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Performance comparison of granular media filter beds

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ABSTRACT

Particle capture, head loss development, and efficiency of in-line coagulation in contact filtration were studied using two types of laboratory filters with varying coagulants. One filter consisted of sand and anthracite, whereas the other filter contained crushed glass and anthracite. Effects of in-line coagulation on filter performance were evaluated using a charge measuring device. Performance of filters was monitored in terms of effluent quality as a function of time and development of head loss by anthracite layer and by the entire filter bed. Filterability index was used to provide a measure of effectiveness of filter media and/or operational variables such as coagulant type and dosage. This study found that the crushed glass–anthracite filter was as effective at removal of particles and metals (Fe and Al) as sand–anthracite filters.

Keywords: Filter material; Filter performance; Filterability index; Metal capture; Particle counts; Turbidity; Head loss buildup

1. Introduction

Granular filtration is one of the unit processes in drinking water treatment plants in which particulate matter originally present in water as well as particulates formed before filtration are removed. These particulates include clay and silt particles, microorganisms such as Cryptosporidium and Giardia, organic matter in particulate form, metal precipitates formed due to the use of coagulants such as aluminum or iron salts, or iron, and manganese precipitates, if pre-oxidation is used.

In conventional drinking water treatment plants, surface water passes through coagulation-flocculation, sedimentation, filtration, and disinfection. Direct filtration (or contact filtration) technique, on the other hand, eliminates flocculation and sedimentation basins where in-line flocculation occurs along the depth of the filter.

Sand, crushed anthracite coal, granular activated carbon, and garnet sand are among the materials commonly used in rapid filters [1]. Granular materials other than sand are also used for encouraging the depth removal of particulates within the bed, adsorption, and ion exchange of the species present such as arsenic, iron and manganese, bacteria, organics, and ammonium ion [2–9].

Rutledge and Gagnon [10] compared the performance of two dual media pressure filters: (i) anthracite–crushed glass and (ii) anthracite–sand. Raw water taken from a lake had been passed through coagulation–flocculation (dissolved air flotation) and disinfection steps before passing through filters. Performance of the two filters was compared in terms of effluent particle counts (as numbers) and log-removal of particles as a surrogate of intestinal parasites. The authors indicated the purpose of the study was to determine the feasibility of crushed glass, emphasizing that the crushing process had not been optimized before the research. Anthracite size, size distribution, and porosity (anthracite layer) values were not provided. Natural sand and prepared crushed glass had different effective sizes (0.33 mm for sand and 0.59 mm for crushed glass) and different uniformity coefficients, an indicator of size distribution (1.82 for sand and 1.58 for glass). In spite of grain size difference in filters, similar log-removals were obtained (1.4–1.6 log removal in glass and sand filters, respectively).

In another study, filtration experiments were undertaken on site at a domestic sewage works [11]. After testing the performance of three grades of recycled glass (coarse with a particle size of 1.5–2.5 mm, medium with a particle size of 0.5–1.45 mm, and fine with a particle size of 0.2–1 mm) to the current sand filters in treatment works, medium-grade glass was selected for further evaluation. Glass media, being a sustainable product, has been shown to treat more flow than sand and achieve similar solids removal to those obtained with sand.

A two-stage tertiary filtration system, including coarser media in the first filter to remove large sized particles and finer media in the following filter to provide a polishing stage to remove finer sized particles, was used to pre-treat high solids loading wastewater of a chemical company, in another study [12]. The facility aimed to further treat this filter effluent in a membrane system for water re-use. Recycled glass media was found to overcome high solids loading without clogging and clean easily with air scouring followed by a backwash.

Recycled glass was found to be a more effective material in removing nitrogen loads from secondary treated effluent than sand [13]. Recycled glass media was compared to sand in a stratified filter bed arrangement including several layers of gravel, crushed glass (or sand), and limestone. The polishing filter including glass medium was also capable of removing some levels of phosphorous (approximately 10% less than the filter including sand) and enteric micro-organisms (2–3 log removal). The performance of monomedium recycled glass filter in a similar arrangement (silica sand together with limestone sand) explained above was tested using a synthetically prepared wastewater for several hydraulic loading rates [14]. Recycled glass medium was found to have significant potential in removing organic materials. It was also reported that glass material can be used as a soil conditioner to improve permeability of slowly draining soils used in on-site treatment systems.

Activated glass filtration media (AFM), manufactured by Dryden Aqua, was reported as an effective filtration media for removal of harmful organic molecules, heavy metals, bacteria, and parasites from water [15,16]. AFM active filter media was manufactured to have surface catalytic properties and negative zeta potential. According to the tests conducted on the medium, it was found to be twice as good as sand in removing the smallest and most toxic particles. Specific benefits were as follows: (i) reduced chemical amounts used in treatment by more than 25%, (ii) eliminated the need for anthracite use in treatment, (iii) reduced use of activated carbon, (iv) reduced chlorinated by-products by more than 50% (since the medium had no biofilm attached on its surface and the surface was not acidic), and (v) reduced ferric and manganese concentrations in filter effluents by more than 50%.

Previous studies in our laboratories [2,3] verified that crushed glass can be a very effective substitute for silica sand, both in monomedium or dual media filters. The present work aims to bring more detailed evaluation of the performance of crushed glass filters compared to sand by focusing on the following issues:

1. Use of a dimensionless number, filterability index (FI), combining particle capture, and head loss generation in filters to compare different filter media compositions.
2. Use of “time to limiting head” which was calculated from “rate constant for head loss buildup” to compare efficiency of filters.
3. Use of “streaming current” measurements conducted on source waters tested to relate particle capturing performance of filters to water characteristics.
4. Use of metal analysis results of samples taken from different layers and head loss measurements of those layers to understand the formation, passage, and penetration of flocs.

2. Material and methods

2.1. Filtration setup

The filtration setup consisted of two identical filtration columns, each with an inner diameter of 10 cm and a height of 2.5 m. The filtration rate was 11.5 m/h for all experiments. The experimental system had ports to apply coagulants as in-line coagulation before raw water enters to the filters. Both filters had their own ports for coagulant addition (Fig. 1).

One filter consisted of silica sand and anthracite and the other consisted of crushed glass particles and anthracite. Silica sand was obtained from a water treatment plant located in Istanbul. Crushed glass was

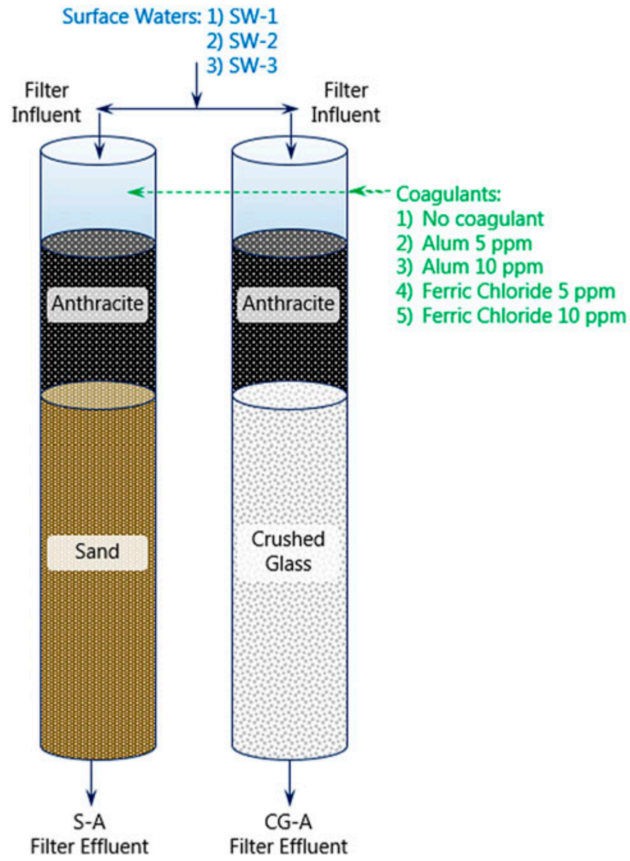


Fig. 1. Filters and filtration experiments.

prepared in our laboratories so that particle size and size distribution were equivalent to sand particles. The typical depth of rapid filters used in water treatment plants in Turkey is 1.0 m. The depth of each layer in filters used in this study was calculated as the "depth to diameter ratio", L/d_{10} [17], where L = depth of the layer and d_{10} = effective size of that layer, both measured in millimeters. Filter columns employed in this study had total depth of 104 cm: 41.5 cm of anthracite placed above 62.5 cm of either silica sand or crushed glass. Thus, the ratio L/d_{10} for the filters was $625/0.78 + 415/1.45 = 1{,}087$, a value consistent with current recommendations [18].

