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Influence of design and media amendments on the performance of stormwater biofilters

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ABSTRACT

Biofiltration systems are a promising retrofit option for site constrained urban areas due to the vertical arrangement of treatment stages that leads to a relatively compact footprint. Existing knowledge about the influence of their design and configuration on hydrological, stormwater pollutant removal, and long term performance is limited and this has been identified as a barrier to their widespread uptake. Long term simulations of lined and unlined biofiltration systems in four contrasting UK climatic regimes are used to comment on the influence of climate, ponding depth, biofilter to drainage area ratio, and infiltration rate on hydrological performance. The results show that local differences in climate significantly impact on performance, and that infiltration rates as low as 0.36 mm/hr are not suitable for high rainfall locations in the UK, unless the biofilter to drainage area ratio is greater than 10%. However, with higher infiltration rates (72 mm/hr) a biofilter occupying only 3% of the impermeable catchment area would be capable of infiltrating 97% of annual rainfall in central England. Preliminary results of adsorption and column tests to assess the effectiveness of media amendments, specifically, Zeolite and Granulated Activated Carbon, for dissolved copper and phosphate removal are here presented.

KEYWORDS

Pollution; Sewers and drains; Floods and flood works.

1 INTRODUCTION

Management of surface runoff is a substantial challenge in most urban areas. Conflicting requirements for preventing flooding as well as protecting the natural environment from pollution can lead to difficult decisions in system design and costly solutions (Ashley et al., 2015). In the UK, the implementation of Sustainable Drainage Systems (SuDS) is recommended by the Environment Agency (EA) and the Scottish Environment Protection

Agency (SEPA) for managing urban stormwater runoff. The terms Water Sensitive Urban Design (WSUD) and urban Green Infrastructures (GI) refer to similar concepts (Fletcher et al. 2014). In England and Wales, SuDS have been commonly considered as a solution to mitigate urban flooding, and the potential for diffuse pollution control has not been fully exploited (Defra, 2011). In contrast, in Scotland, SuDS were originally introduced to improve water quality, but with the requirement to safely convey design storms (Sepa, 2011; Fletcher et al. 2014). Ashley et al., (2015) highlighted the lack of knowledge concerning enhanced pollutant removal processes in SuDS, leading to uncertainties in modelling. They also argue that more confidence on SuDS performance is needed for establishing a regulatory framework thus enabling SuDS implementation to be routine in the UK. SuDS implementation within the new regulatory framework in England and Wales is discussed elsewhere (Ellis and Lundy, 2016). In the USA rising concerns about water pollution have been addressed through the Federal Water Pollution Control Act in 1948, amended in 1972 and commonly known as Clean Water Act (CWA) (USEPA, 1972). The term Best Management Practice (BMP) was introduced for the first time in the CWA and proposed as a method to limit and control pollutants from municipal storm runoff (Fletcher et al. 2014). Pollution control was the driver for research on stormwater runoff monitoring and BMPs in the USA (Hunt et al., 2008, Sansalone and Buchberger, 1997). In Australia, similarly to the UK, the lack of knowledge on SuDS maintenance, the scarcity of monitoring of implemented SuDS and performance assessment, as well as difficulties in assessing long term cost/benefits have been identified as impediments to their uptake by water sector practitioners and prevented councils from developing strong business cases for SuDS projects (Sharma et al., 2016).

The pollutant load associated with stormwater runoff from urban areas consists of a heterogeneous mixture of constituents and it is generally very site specific (Lundy et al., 2012). Roof and road runoff, however, contain both dissolved and particle-bound pollutants that are of major concern for water quality (Andradóttir and Vollertsen 2014; Hvitved-Jacobsen *et al.*, 2010). For particulates, the fraction finer than 150 µm typically represents 90% of the mass deposited; finer fractions are more mobile and have higher affinity with pollutants (Revitt et al., 2014) than coarser fractions. Most of the literature doesn't distinguish between particulate-bound and dissolved pollutants, the latter being the most bioavailable and mobile fraction in urban runoff, more difficult to control and of major concern for the aquatic environment (Gnecco et al., 2008; LeFevre et al., 2015; Lim et al., 2015; Maniquiz-Redillas & Kim, 2016).

