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5   **Performance and surface clogging in intermittently loaded and slow sand filters**
6   **containing novel media**

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11 **Abstract**

12 Slow sand filers are commonly used in water purification processes. However, with the

13 emergence of new contaminants and concern over removing precursors to disinfection by-

14 products, as well as traditional contaminants, there has recently been a focus on technology

15 improvements to result in more effective and targeted filtration systems. The use of new

16 media has attracted attention in terms of contaminant removal, but there have been limited

17 investigations on the key issue of clogging. The filters constructed for this study contained

18 stratified layers comprising combinations of Bayer residue, zeolite, fly ash, granular activated

19 carbon, or sand, dosed with a variety of contaminants (total organic carbon (TOC),

20 aluminium (Al), ammonium (NH_4^+-N), nitrate (NO_3^--N) and turbidity). Their performance

21 and clogging mechanisms were compared to sand filters, which were also operated under two

22 different loading regimes (continuous and intermittently loaded). The study showed that the

23 novel filter configurations achieved up to 97% Al removal, 71% TOC removal, and 88%

24 NH_4^+-N removal in the best-performing configuration, although they were not as effective as

25 sand in terms of permeability. Deconstruction of the filters revealed that the main clogging

26 mechanism was organic matter build-up at the uppermost layer of the filters. The clogging

27 layer formed more quickly on the surface of the novel media when compared to the sand

28 filters, but extended further into the sand filters, the extent dependent on the loading regime.

29 The study shows the potential for an alternative filtration configuration, harnessing the

30 adsorption potential of industrial waste products and natural media.

31 *Keywords:* Clogging mechanism, water purification, fly ash, Bayer residue.

32 **1. Introduction**

33 Slow sand filters are commonly used in water treatment to remove contaminants by physical,

34 chemical and biological mechanisms (EPA 1995), but they may not be effective in the

35 removal of specific contaminants, 'emerging contaminants' (EC), or precursors to

36 disinfection by-products (DBP), such as dissolved organic carbon (DOC) (Gang et al. 2003;

37 Chuang et al. 2011; EPA 2015). Consequently, there has been much research into the use of

38 alternative media, particularly industrial by-products and waste products, or coated media, for

39 use in filters for the treatment of targeted contaminants (Bhatnagar & Sillanpää 2010; Fu &

40 Wang 2011; Rahman et al. 2013). Although these media have been examined extensively at

41 bench-scale level, their efficacy has been infrequently examined in laboratory, pilot or large-

42 scale filters (Bailey et al. 1999; Bhatnagar et al. 2011). This is a major knowledge gap, as a

43 layered configuration in a water filter may allow the utilisation of adsorption properties from

44 each of the media, and may have the ability to control any potential metal leaching.

45 Moreover, the use of waste materials in a filtration unit is potentially an effective and

46 sustainable means of water treatment. While the viability of these media for use in filters

47 depends on their efficacy in the removal of contaminants, it equally depends on their capacity

48 to treat water without clogging.

49 Clogging mechanisms in sand filters for the treatment of wastewater (Hatt & Fletcher 2008;

50 Leverenz et al. 2009) and vertical flow constructed wetlands, which operate in a similar way

51 to filters (Turon et al. 2009; Pedescoll et al. 2009; Hua et al. 2010; Knowles et al. 2011), have

52 been well researched. However, clogging mechanisms in filters for drinking water treatment

53 have not been examined to the same extent. Biological clogging has been the main focus of

54 the research to date (Thullner et al. 2002; Kildsgaard & Engesgaard 2002; Mauclaire et al.

55 2004); however, clogging may also occur by chemical and physical mechanisms (Mauclaire

56 et al. 2004; Le Coustumer et al. 2012). This may be particularly relevant for water filters.

57 Chemical clogging can affect the shapes and stabilities of the pores in the media, in turn,

58 affecting the flow paths (Baveye et al. 1998). Adsorption of substances and metal

59 precipitation from the influent water may also contribute to clogging within filters

60 (Noubactep et al. 2010). Physical clogging may result from compaction due to loads on the

61 surface of the filter and migration of the fine media into the filter. Therefore, the use of a

62 particularly fine media at the surface may result in a filter cake forming at the media-water

63 interface, contributing to physical clogging (Baveye et al. 1998). Where organic carbon is a

64 component of the influent water, clogging is expected (McKinley & Siegrist 2011), which

65 may be due to the extracellular polymer substances (EPS) sometimes present in humic acid.

66 These may form a gel-like, hydrophilic structure as the humic acid accumulates (Tanner et al.

67 1998), increasing the retardation of flow within the filter.

68 Filter head loss is the most common method of determining clogging at operational facilities

69 (EPA 1995). Clogging of filter media may be investigated in a number of ways. One of the

70 most common methods is to measure the field saturated hydraulic conductivity (K_{fs}) (Rodgers

71 et al. 2004; Pedescoll et al. 2009; Le Coustumer et al. 2012). As the filter clogs over time, K_{fs}

72 decreases (Knowles et al. 2011). This can be measured using either a falling head test (ASTM

73 2007) or constant head test (British Standard Institution 1990b), depending on the

74 permeability of the media under consideration. Other common methods of analysis include

75 loss-on-ignition (LOI), chemical analysis of the media at different depths throughout the

76 filter, scanning electron microscopy (SEM) of the biofilm layer, and X-Ray diffraction

77   (XRD) analysis (Rodgers et al. 2004; Pedescoll et al. 2009; Knowles et al. 2010; Nivala et al.