Two primary coagulants were compared: aluminum and ferric salts, i.e. alum, $Al_2(SO_4)_3{\cdot}18H_2O$, and ferric chloride, $FeCl_3{\cdot}6H_2O$. Concentrations of coagulant feed solutions and flow rates of dosage pumps were set to give the desired final concentrations in water being fed to filter columns. Control experiments were conducted without the addition of any coagulants before filtration. Raw waters taken from three different sources were used in the experiments (Fig. 2).

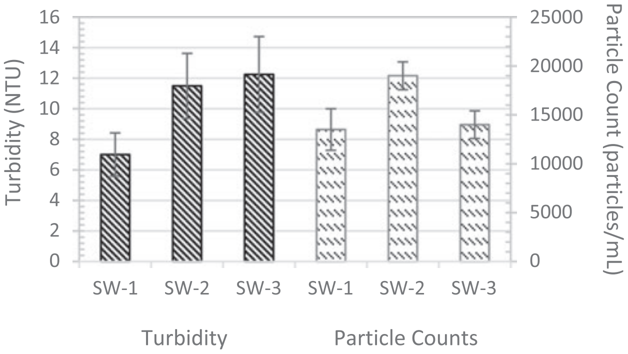


Fig. 2. Characteristics of the three surface waters (SW: Surface Water).

Size and size distribution, density, porosity, and particle shape were evaluated for all filter materials. Table 1 displays properties of the filter media used in this study.

2.2. Filter monitoring and measurements

2.2.1. Particle concentration and head loss

Continuous monitoring of turbidity and particle counts (larger than 2 μm in size) on both raw water and filter effluents was achieved using on-line instrumentation (HACH 1720D Low Range Turbidimeter and ARTI WPC-22 Particle Counter) on the experimental system. These data were recorded using a data logger and transferred to a PC in 1 min intervals. Head loss development was monitored by means of piezometer tubes installed at various heights in the filter columns.

2.2.2. Metal analysis

Fig. 3 shows the filter columns and the sampling ports where samples for metal analysis (iron and aluminum) were taken. Samples of coagulant-added water just before filtration were taken from the top sampling port which was located above the anthracite layer. The sample, coded as "anthracite layer", was taken from the port at a depth equal to 80% of the total depth of the anthracite layer lying below the top surface of anthracite. The third sample was taken from the port where the anthracite and silica sand (or crushed glass) mixes. This section of column is called the "interface". The lower layer of dual media (i.e. silica sand or crushed glass) had two more ports: (i) "sand (or crushed glass) upper layer" and (ii) "sand (or crushed glass) bottom layer" located at depths equal to 25 and 60% of total depth of sand (or crushed

Table 1
Properties of filter media

	Effective size, d_{10} (mm)	Uniformity coefficient, UC (d_{60}/d_{10})	Average particle density, ρ (kg/m^3)	Bed porosity, ε	Sphericity, ψ
Sand	0.79	1.33	2,620	0.38	0.70–0.76
Anthracite	1.45	1.39	1,590	0.58	0.54–0.63
Crushed glass	0.77	1.41	2,500	0.49	0.41–0.43

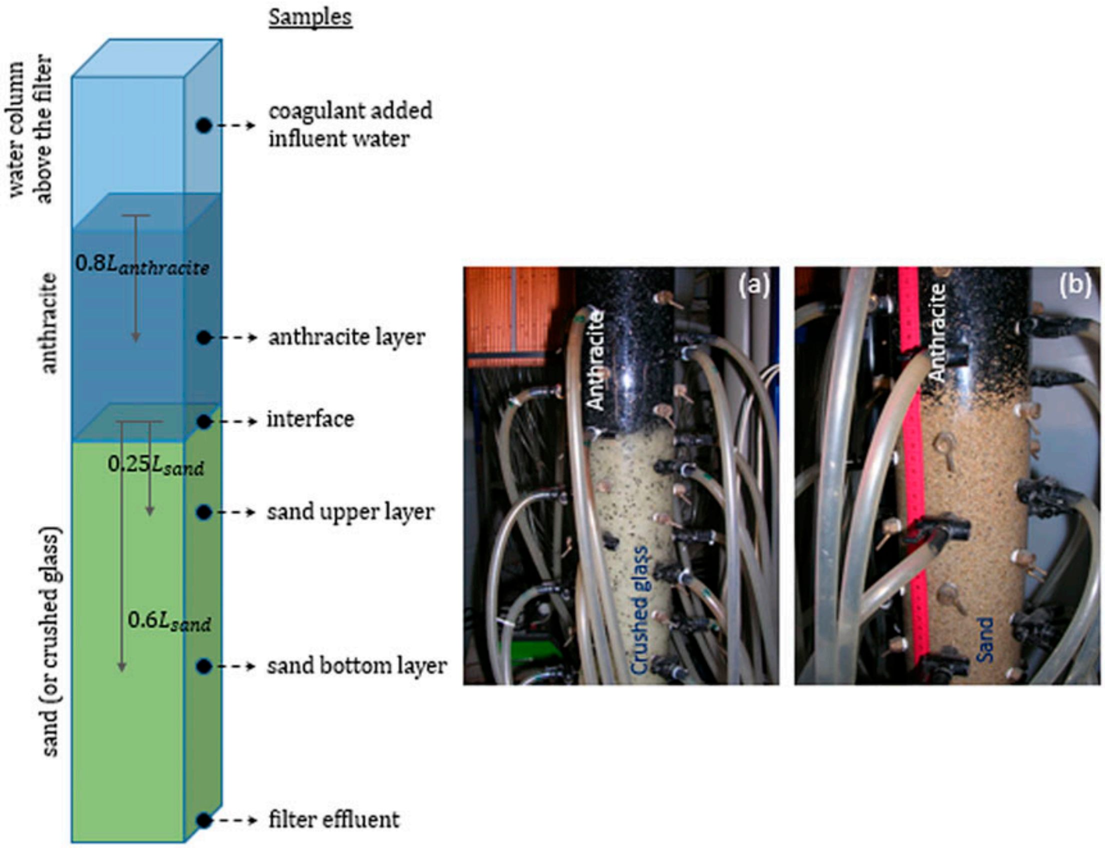


Fig. 3. Pictures of the filter columns used in the study: glass, sand and anthracite layers, head loss manometers and piezometer tubes, sampling ports at various levels and sample codes (a) Crushed glass-anthracite filter and (b) Sand–anthracite filter.

glass) layer measured at distances from the interface. Aluminum concentrations were determined by Eriochrome Cyanine R method, 3500 Al [19]. Samples that were buffered to a pH of 6.0 produce a red to pink complex with Eriochrome cyanine R dye. The complex exhibited maximum absorption at 535 nm. Iron concentrations were quantified spectrophotometrically by using the Phenanthroline Method, 3500 Fe [19]. Iron was reduced to the ferrous state with the addition of hydroxylamine and treated with 1,10-phenanthroline at pH 3.2–3.3. Phenanthroline chelates ferrous iron to form an orange-red complex that exhibits maximum absorption at 510 nm.