Biofiltration systems are one type of vegetated SuDS that are particularly suited for diffuse pollution control. They consist of a vegetated surface layer, multi-layered growing media (0.7-1.0 m deep), and they can be designed for infiltration (unlined system) or for draining to the sewer system (lined system). Biofilters have been adapted to potentially incorporate a number of different designs (Bratieres et al., 2008; Hatt et al., 2008). Vegetation contributes to dissolved pollutant removal through uptake and accumulation in the plants (Read et al., 2010). Media may be engineered to facilitate adsorption of selected dissolved pollutants (Grebel et al., 2013; Pitt et al., 2011). Microbial uptake and degradation is also an important removal mechanism in biofilters, and the presence of zones in the filter bed with air input (for example facilitated by plant roots) and anaerobic zones, typically generated by a saturation zone, allows for engineering of microbiologically mediated treatment (Feng et al., 2012; Zinger et al., 2013; LeFevre et al., 2015).

The vertical arrangement of treatment stages leads to a relatively compact footprint, making biofiltration systems a promising retrofit option even for highly constrained sites. The flexibility with respect to design and pollutant removal mechanisms means that biofilters provide a very adaptable tool for stormwater runoff treatment for many different applications. However, it also generates a requirement for characterisation of the different treatment functions and their interactions in order to tailor functionality to individual sites, both based on the pollutants present in the runoff and the ecological requirements of recipient natural water bodies. The design and choice of plants, media composition, and drainage configuration all impact upon hydrological, water quality and long term performance. LeFevre et al., (2015) highlighted the need for further research to investigate dissolved pollutant fate in biofilters and the use of media amendments to enhance their removal. Biofilter bed lifetime predictions are uncertain due to high variable pollutant loading, and hydraulic failure or loss of pollutant sorption capacity. Amendments have the potential to increase the biofilter adsorption capacity and thus, to extend the biofilter lifetime. Zeolites and Granular Activated Carbon (GAC) are among soil amendments for metal removal suggested by other researchers (Grebel et al., 2013; Pitt et al., 2011).

The current research, carried out by the Universities of Leeds, Sheffield and York, aims to develop a deep understanding of biofiltration system performance, leading to system refinements that optimise performance for prioritised hydrological and water quality objectives. In this paper preliminary findings of this research project are presented. In particular, long term simulations of lined and unlined biofiltration systems in four contrasting UK climatic regimes are used to comment on the influence of climate and design options on hydrological performance. A laboratory test to investigate the influence of amended soil media on treatment performance has been undertaken at the University of Leeds. In this paper we report preliminary results of adsorption and column tests that focused on the removal of dissolved copper and phosphate using Zeolite and GAC as soil amendments. Copper was considered because is one of the selected stormwater priority pollutants representative of anthropogenic sources (Ericksson et al., 2007). Phosphate was selected because its removal is complicated by potential leaching from biofilter media (Liu & Davis, 2014). These laboratory tests were also aimed at identifying the best configuration to undergo field testing in a commercial carpark.

2 METHODOLOGY

2.1 Biofilter Hydrological Model

Any discussion of biofilter water quality performance must be underpinned by a reasonable understanding of the key processes that determine the system's hydrological performance.