78   2012).

79   Clogging becomes evident in filters as surface ponding occurs (for intermittent filters) and

80   the outflow flow rate decreases (Knowles et al. 2011). It is important to ascertain how deep

81   the clogging layer is within the filter, as the filter can be regenerated and the hydraulic

82   conductivity restored. This may be accomplished by replacing the clogging layer of the filter

83   with fresh media (Mauclaire et al. 2004). Current guidelines advise the removal and re-

84   sanding of slow sand filters once a predetermined design head loss has been reached (EPA

85   1995). However, re-sanding beyond the clogging layer leads to excessive and unnecessary

86   cost.

87   In the current paper, the water contaminants studied were DOC, aluminium (Al), nitrate-

88   nitrogen (NO_3^--N), ammonium-nitrogen (NH_4^+-N) and turbidity. Dissolved organic carbon

89   present in source water causes formation of DBPs such as trihalomethanes and haloacetic

90   acids following chlorination, and oral ingestion of DBPs can lead to cancer (Wang et al.

91   2007). Removal of DOC at filtration stage would reduce the potential of DBP formation,

92   which has been the main source of non-compliance in Irish water treatment plants (WTPs) in

93   recent years (Water_Team 2012). Aluminium is added to water during the treatment

94   processes, but only represents a problem when it is present in the source water due to

95   geological leaching (Calderon 2000). The maximum allowable concentration (MAC) for Al

96   in water is 200 µg L^{-1} (SI No 278 of 2007), and epidemiological studies have previously

97   established a link between excessive Al and Alzheimer's disease (Flaten 2001; Bondy 2010;

98   Exley & Vickers 2014), as well as having other human toxicity effects (Nayak 2002). The

99   MAC for NO_3^--N in drinking water is 10 mg L^{-1} (SI No 278 of 2007), as consumption of high

100 levels of NO_3^--N may cause methemoglobinemia, and can have significant environmental

101 impacts on agricultural and aquatic life (Bhatnagar and Sillanpää 2011). Ammonium has a

102 MAC of 0.3 mg L^{-1} in drinking water, and is generally present in source water due to

103 anthropogenic activities. A residual presence of NH_4^+-N at the disinfection stage can cause

104 extra chlorine consumption, and may have a negative effect on the disinfection process (Feng

105 et al. 2012), and can suppress pH and dissolved oxygen in the supply system (Wilczak et al.

106 1996). Legislation states that turbidity must be acceptable to consumers and have no

107 abnormal change (SI No 278 of 2007), except in the case of surface water treatment where

108 1.0 NTU should be achieved. However, turbidity can also be an indication of likely DBP

109 formation, and excessive turbidity can also inhibit disinfection (EPA 2011).

110 This paper focuses on the use of two industrial by-products, Bayer residue ('red mud') and

111 fly ash, and a natural medium, zeolite, combined with granular activated carbon (GAC) and

112 sand, in layered filter configurations. Bayer residue is a waste product of the Al production

113 industry, and is often stored in bauxite residue storage areas close to the production site (EAA

114 2013). Fly ash, a by-product of incineration, is most commonly used in the manufacture of

115 concrete (Mehta 2002). Natural zeolites are known adsorbents of contaminants in water and

116 wastewater treatment (Wang and Peng 2010). Each of these has been used previously for

117 adsorption of contaminants, although not in stratified filter configurations operated at

118 laboratory-scale, and their maximum adsorption capacities are summarised in Table 1.

119 While the performance of slow and intermittently loaded filters is important in the selection

120 of suitable filter media, the hydraulic function and permeability are also crucial parameters

121 that need to be considered. Therefore, the aim of this study was to (1) determine the

122 mechanisms of clogging of the filters and (2) assess the performance of intermittently and

123 constantly loaded filters, each containing the novel filter media and operated for a period of

124 90 days, in the removal of contaminants (DOC, Al, NH_4^+-N, NO_3^--N and turbidity) of water.

125 **2. Materials and methods**

126 2.1 Filter construction

127   Three filter configurations, each replicated at n=3, were examined with constantly loaded and

128   intermittently loaded operational regimes, giving a total of 18 filters (Fig. 1). The first

129   configuration ('Config 1') was a three-layer stratified filter (each layer had a depth of 0.33 m)

130   containing (downwards from the filter surface) Bayer residue, zeolite and coarse sand. The

131   second configuration ('Config 2') was a four-layer filter, with equal layers of 0.25 m media,

132   containing (downwards from the filter surface) flyash (from coal combustion), GAC, zeolite

133   and coarse sand. The effective particle sizes (d_{10}) of each medium are given in Table 2. The

134   third configuration ('Control') was a 1-m deep single layer sand filter with an d_{10} of 0.18 mm

135   and uniformity coefficient of 2.19 (EPA 1995). The filter configurations were chosen based

136   on adsorption results obtained by Grace et al. (2015), where NH_4^+-N removal was focused at

137   the surface of the filter, and Al, TOC, and NO_3^--N further down through the media. Locally

138   available media were chosen, where possible. The configurations took cognisance of the K_{fs}

139   of each media, which was measured prior to the experiment. Each filter had a free-board

140   depth of 0.5 m above the filter surface. Physical and chemical characteristics of the media are

141   detailed in Table 2.