2.2.3. Streaming current

A charge measuring device, streaming current monitor, was used to measure net ionic and colloidal surface charge in the samples. Streaming current is related to the zeta potential by the formula (Eq. (1)):

$$i \propto ZD/N \tag{1}$$

where i = streaming current, Z = zeta potential, D = dielectric constant, and N = viscosity of the fluid.

Water samples from each water source were subjected to a simple titration procedure using two different coagulants identical to those added during filtration experiments in this study. Changes in charge with continuous addition of stock coagulant solution were recorded during the experiment. Measured numerical values (in streaming current, *SC*) do not represent the zeta potential. A zeta potential of –10 MV, for instance, is not –10 *SC* units.

Streaming current measurements can be a quick way to determine optimum coagulation, which is generally defined as the lowest dosage of chemicals that results in the desired removal of contaminants at the lowest total operating cost. Experimental results of *SC* values obtained in this study were used to compare the overall filtration performances of the waters taken from different sources. However, experimental matrix presented in this study is not wide enough to produce results for making interpretations on optimum coagulant type and its optimum dosage.

Table 2 presents the parameters monitored during experiments. Frequency of measurements and point of sampling for each parameter were also listed.

2.3. Filter effectiveness and calculations

2.3.1. Filterability index

Optimal operating conditions in a filtration system requires the following to be satisfied simultaneously: (i) the best possible average filtrate quality, (ii) the lowest possible filter head loss, and (iii) the longest possible filter operation between backwashes. In order to analyze the filtration performances of waters from different sources under different coagulation conditions through different media (either sand–anthracite or crushed glass–anthracite), a numerical index of filtration FI [20] was used. Filterability number or FI takes several factors into account such as ability of a filter to produce a clear filtrate and clogging phenomena reflected as an increase in head loss, and can be represented by the following equation (Eq. (2)):

$$FI = \frac{C}{C_0} H \frac{1}{V} \frac{1}{T} \tag{2}$$

where C = filtrate quality, C_0 = inlet water quality (turbidity or particle counts were used as the units of water quality for both C and C_0), H = head loss (m), V = filtration rate (constant and 11.5 m/h for all of the runs), and t = length of filter run (h).

2.3.2. Time to limiting head

Rate constant for head loss buildup, k_{hL}, was determined using Eq. (3) given below [21]:

$$k_{\mathrm{hL}} = \frac{(h_{\mathrm{L,t}} - h_{\mathrm{L,0}})L}{V_{\mathrm{F}}(C_0 - C_{\mathrm{E}})t} \tag{3}$$

Here, $h_{\mathrm{L,t}}$ and $h_{\mathrm{L,0}}$ represent filter head loss at time t and initial head loss, respectively. Total filter bed depth and filtration rate were constant during filtration: L = 1.04 m and V_F = 11.5 m/h.

The head loss increase rate constant (k_{hL}), was calculated using head loss values measured during any set of filtrations. Particle capture or the solids deposition in the filter (C_0–C_{E}) was expressed either using the turbidity data or the particle counts which were measured continuously. Therefore, for any set of filtrations, there are two average k_{hL} values expressed as *m/NTU* and *m/P.Count*.

Table 2
Parameters monitored during experiments

Parameter	Point(s) of measurement	Time
Turbidity	Filter influent and effluent	Continuous monitoring during filtration
Particle count	Filter influent and effluent	Continuous monitoring during filtration
Head loss	Across the media (anthracite, sand, crushed glass, interface)	Different stages of filtration (ripening, filter run)
Metals (Fe and Al)	Filter influent and effluent	Continuous monitoring during filtration
Metals (Fe and Al)	Across the media (anthracite, sand, crushed glass, interface)	At the end of filter run
Streaming current	Measured in samples of surface waters studied	

Calculated k_{hL} values were integrated to the formula below (Eq. (4)) to obtain the time of filtration runs to limiting head [21].

$$t_{\mathrm{hL}} = \frac{(H_{\mathrm{T}} - h_{\mathrm{L,0}})L}{k_{\mathrm{hL}} V_{\mathrm{F}} (C_0 - C_{\mathrm{E}})} \tag{4}$$

where t_{hL} = time to limiting head (h) and H_{T} = total available head (m).

Table 3 shows quantities calculated using the measurements taken during experiments.

3. Results and discussion

3.1. Particle-capture performance

Filter media particle-capture performance was assessed using measures of both turbidity and particle count removal efficiency. Turbidity removal efficiency by filtration without any added coagulant averaged 70%, with a maximum of 85% depending on initial turbidity. Particle count removal efficiency averaged 60%, with a maximum of 75%. Fig. 4(a) and (b) show relative effluent concentration evaluated in terms of turbidity and particle count, respectively. All experimental conditions produced similar relative effluent concentrations for both filtration media with respect to turbidity. Application of ferric chloride resulted in better effluent turbidity than alum, with the best performance occurring in the 10 mg/L condition. For particle count, differences in removal rates between the two filters were observed only in the no-coagulant condition. In the third surface water (SW-3), the sand–anthracite filter showed better performance than the crushed glass–anthracite filter (relative effluent concentration being 0.33 and 0.43 for sand–anthracite and crushed glass–anthracite filter, respectively). Similar to results obtained with turbidity, application of ferric chloride resulted in better particle count than alum, with the best performance occurring in the 10 mg/L condition.

3.2. Head loss generation

Head loss for the entire filter bed and head loss across different layers were measured during all runs. Development of head loss by the entire bed at different coagulant dosages is shown in Fig. 5. The filter including crushed glass and anthracite produced lower levels of head loss, than the sand and anthracite filter except for the filtration run with the addition of ferric chloride at a dosage of 10 mg/L (for the first surface water studied, SW-1) and alum at a dosage of 10 mg/L (for SW-3). In these particular experiments, head loss at the end of the filter runs was 5–7% higher than that occurring in the sand–anthracite filter, whereas for the rest of the tests, 12–36% lower head loss values were observed in the crushed glass–anthracite filter.

Increase in dosage for the same coagulant resulted in a decrease in the head loss difference between the two filters. Results indicated that high values of head losses which are close to the values observed in sand–anthracite filter can be seen for the filter containing glass particles at the bottom of anthracite during the experimental runs with high coagulant concentrations. No coagulant experiments on both filters, producing similar effluent qualities (see Fig. 4), resulted in 21–25% difference in head loss.

3.3. Filterability index

Difference in FI (shown in Fig. 6) between filtration through a crushed glass-anthracite filter and conventional sand–anthracite filter was calculated using the following equation (Eq. (5)):

Table 3
Calculated quantities for evaluations

Efficiency of particle capture[a]	Removal rate: $\frac{\text{Particles in influent} - \text{Particles in effluent}}{\text{Particles in influent}} \times 100$ $\left[\frac{C_0 - C}{C_0}\right] \times 100$ Relative effluent concentration: $\frac{\text{Particles in effluent}}{\text{Particles in influent}}$ $\frac{C}{C_0}$
Filterability index and difference in filterability index	
Time to limiting head	
Metal removal efficiency	

[a] Calculated using measurements of both turbidity and particle count.