Whilst individual systems will naturally vary in specific details, the basic hydrological mechanisms can be summarised as follows. Stormwater control may be quantitatively described in terms of retention (i.e. the amount of rainfall and runoff that is intercepted and held within the biofilter) and detention (i.e. the lag and attenuation of the runoff hydrograph). Retention processes include interception and evapotranspiration by plants, as well as (in unlined systems) infiltration into the natural soil. During a storm event a biofilter (Figure 1) will retain stormwater within the substrate's micropores until it reaches field capacity. The retained water will eventually return to the atmosphere via evapotranspiration. Once the growing media has reached field capacity, any subsequent rainfall will start to fill larger pores, increasing the soil moisture from field capacity towards saturation. Most biofilters are designed to allow ponding to occur once saturation reaches the surface. Moisture in excess of field capacity will be returned to the ground via infiltration (i.e. the downward movement of rainwater through soil) if the biofilter is unlined, or directed into a storm or combined sewer if the system is lined.

[Approximate location of Figure 1]

The overall hydrological performance of a system depends on a number of key variables: the rainfall characteristics of the location; the area of the biofilter relative to its impervious catchment; the ponding depth; substrate and drainage layer porosity; the evapotranspiration characteristics of the biofilter's vegetation; and the drainage characteristics of the system. The drainage rate depends on either the infiltration capacity of the local soil (unlined system) or the outflow restriction placed between the biofilter and the sewer (lined system). Here we outline a simple conceptual hydrological model of a biofilter, which is then utilised to explore the sensitivity of hydrological performance to four of these variables: the local climatic input, the ponding depth, the area of the biofilter relative to its impervious catchment and the drainage rate.

For green infrastructure/SuDS it is increasingly argued that hydrological performance needs to be assessed using time-series rainfall inputs. Partly this reflects the critical importance of antecedent dry weather conditions in determining the soil moisture deficit; antecedent conditions must be assumed for design rainfall-based analyses, with considerable scope for uncertainty/error. Continuous simulation provides a more complete statistical picture of the system's response to both everyday and extreme rainfall events. Everyday events are particularly important when the implementation of these systems is driven by water quality objectives.

From an engineering perspective, models based on water balance concepts may be used for design and comparison purposes, provided that the key hydrological processes (i.e. evapotranspiration, in-/exfiltration and drainage) are properly accounted for. Winston et al. (2016) utilised an agricultural water balance model, DRAINMOD, which was calibrated and validated for three biofiltration systems in the USA. The model was utilised to explore the sensitivity of the water balance to different input parameters. Not surprisingly, the overflow amount was shown to be highly sensitive to the loading ratio (the ratio of biofilter surface area to contributing catchment area) and to the infiltration capacity of the underlying soil. Winston et al. (2016) also demonstrated the value of continuous simulation modelling for understudying any changes in performance that might arise due to climate change. Daly et al. (2012) implemented a stochastic version of the basic water balance model to enable predictions about treatment performance.

Stovin et al. (2013) presented a modelling based study of green roof retention performance. The study highlighted striking differences in performance for identically-configured green-roof systems subjected to a range of four contrasting UK climatic inputs, corresponding to Sheffield, East Midlands, NW Scotland and Cornwall. While UK climate is classified as temperate maritime according to the Köppen-Geiger climatic classification, mean temperatures vary on a North-South gradient with mean temperatures of over 11°C (South-East) to <4°C (central Scotland). Rainfall varies on a predominantly West-East gradient from >3000mm (West) to <600mm (East). Annual climatic regimes for these four locations were generated with the UKCP09 weather generator tool (UK Climate Projections –

http://ukclimateprojections.defra.gov.uk/) and are presented in detail in Stovin et al. (2013). In summary, the locations range in mean annual rainfall from 496 mm to 2,708 mm (Sheffield: 838 mm; East Midlands: 496 mm; NW Scotland: 2,708 mm; Cornwall: 1,365 mm). Mean monthly temperatures range from 3.5°C (lowest winter monthly mean) to 20.8°C (highest summer monthly mean). Specific mean monthly temperature ranges are as follows: Sheffield: 3.5-16.9°C; East Midlands: 6.2-20.8°C; NW Scotland 4.2-16.1°C; and Cornwall: 8.2-15.6°C). The same four rainfall time series and temperature inputs were used to explore the influence of climatic inputs on biofilter systems using a water-balance continuous simulation model.