142 2.2 Filter operation

143   The intermittently loaded filters were subjected to an initial loading rate (day 1 of

144   experiment) of 0.1 m hr^{-1} following the Irish EPA guidelines for slow sand filters

145   (intermittent filters are not currently used for large-scale drinking water treatment) (EPA

146   1995). The intermittent filters were dosed for 10 min every 2 hr using a peristaltic pump

147   (7528-10, Masterflex L/S Variable-Speed Drive). A head of water of 0.5 m was maintained

148   above the constantly loaded filters.

149   The water feed, the same for both loading regimes, comprised tap water dosed with NH_4^+-N

150   (synthesised using laboratory-grade NH_4Cl to a concentration of 5 mg L^{-1}), NO_3^--N

151   (synthesised using laboratory-grade KNO_3 to a concentration of 20 mg L^{-1}), Al (synthesised

152   using laboratory-grade aluminium powder to a concentration of 2 mg L^{-1}), and DOC

153   (synthesised using laboratory-grade humic acid to a concentration of 10 mg L^{-1}). These

154   concentrations were based on maximum exceedances reported by the Irish EPA

155   (Water_Team 2012). The humic acid was prepared using a method adapted from Abdul et al.

156   (1990), to remove as much of the non-water soluble fraction as possible. This involved

157   mixing the humic acid with deionised water for 25 min and then centrifuging at 1000 RCF for

158   30 min before filtering through 0.45 µm filter paper. Using humic acid as the DOC source

159   resulted in influent suspended solids of approximately 200 mg L^{-1}, greater than would be

160   present in a standard groundwater or surface water source, allowing an investigation of

161   clogging occurrence at a high mass loading rate. Irish water quality reports indicate that in

162   surface water, suspended solids concentrations do not often exceed 35 mg L^{-1} (EPA

163   2008).The two loading regimes were operated in parallel for 90 days.

164   **2.3 Statistical Analysis**

165   Statistical analysis was carried out using SPSS 22 software (IBM, 2014). The data were

166   checked for normal distribution using the Shapiro-Wilk test. The Kruskal-Wallis test for non-

167   parametric data was used, where the null hypothesis is that the distribution, and the medians

168   of the test fields are the same across groups.

169   **2.4 Hydraulic conductivity**

170   The variety of media used in this study meant that both the constant head test (appropriate for

171   media with a K_{fs} of between 10^{-2} and 10^{-5} m s^{-1} (British Standard Institution 1990b) and the

172   falling head test (appropriate for media with a $K_{fs} \leq 10^{-5}$ m s^{-1}, ASTM 2007) were required to

173   analyse the K_{fs} of the filters. Samples of the virgin media were tested initially for K_{fs}. At the

174   end of the 90 day trial, samples of media were collected at incremental depths to analyse K_{fs}

175   variation with depth. The sample collection in each filter was terminated when the K_{fs}

176   measured at a given depth returned to that of the virgin sample.

177   For the constant head test, two undisturbed media cores, 0.5 m in diameter, were taken at

178   0.02 m incremental depths from the filter surface. Water was kept at a constant head over the

179   top of the sample, using an overflow valve to maintain the head of water. The sample was

180   retained in an open-ended vessel to allow the water to flow freely through the sample. The

181   hydraulic gradient was defined as (Rodgers et al. 2004):

$$\frac{dH}{dZ} = 1 + \frac{z}{l}$$

182   where dH/dZ is the hydraulic gradient, z is the head of water, and l is the height of the

183   sample. The hydraulic gradient was calculated using Darcy's law (Craig 2004):

$$\frac{Q}{A} = K_{fs}\left(\frac{dH}{dZ}\right)$$

184   where Q is the volume of water flowing per unit time (m^3 s^{-1}), A is the cross-sectional area

185   (m^2), K_{fs} is the field-saturated hydraulic conductivity (m s^{-1}), and dH/dZ is the hydraulic

186   gradient (m m^{-1}).

187   The falling head test was carried out at incremental depths of 0.05 m from the surface (the

188   difference in incremental depths between constant and falling head tests was due to

189   limitations with the laboratory apparatus). The samples were saturated for 24 hr before being

190   placed in an overflow vessel, which was clamped in an apparatus that allowed a free flow

191   through the base. The reservoir water was de-aired and the manometer was filled. The test

192   was then carried out by allowing the de-aired water in the manometer to flow through the

193   sample. The time taken for the meniscus to fall between two measurements on the manometer

194   was recorded, and the K_{fs} was calculated using (ASTM 2007):

$$K_{fs} = 2.3 \frac{A_2}{A_1} \frac{L}{T} log \left(\frac{h_1}{h_2} \right)$$

195   where K_{fs} is the field-saturated hydraulic conductivity (m s^{-1}), A_2 is the cross-sectional area of

196   the manometer (m^2), A_1 is the cross sectional area of the sample (m^2), L is the height of the

197   sample (m), T is the time taken for the water level to fall (s), h_1 is the height of the water in

198   manometer at t = 0 (m), and h_2 is the height of the water in the manometer at t = T (m).