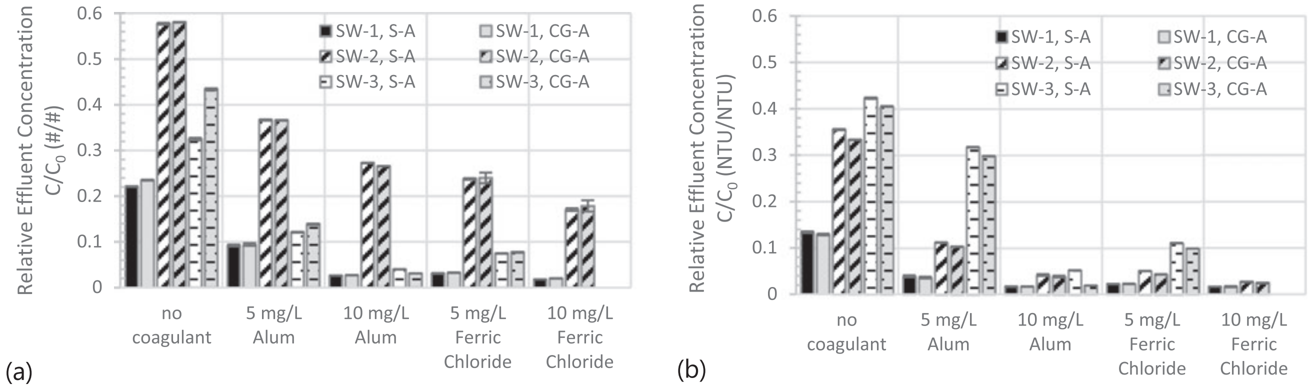


Fig. 4. Relative effluent concentration (C/C_0) as measured by turbidity (a) and particle count (b). SW: Surface Water, S-A: sand–anthracite filter, CG-A: crushed glass-anthracite filter.

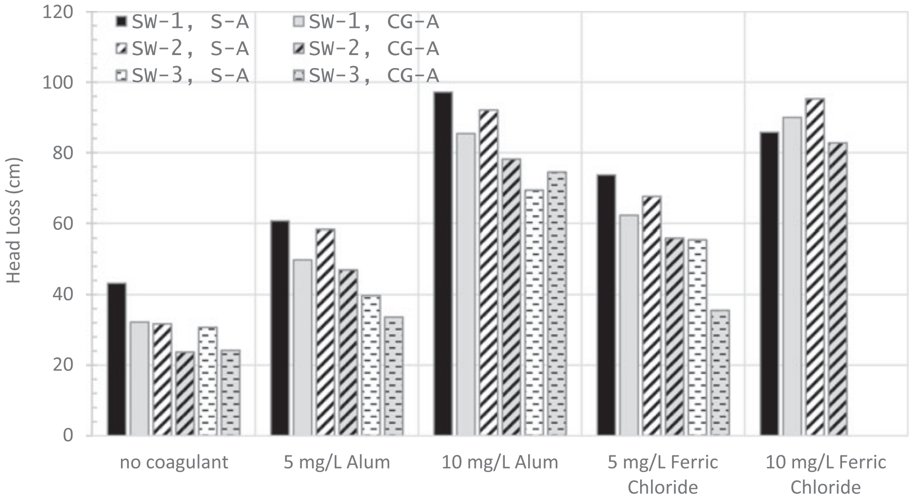


Fig. 5. Total head loss across the filter bed. RW: raw water, SA: sand-anthracite filter, CGA: crushed glass–anthracite filter.

$$\left[\left(FI_{\text{sand/anthracite}} - FI_{\text{crushed glass/anthracite}}\right)/FI_{\text{sand/anthracite}}\right] \times 100 \tag{5}$$

where *FI* is the FI calculated using the data collected during filtration experiments (Eq. 3). In the graphs (Fig. 6) on the left, relative effluent concentration (C/C_0) was put in (*FI*) calculations as turbidity; whereas in the graphs on the right, it was put in (*FI*) calculations as particle count. Both the individual *FI* values and the differences between two filters were calculated for at least three different lengths of filter runs: (i) filter ripening (t/t_f being 10–20%), (ii) filtration run ($t/t_f = 20–100\%$), and (iii) just before backwashing ($t/t_f = 100\%$). Calculated *FI* values for both of the filters and for three source waters tested are given in Fig. 7. As an example, a value on the graphs (Fig. 6) being equal to 30 means that the crushed glass-anthracite filter produced an *FI* which is 30% lower than that observed in sand-anthracite filter for the time of the experiment considered (t/t_f).

In general, difference in values of FI between filters diminished with the use of ferric chloride. Crushed glass–anthracite filter loses the advantage of producing less head loss depending on the increase in coagulant dosage. Especially towards the end of the filter life, both of the filters produced similar effluent qualities. As a result, FI values for those experiments were nearly the same.

Performance of an individual filter run is evaluated in terms of quality of the effluent produced and the length of filter run during which the filter produces clean water. These two primary performance indicators

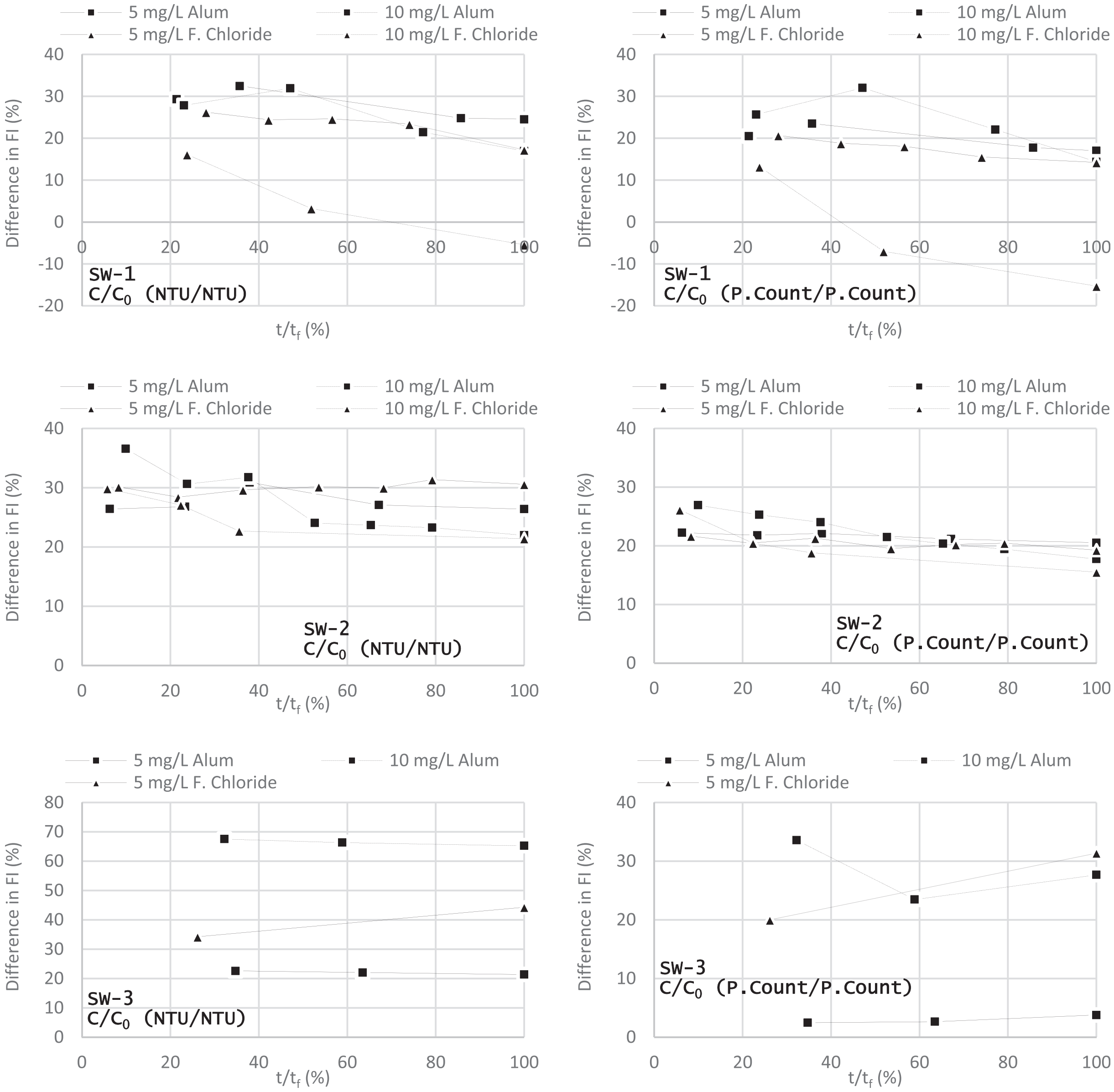


Fig. 6. Difference in FI (Surface Water: SW).

can be combined and expressed in a dynamically changing number as discussed previously. Figs. 6 and 7 show that FI calculated from filtration experiments conducted on crushed glass–anthracite filter, was 20–30% lower than FI values obtained with the sand–anthracite filter. It is also seen that one may not be able to compare directly the performance of filters or the effects of changes on operating conditions for the same filter if only particle removal efficiencies (Fig. 4) or head losses (Fig. 5) are selected as parameters of comparison. As a result, FI or the differences in values of index can be used efficiently as it covers different characteristics of the filter medium.