Two contrasting biofilter to impermeable catchment area scenarios were tested in the simulations: 10% and 3%. The 10% scenario was based on recommendations made in American design guidelines (e.g. Emanuel et al., 2010 and PADEP, 2006). However, the CIRIA manual (Woods-Ballard et al., 2015), based on the Australian Facility for Advancing Water Biofiltration guidelines (Hatt et al., 2009), suggested that the biofilter should occupy 2-5% of the overall area to be drained, recommending a 400-600 mm ponding depth and a soil infiltration rate between 100 to 300 mm/h. For the 10% ratio, a 100 m^2 catchment comprising a 10 m^2 biofilter facility and 90 m^2 impermeable area was assumed. This could represent, for example, a 1 m wide strip serving a 9 m wide road and pavement.

The substrate depth was assumed to be 400 mm, with field capacity of 0.4 m^3/ m^3. Air-filled porosity was assumed to be 0.3 m^3/ m^3 at field capacity. The underlying drainage layer was assumed to be 0.2 m deep, with an air-filled porosity of 0.4 m^3/ m^3.

Hydrological performance was tested against recommended restrictive criteria for infiltration into natural soil for unlined systems and for discharge rate for lined SuDS. Two infiltration/drainage rates were evaluated. The first assumes that the maximum outflow from the biofilter is restricted to a greenfield runoff rate of 2 l/s/ha. This equates to 0.02 l/s over the 100 m^2 catchment area, equivalent to a depth rate for the 10 m^2 biofilter of 0.002 mm/s or 7.2 mm/hr. Where the biofilter is intended to drain into the natural sub-soil, it is typically recommended that the infiltration rate should be no lower than 1 x 10^{-7} m/s (Woods-Ballard et al., 2015). This is equivalent to 0.36 mm/hr, which defines the lower drainage rate examined here and represents the worst-case, most conservative criterion.

Three overflow designs allowing ponding depths of 100, 200 and 400 mm were tested.

Evapotranspiration (ET) was assumed equal to Potential ET (PET), calculated using the Thornthwaite formula, as per Stovin et al., 2013. ET was assumed not to occur if there was insufficient soil moisture available. Whilst it is expected that actual ET rates would fall below PET when soil moisture falls below field capacity (Berretta et al., 2014), the far deeper substrates in biofilters compared with green roofs justifies the omission of this effect from this preliminary model.

2.2 Laboratory evaluation of biofilter pollutant removal capacity

The first phase of laboratory testing focused on dissolved metals and phosphorus.

The experimental set-up (Figure 2a) included 4 PVC columns (internal diameter 150 mm, height 1300 mm), with sandblasted inner wall for preferential flow prevention. The outlet drainage consisted of a perforated pipe (internal diameter 20 mm) at the bottom of the column and a raised outlet to establish a saturation zone 400 mm deep. The set-up included inlet and outlet 25 L tanks and a peristaltic pump for constant inflow rate.

The tested biofilters were designed in accordance with CIRIA and Monash University guidelines (Woods-Ballard et al., 2015; Payne, et al., 2015). Vegetation was not assessed within this study, but will be included in subsequent laboratory and field tests. A control column (C), which consisted of a filter media, a saturation zone and a drainage layer at the bottom, was tested against three configurations that included an amendment layer below the filter media. The three tested configurations with amendments included a Zeolite layer (Z), a Granular Activated Carbon layer (GAC), and a combination of both (Z+GAC).

The chosen amendments are commonly used in water treatment (Chaudhary et al., 2003; Kim et al., 2009; Wong et al., 2013), but to the best of our knowledge have been little used for runoff treatment through biofiltration systems. Both GAC and Zeolite have a high cation exchange capacity and a high specific surface area (respectively 350-1000 m^2g^{-1} and 30-180 m^2g^{-1}) favouring adsorption capacity (Grebel et al., 2013; Yates, 1968) and providing a growing surface for microorganisms (Baraee et al., 2015; Feder et al., 2015; Srivastava & Majumder, 2008). GAC has a high capacity to adsorb PAH without compromising plant health (Hale et al., 2012).