199   **2.5 Chemical Composition**

200   Media samples were taken at incremental depths of 0.02 m from the surface. Organic matter

201   analysis was carried out by LOI, as described in Schulte and Hopkins (1996). Metal analysis

202   was carried out using the Mehlich soil extractant method (Mehlich 1984). Total exchange

203   capacity was carried out using the method described in Ross (1995). Ammonium was

204   analysed using the 1 N KCl method (Dahnke 1990) and NO_3^--N was analysed using the

205   saturated paste extract method (Gavlak et al. 2003). The media characteristics were

206   determined using BS 1377:2 (British Standard Institution 1990a).

207   **3.   Results and Discussion**

208   **3.1 Filter performance**

209   The performance of each filter configuration is summarised in Table 3. Turbidity removal

210   was effective among all configurations, although it did not reduce to below the MAC of 1.0

211   NTU (SI No 278 of 2007). However, the initial turbidity of the synthetic water was much

212   higher than would be expected in raw water prior to treatment. Dissolved organic carbon

213   removal was most effective in Config 2, where higher percentage removals were exhibited in

214   both loading regimes, in comparison to the other configurations. These removal rates, of 63-

215   71%, were also higher than those found in conventional WTPs, which can be expected to

216   remove 10-50% of DOC (Kim & Kang 2008). Effective DOC removal is important, given

217   that DOC is a pre-cursor to many DBP, and has been identified as a major problem in Irish

218   drinking water treatment plants (EPA 2015).

219   Aluminium removal was greater than 94% in all filters, and there was no significant

220   difference in the performance of Config 1 and 2 (p=0.114). In all cases the filters were able to

221   reduce the concentration of Al to below the MAC of 200 μg Al L^{-1} (SI No 278 of 2007).

222   Aluminium removal could attributed to ion exchange, a common practice for Al removal

223   from aqueous solutions (Pesavento et al. 1998). This is particularly likely in the case of Bayer

224   residue which exhibits a high total exchange capacity (Table 1). Aluminium adsorption may

225   also have been enhanced by molecular interaction with humic acid present in the water feed

226   (Elfarissi et al. 1998; Tombácz et al. 2000).

227   All filters were capable of NH_4^+-N removal, with significantly higher removal of NH_4^+-N

228   occurring in Config 1 and 2, versus the Control, when operated under intermittent loading

229   (p=0.01). The trend of NO_3^--N production, coupled with high NH_4^+-N present in the influent

230   supply and a carbon source, suggested that nitrification was occurring within the filters.

231   **3.2 Visual observation**

232   After 90 days of operation, clogging was observed across all filter configurations. The

233   clogging layer was most likely caused by a build-up of organic matter, due to the loading of

234   humic acid with high suspended solids, and biomass accumulation (Mauclaire et al. 2004).

235   Despite the filters not being seeded with biomass, the occurrence of nitrification indicates that

236   a biomass layer was present. This was evident in both Config 1 and 2, where a gel-like layer

237   was visible on the surface of each column operated in either loading regime (Fig. S1). The

238   Control was slightly different, in that a colour change was evident at the uppermost layer, but

239　the larger particle size meant the gel-like layer was less obvious. This was verified by the

240　organic matter analysis (Section 3.4) on all three configurations, where the organic matter

241　content decreased with depth from the surface in the filter.

242　**3.3 Hydraulic Conductivity**

243　The K_{fs} results are displayed in Fig. 2. The results are normalised against the K_{fs} of the virgin

244　media, where K is the hydraulic conductivity of the clogged media and K_v is the K_{fs} of the

245　virgin media. There was some variation in the K_{fs} of the Control filters, which could be due

246　to the arrangement of the coarse and fine sand during column construction, followed by

247　migration of the fines. There is also evidence of scatter in Config 1. This may be due to the

248　nature of the Bayer residue (which has a mud-like texture) and voids within the layer of

249　media that were evident upon deconstruction. However, a general trend of increasing

250　conductivity with depth below the surface was observed.

251　The extent of clogging in the Control filters differed depending on the operational regime

252　(Fig. 2). The constantly loaded filters showed that the clogging layer extended further, to

253　approximately 0.3 m below the surface of the filter, whereas the K_{fs} of the intermittent filters

254　returned to that of the virgin media between 0.15 and 0.2 m below the surface of the media.

255　This was evident by both a colour change of the sand and the hydraulic conductivity results.

256　This was greater than the clogging depth reported by Zhao et al. (2009), who observed a

257　decreasing K_{fs} near the filter surface of an organically-loaded, continuously fed, laboratory-

258　scale, vertical flow constructed wetland, operated for 2 months, and concluded clogging

259　occurred in the top 0.15 m layer. The clogging mechanisms of Config 1 and Config 2, i.e. the

260　gel-like layer on the surface of the media, did not differ much between the two loading

261　regimes. This is most likely due to the nature of the media; both fly ash and Bayer residue

262   having a small particle size (Table 2) meant that the clogging layer formed more quickly on

263   the surface of the media (Hand et al. 2008; Thullner 2010).

264   Nieć & Spychała (2014) found that for medium to high permeability of sand and gravels, the

265   constant head method is reliable, but that a method taking into account capillary rise

266   measurements is more acceptable for fine sands. Methods that take into account filter

267   resistance may also be appropriate in estimating the true permeability, which could be used in

268   filter design (Li et al. 2005). These could be investigated in future filter designs.