3.4. Time to limiting head

In design an operation of granular filters, the time required for a filter to reach limiting head loss and the time at which turbidity (or particle) breakthrough is observed are factors of equal importance. Filter beds used in water treatment are usually backwashed when the filters get clogged (and reached that limiting head loss). Clogging head, allowed for retention of particles through the filter, should not be less than 1.5 m for a filter run length of 24 h. Limiting time for filter run calculations were based on the following assumptions: (i) the filter is taken out of operation and is backwashed when limiting head reaches 2 meters, and

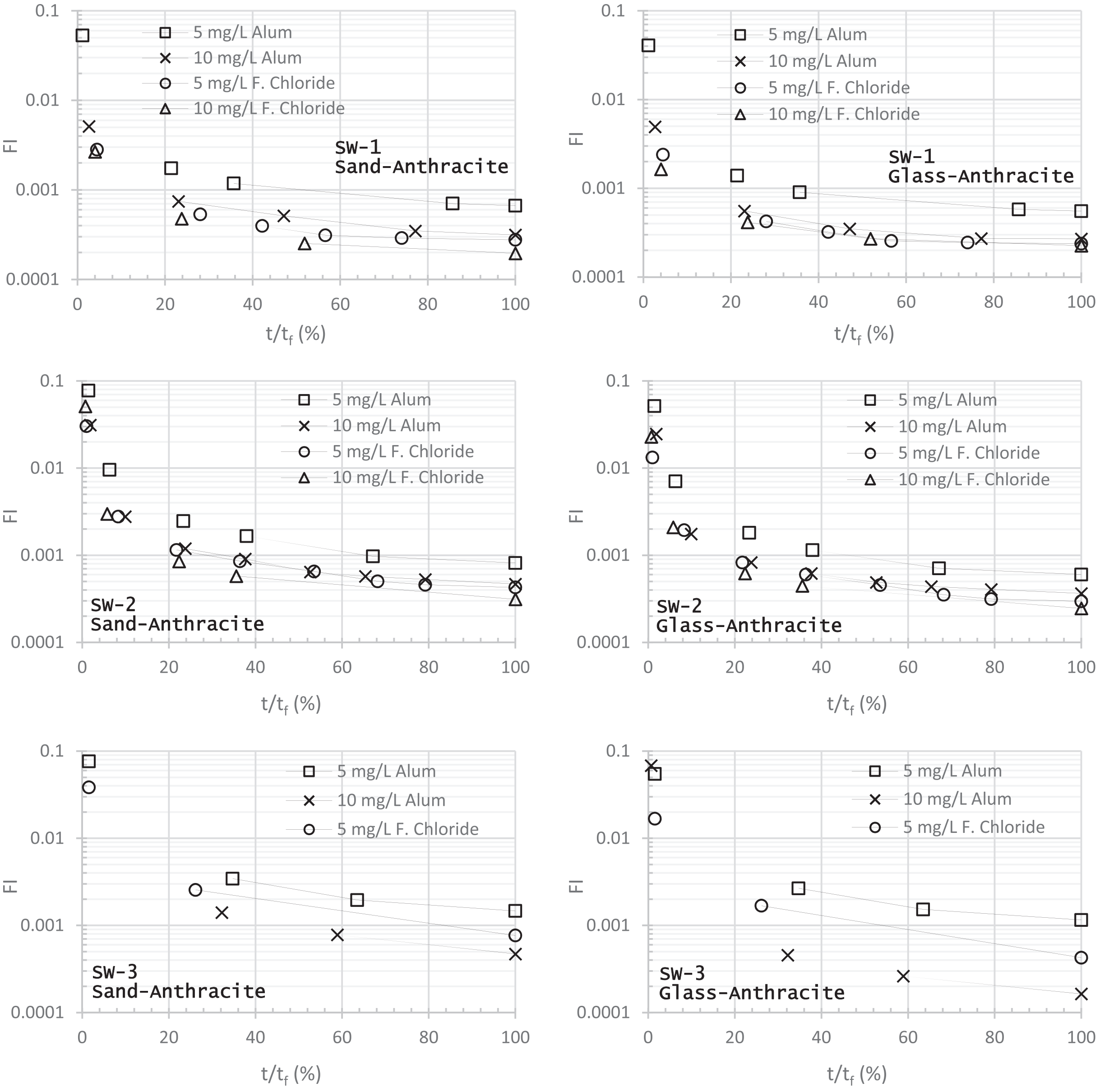


Fig. 7. FI values for sand–anthracite and crushed glass–anthracite filters obtained from the filtration experiments conducted on three different source waters (Surface water: SW).

(ii) effluent turbidity and particle counts are maintained at a steady-state value of 0.01 NTU and 100 particles/mL, respectively.

Change in duration of a filter run to reach limiting head is given in Fig. 8 (for SW-1 and SW-2). In the graphs on the left, the unit of rate constant for head loss buildup, k_{hL}, is *m/NTU*; whereas in the graphs on the right k_{hL} was calculated as *m/P.Count*.

Projected t_{hL} values for the first surface water (SW-1) showed that placing crushed glass below anthracite instead of sand improves efficiency by allowing the filter to be operated 26–34% longer periods between backwashes. It was also seen that only at high dosages of ferric chloride, both filters would be taken out of service and backwashed after producing approximately the same amount of filtered water. The crushed glass–anthracite filter will allow 15–20% longer operations than sand–anthracite filter for treatment of SW-2 in direct filtration mode.

3.5. Streaming current

Fig. 9 shows the results of streaming current measurements performed on samples of three different

