



Research article

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



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ARTICLE INFO

Article history:

Received 15 February 2016

Received in revised form

6 May 2016

Accepted 9 May 2016

Keywords:

Clogging mechanism

Water purification

Fly ash

Bayer residue

ABSTRACT

Slow sand filters are commonly used in water purification processes. However, with the emergence of new contaminants and concern over removing precursors to disinfection by-products, as well as traditional contaminants, there has recently been a focus on technology improvements to result in more effective and targeted filtration systems. The use of new media has attracted attention in terms of contaminant removal, but there have been limited investigations on the key issue of clogging. The filters constructed for this study contained stratified layers comprising combinations of Bayer residue, zeolite, fly ash, granular activated carbon, or sand, dosed with a variety of contaminants (total organic carbon (TOC), aluminium (Al), ammonium ($\text{NH}_4\text{-N}$), nitrate ($\text{NO}_3\text{-N}$) and turbidity). Their performance and clogging mechanisms were compared to sand filters, which were also operated under two different loading regimes (continuous and intermittently loaded). The study showed that the novel filter configurations achieved up to 97% Al removal, 71% TOC removal, and 88% $\text{NH}_4\text{-N}$ removal in the best-performing configuration, although they were not as effective as sand in terms of permeability. Deconstruction of the filters revealed that the main clogging mechanism was organic matter build-up at the uppermost layer of the filters. The clogging layer formed more quickly on the surface of the novel media when compared to the sand filters, but extended further into the sand filters, the extent dependent on the loading regime. The study shows the potential for an alternative filtration configuration, harnessing the adsorption potential of industrial waste products and natural media.

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1. Introduction

Slow sand filters are commonly used in water treatment to remove contaminants by physical, chemical and biological mechanisms (EPA, 1995), but they may not be effective in the removal of specific contaminants, 'emerging contaminants' (EC), or precursors to disinfection by-products (DBP), such as dissolved organic carbon (DOC) (Gang et al., 2003; Chuang et al., 2011; EPA, 2015). Consequently, there has been much research into the use of alternative media, particularly industrial by-products and waste products, or coated media, for use in filters for the treatment of targeted contaminants (Bhatnagar and Sillanpää, 2010; Fu and Wang, 2011; Rahman et al., 2013). Although these media have been examined extensively at bench-scale level, their efficacy has been infrequently examined in laboratory, pilot or large-scale filters (Bailey et al., 1999; Bhatnagar et al., 2011). This is a major knowledge

gap, as a layered configuration in a water filter may allow the utilisation of adsorption properties from each of the media, and may have the ability to control any potential metal leaching. Moreover, the use of waste materials in a filtration unit is potentially an effective and sustainable means of water treatment. While the viability of these media for use in filters depends on their efficacy in the removal of contaminants, it equally depends on their capacity to treat water without clogging.

Clogging mechanisms in sand filters for the treatment of wastewater (Hatt and Fletcher, 2008; Leverenz et al., 2009) and vertical flow constructed wetlands, which operate in a similar way to filters (Turon et al., 2009; Pedescoll et al., 2009; Hua et al., 2010; Knowles et al., 2011), have been well researched. However, clogging mechanisms in filters for drinking water treatment have not been examined to the same extent. Biological clogging has been the main focus of the research to date (Thullner et al., 2002; Kildsgaard and Engesgaard, 2002; Mauclaire et al., 2004); however, clogging may also occur by chemical and physical mechanisms (Mauclaire et al., 2004; Le Coustumer et al., 2012). This may be particularly

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relevant for water filters. Chemical clogging can affect the shapes and stabilities of the pores in the media, in turn, affecting the flow paths (Baveye et al., 1998). Adsorption of substances and metal precipitation from the influent water may also contribute to clogging within filters (Noubactep et al., 2010). Physical clogging may result from compaction due to loads on the surface of the filter and migration of the fine media into the filter. Therefore, the use of a particularly fine media at the surface may result in a filter cake forming at the media-water interface, contributing to physical clogging (Baveye et al., 1998). Where organic carbon is a component of the influent water, clogging is expected (McKinley and Siegrist, 2011), which may be due to the extracellular polymer substances (EPS) sometimes present in humic acid. These may form a gel-like, hydrophilic structure as the humic acid accumulates (Tanner et al., 1998), increasing the retardation of flow within the filter.

Filter head loss is the most common method of determining clogging at operational facilities (EPA, 1995). Clogging of filter media may be investigated in a number of ways. One of the most common methods is to measure the field saturated hydraulic conductivity (K_{fs}) (Rodgers et al., 2004; Pedescoll et al., 2009; Le Coustumer et al., 2012). As the filter clogs over time, K_{fs} decreases (Knowles et al., 2011). This can be measured using either a falling head test (ASTM, 2007) or constant head test (British Standard Institution, 1990b), depending on the permeability of the media under consideration. Other common methods of analysis include loss-on-ignition (LOI), chemical analysis of the media at different depths throughout the filter, scanning electron microscopy (SEM) of the biofilm layer, and X-Ray diffraction (XRD) analysis (Rodgers et al., 2004; Pedescoll et al., 2009; Knowles et al., 2010; Nivala et al., 2012).