The tested configurations (Figure 2b) were characterized from top to bottom by:

- Protection layer (100 mm): 5.0-8.0 mm coarse gravel;
- Top filter layer (100 mm): mix (by volume) of 60% 0.6-1.8 mm sand, 10% vermiculite, 10% perlite, 20% topsoil;
- Filter media layer (200 mm): mix (by volume) of 80% 0.6-1.8 mm sand, 10% vermiculite, 10% perlite. In C, this layer is 300 mm;
- Amendment layer (100 mm): 0.7-1.6 mm Zeolite or 0.59-1.68 mm GAC. In the Z+GAC configuration, this layer was divided equally in two, one for each amendment. In C, this layer was absent.
- Transition layer (100 mm): 0.15-0.30 mm sand;
- Saturation layer (300 mm): 3 layers of 100 mm height characterized by a mixture (by volume) of 5% woodchips and sand of different gradation: 0.3-0.6 mm, 0.6-1.18 mm and 1.18-2.36 mm sand;
- Drainage layer (100 mm): characterized from top to bottom: 2.0-3.0 mm fine gravel (30 mm), 3.0-5.0 mm medium gravel (30 mm), 5.0-8.0 mm coarse gravel (40 mm).

Each media was light hand-compacted according to construction practice for biofilter columns used in research studies (Le Coustumer et al., 2012).

Sand was supplied by David Ball Specialist Sands, Zeolite by RS Minerals (ZeoClin), and Granular Activated Carbon by Eurocarb (PHO 12x30 AW).

[Approximate location of Figure 2]

The synthetic stormwater consisted of tap water with added pentahydrate copper sulphate ($CuSO_4.5H_2O$), zinc chloride (ZnCl), and cadmium chloride (CdCl) to achieve a typical stormwater dissolved metals concentration for the UK roadside (Revitt et al., 2014). Phosphorus was already present in tap water as an additive to prevent lead from leaching in old pipes; consequently it has not been added. The inflow synthetic stormwater characteristics are presented in Table 1.

[Approximate location of Table 1]

The total applied volume of synthetic stormwater for each column was equivalent to an average annual runoff volume based on rainfall data for Leeds (UK Met Office, 1996-2015). The dosing volume was calculated based on a 2% biofilter to catchment area, and an average annual rainfall depth for Leeds of 711.2 mm. The filters were dosed with 25 L of water for each run for a total of 26 runs. The dry periods between two runs varied from a minimum of 0.5 to a maximum of 494 hours. It should be noted that the 2% biofilter to catchment area was adopted here to evaluate the performance of biofilters in the most conservative condition, with the lowest value recommended in the CIRIA manual (Woods-Ballard et al., 2015). This is justified by the need to maintain a compact footprint to maximize retrofitting potential. This criteria would be contingent on either the subsoil supporting high rates of infiltration or relatively high rates of discharge to sewer being permitted.

A grab sample of the inflow prior to dosing was taken for each run. All outflow water from each column was collected in tanks. The outflow sampling method varied within the test. Specifically, discrete samples were taken every 5 L outflow volume for the first 10 runs, every 12.5 L for the following 6 runs, and one composite sample for the remaining runs until the end of the test. At the end of each run the collected outflow was stirred and a sample was taken.

Flow rate was recorded at sample collection. Data is presented in terms of event mean concentrations (EMC) defined as the total pollutant mass of an individual event (or run) divided by its total runoff volume (Andradottir & Vollertsen, 2015). One litre of collected outflow has been sampled for the first 4 runs to assess suspended solids concentration and potential media wash-off.