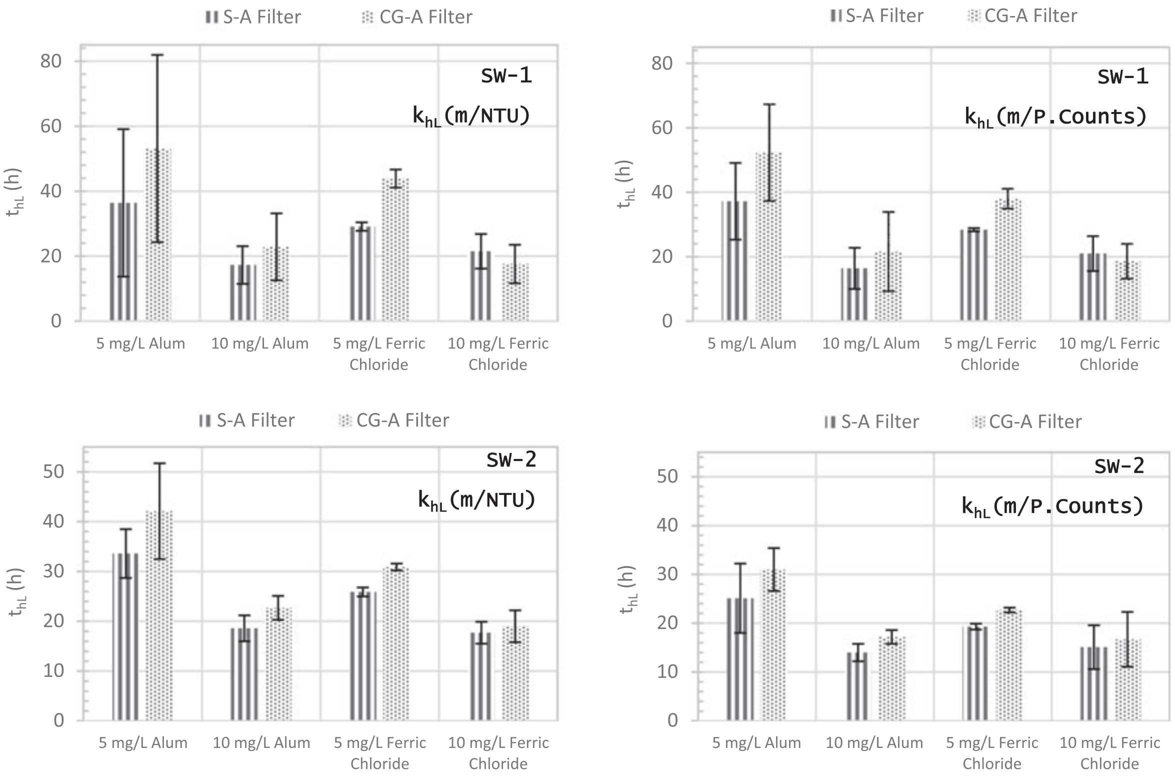


Fig. 8. Variation of t_{hL}, time to limiting head loss, with coagulants for both of the filters.

sources used in this study. Samples were coagulated with alum ($Al_2(SO_4)_3{\cdot}18H_2O$) and ferric chloride ($FeCl_3{\cdot}6H_2O$). Coagulant dose was expressed in mg/L (left-hand graph) and in millimoles/L (right-hand graph). “Al as mM metal coagulant” was 17 and 19% lower than “Fe as mM coagulant” when added in concentrations of 5 and 10 mg/L (as $Al_2(SO_4)_3{\cdot}18H_2O$ and $FeCl_3{\cdot}6H_2O$), respectively. Results of SC measurements demonstrated different trends for raw waters taken from different sources. Initial SC values show that the most negative value (SC was between –2.0 and –2.5) belongs to SW-3. SW-1 and SW-2 had SC values of –1.5 and –1.0, respectively. SC titration curve of SW-3 corresponds to the sharpest curve. Steepness of the curve indicates that surface charge on particles changes rapidly. On the other hand, change on streaming current were minimal with increasing dosages of the coagulants added into the titration

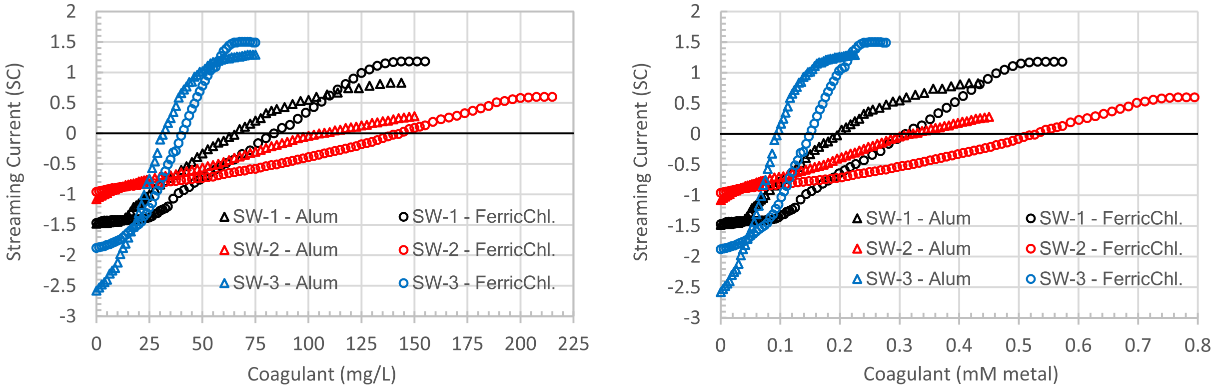


Fig. 9. Variation of streaming current with different coagulants and dosages.

medium for SW-2 (highest streaming current among the waters studied).

As previously explained, when head loss developments of three waters are compared, it was understood that generally highest head loss was observed in SW-1 and the lowest head loss was observed in SW-3. SW-1 was a source water having turbidity and particle counts that are considered in the range of direct filtration. It was also emphasized above that highest removal efficiencies (both in terms of turbidity and particle counts) were obtained with this water. Although SW-3 did not have significantly different levels of turbidity and particle counts, especially for low dosages of coagulants, it was not possible to reduce effluent turbidity and particle counts to the desired levels. Fig. 9, titration curves of coagulants, might be used to explain this result: among the three source waters tested, SW-3 was the feed water with lowest initial streaming current. It is seen that especially for the high coagulant concentrations there was a rapid increase in SC, corresponding the rapid change of charges on particles.

SW-2 source water might not be considered as source water for which the direct filtration with in-line coagulant addition is applicable, since it has relatively high levels of turbidity and particle counts. Filtrate qualities (Fig. 4) showed that it was not possible to reduce particle concentrations even with the high dosages of coagulants applied. This result can also be supported with the streaming current curves (Fig. 9). There was almost no change in streaming current with the addition of high concentrations of coagulants. Increased levels of coagulants together with flocculants/filter aids may need to be used with pH adjustment for this type of water to achieve high particle-capture efficiencies.

3.6. Metal removal efficiency and residual metals

Metal analyses were conducted in each experiment to: (i) examine the profile of Al or Fe from raw/coagulated to the treated water and identify the role of filter materials influencing profile changes, and (ii) evaluate metal removal efficiency of filters.

Fig. 10 shows filter performance comparing two parameters on the same graph. The primary vertical axis shows the efficiency of metal capture through the filter. Both of the filters receive coagulant-added water at the same dosage. These applied coagulant concentrations, mg/L Fe or mg/L Al, were measured by taking samples from the ports indicated as "coagulant added influent water" on Fig. 3. Actual metal concentrations tried to be maintained in filter influent (e.g. 5 or 10 mg/L ferric chloride or alum) were consistent. For example, during filtration of SW-1 in which 10 mg/L ferric chloride (as $FeCl_3\,6H_2O$) was dosed, water should have a total iron concentration of 2.07 mgFe/L (molar masses of iron(III) chloride hexahydrate and iron are 270.3 and 55.8 g/mol, respectively). Measured concentrations of Fe in filter influents for sand–anthracite and crushed glass–anthracite filter were 2.22 and 2.02 mg/L, respectively. As a result, deviations from the calculated concentrations for that particular experiment were 2–7%.

In this study, metal capturing efficiency of a filter was calculated as follows (Eq. (6)):

$$\text{Metal removal efficiency } (\%) = \frac{C_{\text{coagulant added filter influent water}} - C_{\text{filter effluent}}}{C_{\text{coagulant added filter influent water}}} \times 100 \qquad (6)$$

where $C_{\text{coagulant added filter influent water}}$ = metal concentration measured in raw water entering to filter in which