269   **3.4 Organic Matter Composition**

270   Fig. 3 displays the organic matter composition (percentage organic matter per dry weight) of

271   samples taken throughout the clogging layer. In each configuration and each loading rate,

272   more organic matter was measured on the surface of the filter than at further depths. This is

273   consistent with the hydraulic conductivity and observation results, which report a layer of

274   organic matter at the surface, leading to the clogging of the filters. The Bayer residue in

275   Config 1 had higher initial concentrations of organic matter present, which appeared to leach

276   down as water passed through the filter, shown by the percentage decreasing below that of

277   the virgin media with depth from the surface.

278   **3.5 Performance Outlook**

279   The performance results indicate that Bayer residue and fly ash are the two most feasible

280   media for use in water filters, based on contaminant removal. However, these two media

281   were also prone to clogging. To carry these media forward to a pilot-scale study, a redesign

282   of the filter configuration is required, to assess whether clogging is as likely to occur if these

283   media were not at the uppermost layer of the filter. In some locations, the removal of DOC

284   may be more important than the footprint of the filter, in which case these media may indeed

285   be the most successful. The adsorption capacity of both media is undoubtedly powerful, as

shown in the filter performance in this study, and is well documented in literature (Cengeloglu et al. 2006; Li et al. 2006; Kurniawan et al. 2006; Bhatnagar and Sillanpää 2010). Laboratory-scale configurations have not been previously investigated, and a lack of follow-through from bench-scale adsorption testing to laboratory or pilot-scale continuous flow experiments has been noted in the literature (Ali 2012; Bhatnagar & Sillanpää 2010). However, for some treatment plants, footprint of the filter is essential, and the permeability of the media may be most important. Cost may also be a criterion in filter selection, as media costs can differ extensively from location to location. For example, countries with reserves of natural zeolite may find it an extremely low cost material, whereas countries relying on production of synthetic zeolites may find it to be an expensive resource and it may not be feasible for use (Misaelides 2011). It is also important to note the risks that may be present when dealing with industrial by-products such as Bayer residue and fly ash, namely, the potential for metal leaching from the media. Aluminium and iron were the only metals found in desorption studies (Grace et al. 2015), but these can be absorbed using a combination of filter media (Grace et al., unpublished data). In order to harness the adsorption potential of filter media, yet prevent any leaching of contaminants, a stratified filter must be carefully designed to target the removal of specific contaminants, and toxicity analysis of the final effluent must be conducted, particularly during the initial stages of operation.

Across all configurations and both loading regimes, the build-up of organic matter was the main clogging mechanism, shown by observation, hydraulic conductivity analysis, and chemical characterisation of the percentage organic matter present. Based on performance alone, Config 2 was the most effective filter configuration, and continuous loading was most effective regarding DOC removal. However, the fine particles of fly ash meant that clogging was a significant problem, meaning that this option is not viable for long-term use. Config 1 was consistent across both loading regimes and the clogging layer did not extend as deeply as

311 in Config 2. This means it would be more cost effective for replenishment. The Control

312 performed best in the constantly loaded regime, although the clogging layer extended 0.1 m

313 deeper in this regime than in the intermittently loaded regime. This would suggest that the

314 current method of removing 0.15 - 0.3 m of sand, in 20 or 30 stages, is in fact, a conservative

315 estimate of the extent of clogging, and that intermittently loaded filters would require less

316 sand removal than constantly loaded filters.

4. Conclusion

318 The current model of a sand filter for water treatment is very effective for certain

319 contaminants. However with urbanisation, population growth and industrial development,

320 there is constant pressure on current water resources and infrastructure to meet the demand

321 for supply and treatment quality. With variations from place to place in contaminants and

322 source water quality, a more effective approach may be to focus on site-specific designs. The

323 use of novel media could allow this, as well as targeting contaminants that fail to be removed

324 by traditional sand filters. For a thorough approach to media selection, it is important to look

325 at performance potential, but it is also vital to understand the clogging mechanisms. To date

326 there has been little prior research on clogging of drinking water filters. This study focuses on

327 clogging of novel media, while maintaining efficient water treatment.

328 A layer-by-layer deconstruction of the filters showed that the main clogging mechanism in all

329 cases was a build-up of organic matter on the surface of the media. Although Configuration 2

330 (fly ash, GAC, zeolite and sand) was effective in DOC removal, this configuration was more

331 likely to clog than the Control (sand), therefore requiring a larger area for filtration. The flow

332 regime had an impact on the extent of clogging in the sand filters, with constantly loaded

333 sand filters found to have clogged to a deeper depth below the surface than those that were

334 intermittently loaded. The current method of re-sanding the slow sand filter (wherein 20 or 30

335   scrapings are carried out, removing 0.010 - 0.015 m each time) may be conservative, as the

336   clogging layer did not extend below 0.3 m, despite a high suspended solids concentration in

337   the influent. Further research should investigate the potential of using these media in an

338   alternative configuration to harness the positive adsorption of the media, while reducing the

339   potential for clogging. Following this, given the variable nature of both the media and the

340   water supply, site-specific testing is essential to the development of a design.