Clogging becomes evident in filters as surface ponding occurs (for intermittent filters) and the outflow flow rate decreases (Knowles et al., 2011). It is important to ascertain how deep the clogging layer is within the filter, as the filter can be regenerated and the hydraulic conductivity restored. This may be accomplished by replacing the clogging layer of the filter with fresh media (Mauclair et al., 2004). Current guidelines advise the removal and re-sanding of slow sand filters once a predetermined design head loss has been reached (EPA, 1995). However, re-sanding beyond the clogging layer leads to excessive and unnecessary cost.

In the current paper, the water contaminants studied were DOC, aluminium (Al), nitrate-nitrogen ($\text{NO}_3\text{-N}$), ammonium-nitrogen ($\text{NH}_4\text{-N}$) and turbidity. Dissolved organic carbon present in source water causes formation of DBPs such as trihalomethanes and haloacetic acids following chlorination, and oral ingestion of DBPs can lead to cancer (Wang et al., 2007). Removal of DOC at filtration stage would reduce the potential of DBP formation, which has been the main source of non-compliance in Irish water treatment plants (WTPs) in recent years (Water Team, 2012). Aluminium is added to water during the treatment processes, but only represents a problem when it is present in the source water due to geological leaching (Calderon, 2000). The maximum allowable concentration (MAC) for Al in water is $200 \mu\text{g L}^{-1}$ (SI No 278 of 2007), and epidemiological studies have previously established a link between excessive Al and Alzheimer's disease (Flaten, 2001; Bondy, 2010; Exley and Vickers, 2014), as well as having other human toxicity effects (Nayak, 2002). The MAC for $\text{NO}_3\text{-N}$ in drinking water is 10 mg L^{-1} (SI No 278 of 2007), as consumption of high levels of $\text{NO}_3\text{-N}$ may cause methemoglobinemia, and can have significant environmental impacts on agricultural and aquatic life (Bhatnagar and Sillanpää, 2011). Ammonium has a MAC of 0.3 mg L^{-1} in drinking water, and is generally present in source water due to anthropogenic activities. A residual presence of $\text{NH}_4\text{-N}$ at the disinfection stage can cause extra chlorine consumption, and may have a negative effect on the disinfection process (Feng

et al., 2012), and can suppress pH and dissolved oxygen in the supply system (Wilczak et al., 1996). Legislation states that turbidity must be acceptable to consumers and have no abnormal change (SI No 278 of 2007), except in the case of surface water treatment where 1.0 NTU should be achieved. However, turbidity can also be an indication of likely DBP formation, and excessive turbidity can also inhibit disinfection (EPA, 2011).

This paper focuses on the use of two industrial by-products, Bayer residue ('red mud') and fly ash, and a natural medium, zeolite, combined with granular activated carbon (GAC) and sand, in layered filter configurations. Bayer residue is a waste product of the Al production industry, and is often stored in bauxite residue storage areas close to the production site (EAA, 2013). Fly ash, a by-product of incineration, is most commonly used in the manufacture of concrete (Mehta, 2002). Natural zeolites are known adsorbents of contaminants in water and wastewater treatment (Wang and Peng, 2010). Each of these has been used previously for adsorption of contaminants, although not in stratified filter configurations operated at laboratory-scale, and their maximum adsorption capacities are summarised in Table 1.

While the performance of slow and intermittently loaded filters is important in the selection of suitable filter media, the hydraulic function and permeability are also crucial parameters that need to be considered. Therefore, the aim of this study was to (1) determine the mechanisms of clogging of the filters and (2) assess the performance of intermittently and constantly loaded filters, each containing the novel filter media and operated for a period of 90 days, in the removal of contaminants (DOC, Al, $\text{NH}_4\text{-N}$, $\text{NO}_3\text{-N}$ and turbidity) of water.

2. Materials and methods

2.1. Filter construction

Three filter configurations, each replicated at $n = 3$, were examined with constantly loaded and intermittently loaded operational regimes, giving a total of 18 filters (Fig. 1). The first configuration ('Config 1') was a three-layer stratified filter (each layer had a depth of 0.33 m) containing (downwards from the filter surface) Bayer residue, zeolite and coarse sand. The second configuration ('Config 2') was a four-layer filter, with equal layers of 0.25 m media, containing (downwards from the filter surface) flyash (from coal combustion), GAC, zeolite and coarse sand. The effective particle sizes (d_{10}) of each medium are given in Table 2. The third configuration ('Control') was a 1-m deep single layer sand filter with an d_{10} of 0.18 mm and uniformity coefficient of 2.19 (EPA, 1995). The filter configurations were chosen based on adsorption results obtained by Grace et al. (2015), where $\text{NH}_4\text{-N}$ removal was focused at the surface of the filter, and Al, TOC, and $\text{NO}_3\text{-N}$ further down through the media. Locally available media were chosen, where possible. The configurations took cognisance of the K_{fs} of each media, which was measured prior to the experiment. Each filter had a free-board depth of 0.5 m above the filter surface. Physical and chemical characteristics of the media are detailed in Table 2.

2.2. Filter operation

The intermittently loaded filters were subjected to an initial loading rate (day 1 of experiment) of 0.1 m h^{-1} following the Irish EPA guidelines for slow sand filters (intermittent filters are not currently used for large-scale drinking water treatment) (EPA, 1995). The intermittent filters were dosed for 10 min every 2 h using a peristaltic pump (7528-10, Masterflex L/S Variable-Speed Drive). A head of water of 0.5 m was maintained above the constantly loaded filters.

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