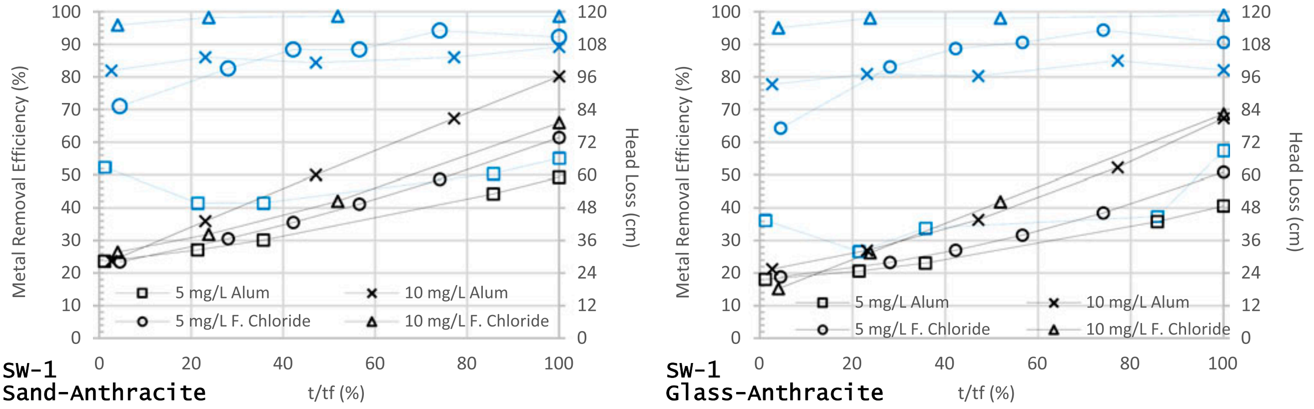


Fig. 10. Head loss development (continuous lines) and efficiency of Fe (or Al) capture (dashed lines) at the filter effluent.

coagulant was added at a desired concentration (mg/L), $C_{\text{filter effluent}}$ = metal concentration in filter effluent (mg/L).

The secondary vertical axis shows head loss measured during the entire length of each filter run. t, and t_f in dimensionless time scale show the time at which a sample from effluent was taken (and head loss through the filter was recorded), and the end of filter run, respectively. The ratio, t/t_f, was expressed as percentages, such as a $t/t_f = 20$ shows the first 20% of a filter run.

Fig. 10 shows that alum in 5 mg/L concentration did not reduce metal concentrations at the effluent. Higher metal removal efficiencies were observed as the concentration of coagulant increases. Ferric chloride, at a dosage of 10 mg/L, was highly effective for reducing metal concentrations by forming flocs through the pores of filter bed. Metal capturing efficiency was 98% for both of the filters. Head loss increase rate in crushed glass–anthracite filter was almost less than that observed in sand–anthracite filter.

Retention time (empty bed contact time) for filter depths used in this study was about 6–7 min. Therefore, results of samples taken during the first 20-min duration of filtration would not represent the effects of coagulant addition and residual metals present in effluent water, since water reaching to the bottom of filter within this period was actually the backwash water remaining in filter column.

Metal removal efficiencies have been calculated for different periods of filter runs and shown in Table 4.

Samples from various levels of filters (as shown in Fig. 3) were withdrawn at the end of each filter run for Fe or Al analysis. Head losses at the same levels were also recorded before the run was terminated.

Figs. 11 and 12 show head loss development and metal removals at various depths of filter beds for SW-1 and SW-2, respectively. Total iron and aluminum concentrations were measured in samples. Average iron concentrations in SW-1 and SW-2 were 320–430 μgFe/L, respectively. Surface waters examined in this study contained negligible levels of aluminum (10–20 μgAl/L). As expected, ferric chloride and alum additions increased total Fe and Al concentrations, respectively. Metal concentrations measured in samples drawn from different layers as well as in filter effluent correspond to small sized flocs and/or soluble metal which did not form flocs large enough to be captured by filter media.

In experiments conducted on SW-1, metal removals in sand–anthracite filter were slightly higher than those obtained in crushed glass–anthracite filter. This difference diminishes with increasing coagulant doses. It was also seen that ferric chloride was more capable of coagulating and hence removing particulates. It was also observed that head losses generated along the depth of the anthracite layer in the crushed glass–anthracite filter were higher than those that occurred in the anthracite layer of the sand–anthracite filter. When higher doses of coagulant were used (10 mg/L ferric chloride), head loss development for all layers, including the crushed glass at the bottom, became higher than the sand–anthracite filter.

It is actually expected that two filters, under the same operating conditions, should produce same head loss if they contain filter media of same particle size and size distribution. It might also be expected for anthracite layers of the two filters in consideration here, since the material (anthracite) was obtained and prepared from the same source. Although the granular medium at the bottom of the anthracite has same size and size distribution and is very similar in particle densities (sand and crushed glass), they have differences in shape. As seen from Table 1, crushed glass material has lower sphericities, a parameter indicating shape, causing it to interfere and mix into the anthracite layer. This situation was also visually observed when the filter beds were prepared before the experiments (see the picture in Fig. 3).

Table 4
Metal removal efficiencies

		Alum		Ferric chloride	
		5 mg/L	10 mg/L	5 mg/L	10 mg/L
Raw water 1	Sand–anthracite	40–55	85–90	80–92	>98
	Crushed glass–anthracite	26–55	80–92	77–90	>98–99
Raw water 2	Sand–anthracite	45–45	75–80	90–94	95–98
	Crushed glass–anthracite	43–43	65–80	85–95	93–98
Raw water 3	Sand–anthracite	50–64	75–92	38–50	
	Crushed glass–anthracite	60–60	85–94	50–50	

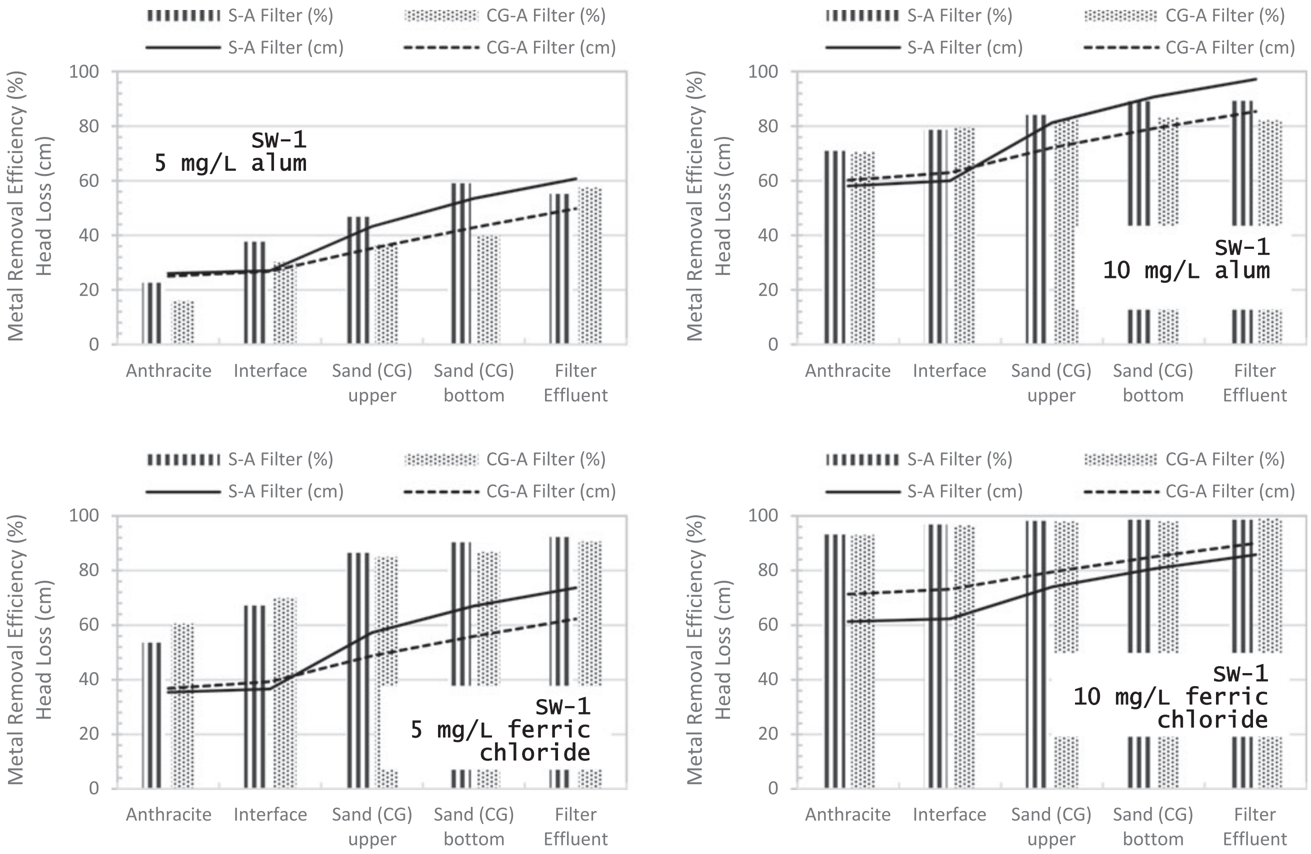


Fig. 11. Head loss development and the metal removal efficiencies at various levels of the filter (SW-1).