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Table 1: Previous use of media for adsorption

Media	Contaminant	Adsorption Capacity (mg g^{-1})	Comment	Reference
Bayer Residue	Phosphorus	1.1	Initial Concentrations between 0.1 and 1mg L^{-1}	Huang et al. 2008
		345.5	Activated Bayer residue	Li et al. 2006
	Nitrate	115.3	Raw Bayer residue	Cengeloglu et al. 2006
		363.2	Activated bayer residue	Cengeloglu et al. 2006
	Al	0.064		Komnitsas et al. 2004
Fly Ash	Phosphorus	75.4	Activated fly ash	Li et al. 2006
		42.6	Raw fly ash	Vohla et al. 2011
	Flouride	20.3		Bhatnagar et al. 2011
	Copper	2.36		Gupta & Ali 2000
	Zinc	2.54		Gupta & Ali 2000
Zeolites	Ammonium	2.13		Widiastuti et al. 2011
	Lead	9.97		Nguyen et al. 2015
	Copper	8.53		Nguyen et al. 2015
	Zinc	5.83		Nguyen et al. 2015

Table 2: Characterisation of media

Media	Sand[a]	Zeolite[b]	Bayer Residue[c]	Fly ash[d]	GAC[e]
Chemical					
SiO_2 (%)	97.72	65-72	8.9	60.37	-
Fe_2O_3 (%)	1.26	0.8-1.9	43.8	8.27	-
Al_2O_3 (%)	0.21	10.0-12	15.04	20.53	-
K_2O (%)	0.05	2.3-3.5	-	1.89	-
$L.O.I^f$ (%)	0.36	9.0-12	9.5	-	-
CaO (%)	-	2.5-3.7	6.6	2.26	-
MgO (%)	-	0.9-1.2	0.09	1.95	-
Na_2O (%)	-	0.3-0.65	5.32	0.65	-
TiO_2 (%)	-	0-0.1	9.2	1	-
MnO (%)	-	0-0.08	-	-	-
SO_4 (%)	-	-	0.41	2.15	-
P_2O_5 (%)	-	-	0.36	0.22	-
Iodine No (mg gm^{-1})	-	-	-	-	1100
Moisture (%)	-	-	-	-	5
Ash (%)	-	-	-	-	4
Un-determined		-	-	0.71	-
Ca (%)*	-	-	-	-	9.85
K (%)*	-	-	-	-	77.73
Al (mg kg^{-1})*	87	263	8388	1223	49
Cu (mg kg^{-1})*	4.65	1.18	4.17	4.35	0.55
Fe (mg kg^{-1})*	39	23	59	189	14
Mg (mg kg^{-1})*	22.57	12.8	0.18	13.59	3.78
Mn (mg kg^{-1})*	3	15	1	22	2
Na (mg kg^{-1})*	15	655	18280	175	219
P (mg kg^{-1})*	4	3	10	1044	87
Zn (mg kg^{-1})*	1.37	1.64	0.4	4.43	0.4
Total exchange capacity (meq 100 g^{-1})*	0.96	22.13	108.8	0.06	0.58
Organic Matter (%)*	0.1	0.1	5.7	0.22	72.3
Effective size, d_{10} (mm)	1.31	1.02	0.06	0.06	0.58

[a] Irwin's Quality Aggregates; [b] Zeolite Ireland Ltd; [c] Rusal Aughinish; [d] ESB Energy International Moneypoint; [e] Indo German Carbons Ltd
[f] Loss on Ignition *Analysis by Brookside Laboratories

Table 3: Filter Performance after 90 days of operation with standard deviation

		Continuous Loading			Intermittent Loading		
		Control	**Config 1**	**Config 2**	**Control**	**Config 1**	**Config 2**
Average hydraulic loading rate[a] (L m^{-2} d^{-1})		1424 ± 48	600 ± 268	577 ± 12	676 ± 30	592 ± 228	655 ± 208
Influent (mg L^{-1})	**DOC**	5.7 ± 2.2	5.7 ± 2.2	5.7 ± 2.2	6 ± 1.7	6 ± 1.7	6 ± 1.7
	Al	1.5 ± 0.2	1.5 ± 0.2	1.5 ± 0.2	1.5 ± 0.2	1.5 ± 0.2	1.5 ± 0.2
	NH_4^+-N	5.9 ± 2	5.9 ± 2	5.9 ± 2	5.8 ± 1.3	5.8 ± 1.3	5.8 ± 1.3
	NO_3^-N	19.9 ± 3.2	19.9 ± 3.2	19.9 ± 3.2	18.3 ± 4.1	18.3 ± 4.1	18.3 ± 4.1
Turbidity (NTU)	**Influent**	19 ± 7	19 ± 7	19 ± 7	19 ± 4	19 ± 4	19 ± 4
	Effluent	6 ± 1	4 ± 2	2 ± 1	6 ± 1	5 ± 3	4 ± 3
% Removal	**DOC**	16.7 ± 3.3	34.9 ± 10.9	71.5 ± 4.8	24.3 ± 2.5	34.4 ± 4.8	63.3 ± 10
	Al	93.6 ± 3.0	93.5 ± 3.2	97.6 ± 0.2	94 ± 0.8	94.7 ± 4.4	96.4 ± 2.9
	NH_4^+-N	56.5 ± 1.9	75.9 ± 10.9	88.5 ± 1.9	60.2 ± 3	84.7 ± 2.1	84.8 ± 2.8
	NO_3^-N	-29.8 ± 10.3	-28.1 ± 5.4	12 ± 1.6	-30.6 ± 13.9	-44.3 ± 5.4	-16.3 ± 19.3
Avg mass removal (g m^{-3} d^{-1})	**DOC**	1.6 ± 0.25	1.3 ± 0.36	2.4 ± 0.15	1.0 ± 0.08	1.2 ± 0.13	2.4 ± 0.38
	Al	2.6 ± 0.06	0.9 ± 0.02	0.9 ± 0.003	1.0 ± 0.01	0.9 ± 0.04	1.0 ± 0.03
	NH_4^+-N	4.6 ± 0.45	2.7 ± 0.33	3.0 ± 0.09	2.4 ± 0.19	2.9 ± 0.07	3.2 ± 0.09
	NO_3^-N	-7.9 ± 5.73	-3.3 ± 0.73	1.8 ± 0.12	-3.5 ± 1.82	-4.4 ± 0.51	-1.7 ± 2.23