An adsorption test was performed on the tested media. 1 g of media in 230 ml of synthetic stormwater with the same characteristics used for the column test (Table 1) was shaken in a glass bottle at 100 rpm on an orbital shaker table. Three replicates for each media were tested (Paus et al., 2014). Samples were taken after 24 hours. The adsorption equilibrium was determined using the following equation:

$$Q_e = \frac{(C_0 - C_{24h}) \times V}{W}$$

Where Q_e is the adsorption capacity at equilibrium (mg g^{-1}), C_0 initial solution concentration, C_{24h} concentration of solution after 24 hours, V is the volume of the solution (L), and W is the dry mass of adsorbent used (g) (Sansalone et al., 2013).

All water samples for the column and adsorption tests were analysed for pH, conductivity, dissolved oxygen, temperature, dissolved Cu, Cd, Zn and orthophosphate. For the metals analysis, inductively coupled plasma mass spectrometry (ICP-MS ELAN DRC-e) was performed. Orthophosphate was measured through the use of a HACH DR/890 with method 8048. A HQ40d portable multi parameter meter was used for pH, conductivity, dissolved oxygen, and temperature.

The removal performance of the biofiltration systems for each constituent was determined using the following equation:

$$RP = \frac{M_{inflow} - M_{ouflow}}{M_{inflow}} \times 100$$

Where RP is the constituent removal performance in %; M_{inflow} is the inflow mass of constituent and $M_{outflow}$ is the outflow mass of constituent (Fuerhacker et al., 2011).

3 RESULTS AND DISCUSSION

3.1 Biofilter hydrological performance

The results of the simulations are described in terms of stormwater inflow that either becomes evapotranspiration, infiltration/drainage, or it overflows from the system. Figure 3 shows the long-term distribution of inflow for the 10% biofilter to catchment area scenario for each of the four climatic regimes with each of the six different parameters (two infiltration/drainage rates; three ponding depths).

Figure 3 shows that, with the exception of the East Midlands location (where annual rainfall is relatively low and temperatures are relatively high), evapotranspiration makes only a small (<10%) contribution to the water budget. Jennings et al., (2015) also highlighted the relative unimportance of evapotranspiration compared with the infiltration rate. Conversely, the infiltration/drainage rate has a significant impact upon performance, with systems designed to permit up to 2 L/s/ha (i.e. high drainage rate) reducing overflows in all cases to less than 10% of total inflow volume. In these cases increasing the ponding depth does reduce overflows, but the overall benefit is minor. For the NW Scotland case, the combination of high rainfall and low evapotranspiration means that systems with a 10% area ratio, relying on low natural infiltration rates will not perform satisfactorily, overflowing almost all the inflow. In this case an increase in the ponding depth does offer improved performance, but with more than 85% overflowing it cannot be judged to be satisfactory. In contrast, with lower rainfall and higher ET rates, the same device located in the East Midlands would eliminate overflows with drainage restricted to 7.2 mm/hr, and overflow only 26-32% of the inflow volume (depending upon the ponding depth) even with the minimum drainage/infiltration rate.

[Approximate location of Figure 3]

This preliminary assessment highlights that local differences in climate significantly impact on the performance levels that can be expected. Infiltration rates as low as 0.36 mm/hr are not suitable for high rainfall locations in the UK, unless the drainage area ratio is significantly reduced compared with the 10:1 ratio considered here. Ponding depth – and by extension substrate depth and drainage layer depth – has a relatively small influence on performance, whereas the drainage/infiltration rate is far more critical.

Figure 4 shows the impact of reducing the ratio in the Sheffield scenario from 10% (S) to 3% (Sm). Not surprisingly, the amount of ET is reduced due to the smaller bioretention surface area. Similarly, removal due to drainage/infiltration will be reduced and overflow will be greater, equivalent to over 85% of rainfall with low drainage/infiltration capacity.