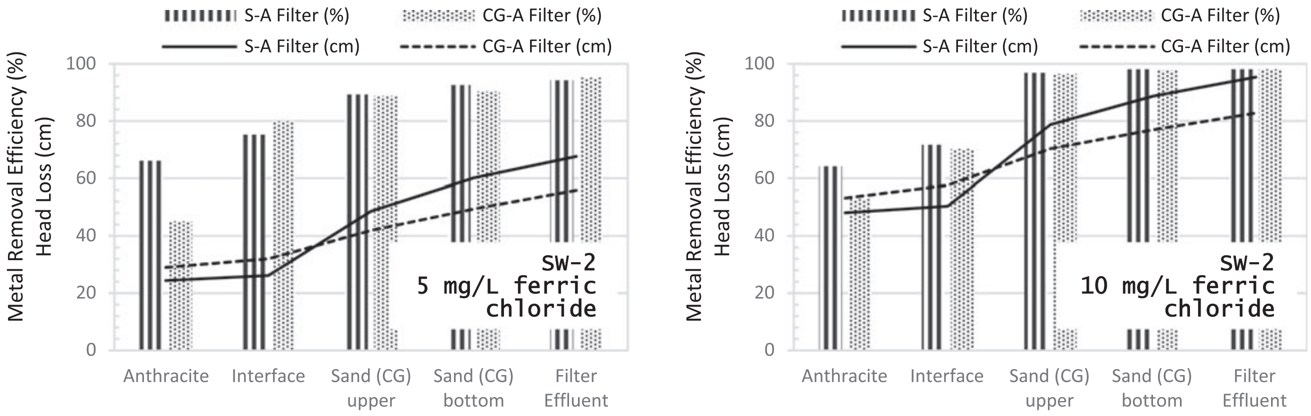
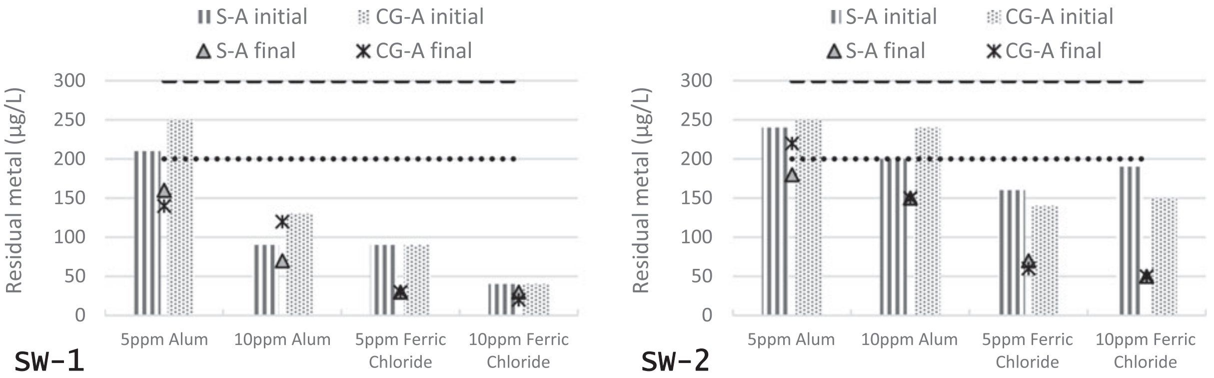


Fig. 12. Head loss development and the metal removal efficiencies at various levels of the filter (SW-2).

Metal removal rates were less than 70% for anthracite and interface layers of the filters during experiments with SW-2 even with high doses of coagulants applied (Fig. 12). These layers (anthracite and interface) produced higher head losses in the crushed glass–anthracite filter as observed in the experiments with SW-1. As discussed previously, SW-2 source water might require classical coagulation–flocculation–sedimentation–filtration sequences for better chemical coagulation and high levels of particle and turbidity removals.

Evaluation of the results of analysis was made with Turkish Standards (TS266), Water Intended for Human Consumption Regulation (WHCR), World



SW-1: alum coagulated experiments pH: 7.7-7.8, ferric chloride coagulated experiments pH: 7.4-7.6

SW-2: alum coagulated experiments pH: 7.7-7.9, ferric chloride coagulated experiments pH: 7.7-7.8

Water temperature: 20.5-21.5 °C

Fig. 13. Residual metal concentrations in filter effluents.

Health Organization (WHO), Environmental Protection Agency (USEPA) and European Union (EU) drinking water directive. Recommended guideline value for conventional water treatment plants for Al and Fe in Turkey is 200 µg/L. WHO, EPA, and EU have specified aluminum levels to be 200 µgAl/L in drinking water. Total iron standard is set as 300 µgFe/L by WHO and EPA. It is 200 µgFe/L according to EU.

Residual metal concentrations in filter effluents (both for the ripening stage and at the end of filtration) and filtered water pH values are given in Fig. 13. Alum or ferric chloride addition (5–10 mg/L) efficiently coagulated SW-1 (graph on the left) and the filters removed those alum/ferric chloride flocs. Total Al (or Fe) in filtered water was less than 175 µg/L. Residual aluminum concentrations in filter effluent during the ripening stage of filters were at (for alum addition at 5 mg/L) or below 250 µg/L. Concentrations of aluminum in effluent water were around the level of the drinking water standard (200 µgAl/L) in SW-2 (graph on the right). Total Fe (present naturally and added due to the use of ferric chloride as a coagulant) was efficiently removed by filtration.

4. Conclusions

The aim of this study was to evaluate the performance of granular filters using different performance indicators, with emphasis on the use of a different material, and crushed glass. Sand and crushed glass material were compared in dual media filters containing anthracite at top. Three different source waters were used. Filters were operated at contact filtration mode in which coagulants were added in-line before filtration. Two different coagulants (alum and ferric chloride) were used at two different dosages. No coagulants were added before filtration in control experiments.

Major conclusions are as follows:

(1) The crushed glass–anthracite filter was effective at filtering surface water in the contact filtration mode, producing less head loss than sand–anthracite filter except the runs during which high dosages of coagulants were applied.
(2) The crushed glass–anthracite filter produced 20–30% less *FI* than sand–anthracite filter.
(3) Placing crushed glass below anthracite instead of sand improved efficiency by allowing the filter to be operated 15–35% longer periods between backwashes.
(4) Although metal removal efficiencies were low during the initial stages of filtration in the crushed glass–anthracite filter, the final values were close to those obtained in the sand–anthracite filter, indicating that total metal concentrations were efficiently removed by filtration decreasing residuals below regulated values.
(5) Although granular medium at the bottom of anthracite (sand or crushed glass) has the same size, size distribution, and similar particle densities, they have differences in shape. Crushed glass particles have lower sphericities (shape indicating parameter), causing it to interfere and mix into anthracite layer. The volume of voids or porosity decreases in anthracite layer

of crushed glass–anthracite filter. This was confirmed with 15–20% increased head losses in anthracite layer of crushed glass–anthracite filter compared to anthracite layer of sand–anthracite filter. Despite this local increase in head loss (anthracite layer), crushed glass–anthracite filter would be operated in longer filter runs with no deterioration in effluent water quality.

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