[a]Average hydraulic loading rate was calculated from the initial and final hydraulic loading rates, to best describe the columns as clogging occurred.
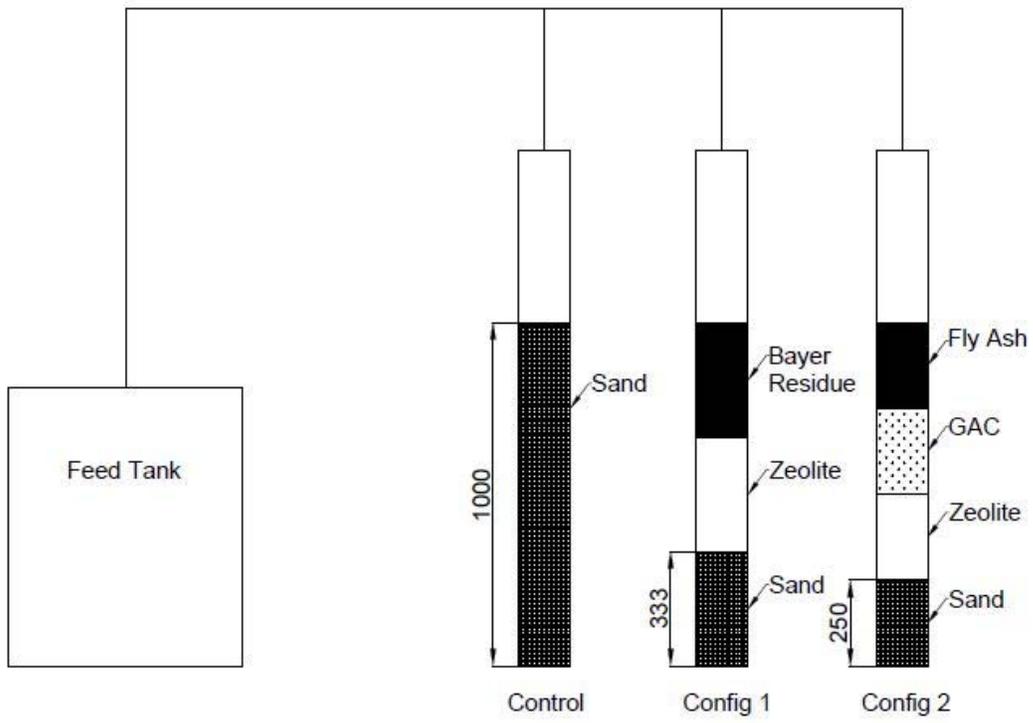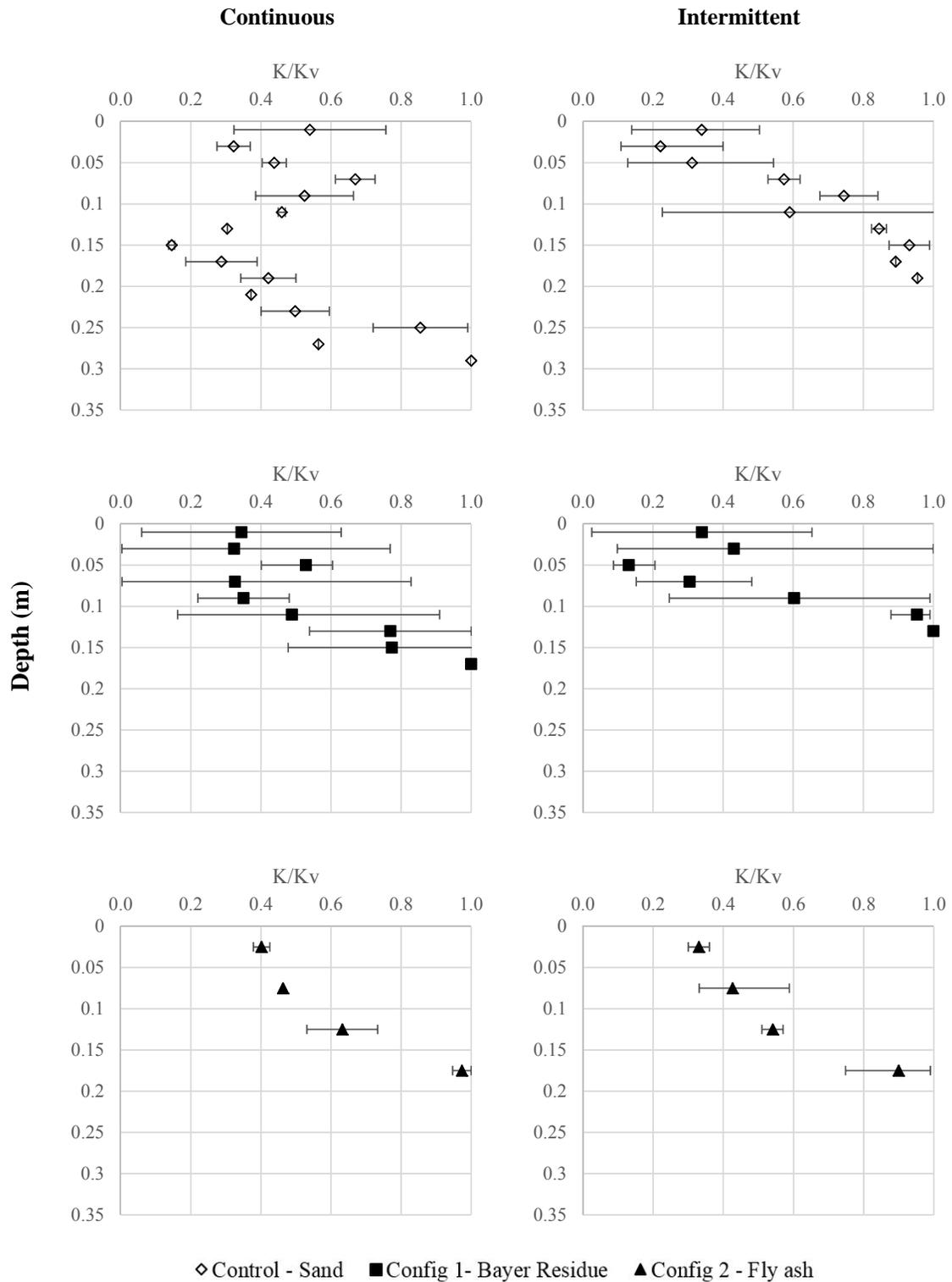
Figure 1: Filter configurations

