that has transported to the collector surface will be collected. Particles may be detached from the collector in practical conditions, and thus the lower attachment efficiency may contribute to lower overall removal observed in the experimental results. Judd and Solt (1989) also noted that the capture of particles due to electric field was lower by a factor of 2–3 compared to that predicted by the theory.

Effect of particle size

It is important to examine the removal effectiveness of electrofiltration for particles with different diameters because waterborne pathogens have a wide range of sizes with most bacteria in the range of 0.3–10 μm (Hinds, 1999). Figure 7 shows both the theoretical and experimental pC^* as a function of particle size in the range of 0.3–4 μm . When there was no applied voltage on the electrodes, the theoretical analysis predicted that pC^* dramatically decreases from 2.84 to 0.65, as the particle size decreases from 4 to 0.3 μm . This indicates that smaller particles are more difficult to be captured during a conventional filtration process. By contrast, when a 12 V voltage was applied on the electrodes, the theoretical pC^* exhibited a U-shape curve. The lowest theoretical pC^* appears at 2 μm . For particles smaller than 2 μm the theoretical pC^* increases sharply as particle size decreases, and for particles larger than 2 μm the theoretical pC^* slightly increases. The two curves of theoretical pC^* (0 and 12 V) seem to converge at the point of 4 μm , implying that electrofiltration would be less effective for particles larger than 4 μm . This theoretical prediction result clearly demonstrates a significant enhancement in the removal of smaller particles (e.g., waterborne bacteria) by electrofiltration compared to a

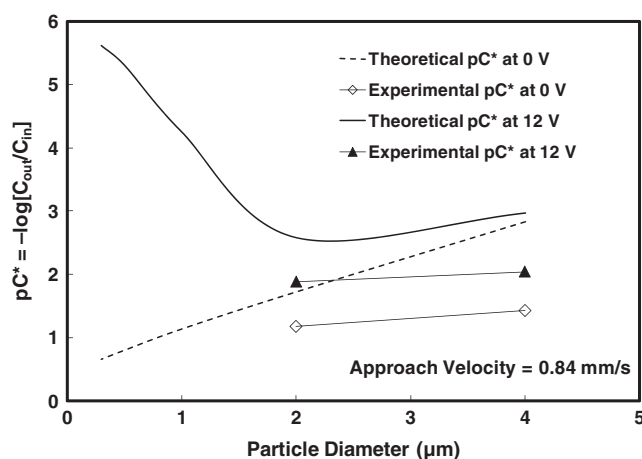


FIG. 7. Comparison of theoretical and experimental removal performance as a function of particle size (for particles smaller than 4 μm).

conventional filtration process. The theoretical model also over predicted the removal performance compared to the experimental results possibly due to the reasons previously described. The experimental data at 2 and 4 μm clearly showed an improvement in pC^* due to the electric field (increasing from 1.18 to 1.88 for 2 μm particles, and from 1.43 to 2.04 for 4 μm particles). The results agreed with those in Table 3, showing that the electrofiltration process is particularly efficient in removing 2–4 μm particles. However, due to the limitation of the measurement range of the optical particle counter, particles less than 2 μm were not able to be counted. Future research using submicrometer surrogates and more sensitive analytical instruments is needed to verify the experimental enhancement of electrofiltration for submicrometer particle removal.

Estimate of energy consumption for electrofiltration

The energy consumption of electrofiltration can be estimated from the voltage and amperage data. Table 4 lists the energy consumption at different applied voltages, based on the approach velocity of 0.84 mm/s. As the voltage increases, the energy consumption increases as a square of the applied voltage. The energy requirement for operating drinking water plants is a major operating cost. The average drinking water plant uses around 370 Wh/m^3 , and this can be higher for plants using advanced treatment technologies (Burton, 2006). The energy consumption at 16 V approaches the edge of economic feasibility for practical

TABLE 4. ESTIMATED ENERGY CONSUMPTION FOR ELECTROFILTRATION

Voltage (V)	Current (A)	Power (W)	Energy consumption (Wh/m^3)
8	3.4	27	100
12	6.2	74	273
16	8.2	131	482
24	14	336	1,235

Approach velocity = 0.42 mm/s.

applications. Based on the results in Fig. 5, particle removal efficiency at 16 V did not improve significantly compared to 12 V. Hence, 8–12 V may be the most cost-effective range in electrofiltration. The voltage parameters appropriate for use in a full-scale application are yet to be confirmed in a long-term test.

It should be noted that because this work used pre-sedimentation basin water as the filter influent, the influent turbidity (12–37 NTU) was much higher than that of the practical filter plant (typically less than a few NTU due to treatment with more chemicals additives). It implies that the filtration capacity would be saturated in a much shorter time, which makes a long-term testing not suitable under such experimental conditions. Future work is needed to test the long-term performance of the E-filter with a lower turbidity influent.

Conclusions

The pilot-scale E-filter unit with vertical mesh electrodes embedded in a sand filter is effective for removal of waterborne particles. Both the turbidity and the particle count of the filter effluent were reduced when a voltage was applied across the electrodes, compared to the results with no power (0 volts applied). When the electrode voltage was 12 V, the particle removal coefficient pC^* reached 1.86, compared to 1.48 when the power was off. The parameters that affect the effectiveness of electrofiltration include water approach velocity and electrode voltage. It was found that a lower approach velocity leads to higher removal efficiency, due to a longer residence time for a particle to migrate in the electric field. A higher voltage, corresponding to a stronger electric field, also resulted in higher particle removal. Both theoretical analysis and experimental results indicated that the conventional filtration process lacks the capability of effectively removing small particles ($<4\ \mu\text{m}$), the sizes of which represent many waterborne pathogens (bacteria). This study showed that the electrofiltration technology significantly enhances the capture of these small particles, which is beneficial to the drinking water industry. The estimated power consumption increases as a square function of the applied voltage; thus, taking both energy cost and filtration performance into consideration, 8–12 V is recommended as a cost-effective operating voltage range for electrofiltration.

Nomenclature

a_c : radius of spherical collector (sand)
 d_p : particle diameter
 q : particle surface charge
 C_{in} : particle concentration at the filter inlet
 C_{out} : particle concentration at the filter outlet
 E : electric field strength
 F_E : electric force
 H : Hamaker constant
 L_H : Depth of filtration media
 L_S : spacing between the electrodes
 N_{EF} : nondimensional external electric field force
 N_G : nondimensional gravity force
 N_{LO} : nondimensional van der Waals force
 U : electric potential across the electrodes
 V_m : particle migration velocity due to electric field

V_p : particle settling velocity
 V_w : approach velocity of water
 μ : viscosity of water
 ζ : zeta potential
 ϵ_D : dielectric permittivity of water
 ρ_p : particle density
 ρ_w : water density
 ϵ : filter media porosity
 η_{single} : single collector efficiency
 pC^* : particle log-removal coefficient

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No competing financial interests exist.

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