Figure 2: Relative hydraulic conductivity variation in the uppermost layer with maximum and minimum values
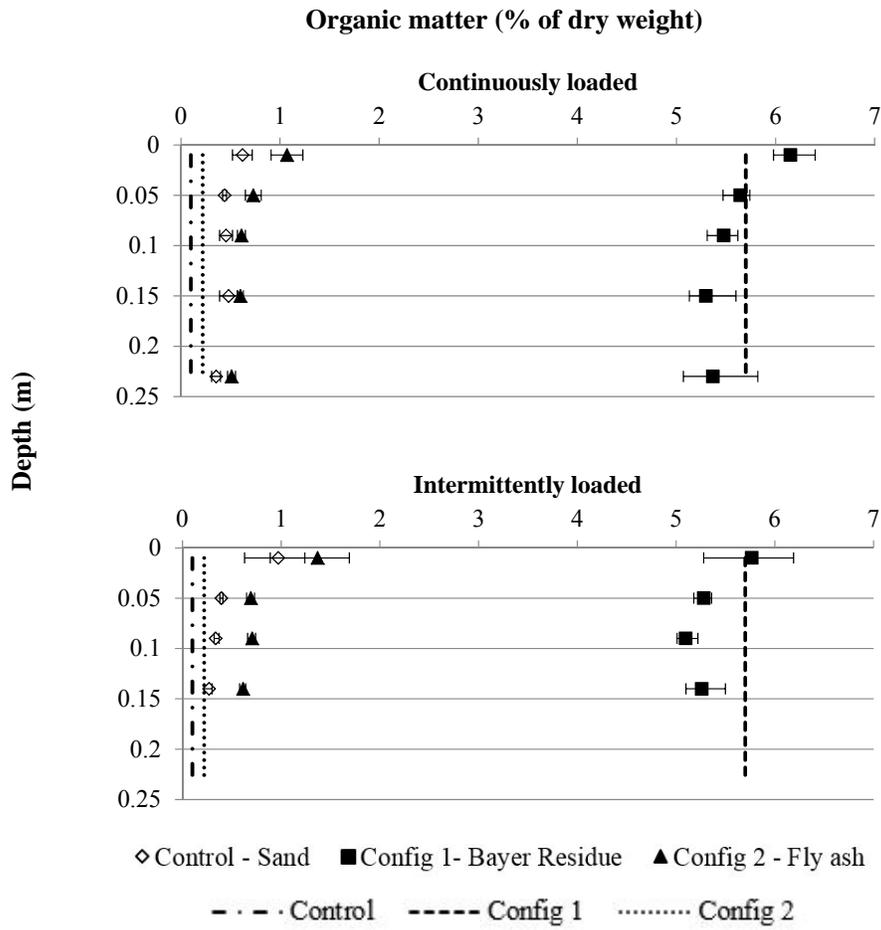
Organic matter (% of dry weight)

Figure 3: Organic matter percentages of dry weight of media (with max and min) per depth in each configuration, with initial concentrations shown as vertical lines