However, the final column in Figure 4 confirms that excellent performance can be achieved, even when the biofilter occupies only 3% of the catchment, as long as the drainage/infiltration conditions are favourable. With infiltration of 72 mm/hr, the annual overflow is less than 3% of annual rainfall.

[Approximate location of Figure 4]

3.2 Pollutant removal efficiency

The adsorption test results are shown in Table 2. They show the higher adsorption capacity of Zeolite and GAC over the sand for dissolved Cu with Qe respectively of 31.01, 39.77 and 4.24 mg/g. Topsoil is important for the removal of dissolved copper but, in contrast to zeolite and GAC, it may act as a source of phosphate. It is expected that the two amended media could extend the biofilter lifetime by increasing the adsorption capacity for dissolved metals when topsoil contribution diminishes.

[Approximate location of Table 2]

The outflow EMC values of pH, conductivity, temperature and dissolved oxygen for the four configurations are shown in Table 3. There was found to be no statistically significant difference between the inflow and outflow pH values in all configurations. However, the GAC-configuration outflow values were found to be generally higher than the inflow, as supported by the adsorption test data where the average outflow pH was 8.10 (±0.27) against an inflow value of 5.98 (±0.52) ($p>0.003$).

Measured dissolved oxygen showed a statistically significant difference between inflow and outflow ($p<0.001$) and correlated positively with the dry period between runs for all the tested configurations. This is probably due to saturation anoxic zone (Zinger et al., 2007) oxidation and microbial processes (Subramaniam et al., 2014). The system with GAC as an amendment layer showed the lowest concentration of dissolved oxygen among all the configurations. The anaerobic condition in the saturation zone favours denitrification processes (Blecken et al., 2010).

[Approximate location of Table 3]

The removal performance of dissolved Cu was evaluated for each configuration. The inflow concentration ranged from 177 to 377 μgL^{-1} with a median of 256 μgL^{-1} (Figure 5). The median removal performance for the C, Z+GAC, Z, and GAC configurations were 91% (±1.34%), 93% (±1.05%), 96% (±0.89%), and 96% (±0.49%) respectively (Figure 6). Data for Cu in runs 1-3 and 14 for the configurations C and Z+GAC were not available.

The results show that all four configurations were able to control dissolved Cu. There is a statistical difference for copper removal performance between the control and amended configurations. For the Z and GAC configurations, the removal performance behaviour was more consistent than for the C and Z+GAC configurations where greater variability and lower values were observed. By the end of the experiment, which corresponded to one year of inflow, a slightly decreased performance of the control configuration was observed compared to the amended columns. Further experiments are required to validate these results.

The median outflow concentration observed for configuration C, Z+GAC, Z, and GAC were 19 ug/L (± 2.77), 15 ug/L (± 2.65), 9 ug/L (± 1.57), and 7.78 ug/L (± 0.84) respectively.

Regulatory standards for copper to achieve “good” chemical status are provided depending on water hardness (according to The Water Framework Directive 2000/60/EC). For water hardness varying from 0 to values greater than 250 mgL^{-1}, the annual mean concentration of dissolved copper varies from 1 μgL^{-1} to 28 μgL^{-1}. The Italian Water quality standards for discharges into water bodies (Decree by Law 152/06) refers specifically to dissolved Cu limits of 100 μgL^{-1}. American discharge limits for total Cu are 4.8 μgL^{-1} and 13 μgL^{-1} for saltwater and freshwater respectively (Federal Register, CA, 2000).

[Approximate location of Figure 5]

[Approximate location of Figure 6]

The removal performance of orthophosphate was evaluated for each configuration. The inflow concentration values ranged from 1.84 to 3.07 mg L^{-1} with a median of 2.59 mg L^{-1} (Figure 7). The median removal performance values for C, Z+ GAC, Z, and GAC were 14% (± 22.77%), 14% (±17.93%), 18% (±19.73%), and 4.63% (±11.09%) respectively (Figure 8). These low removal efficiencies may be explained by the leaching of orthophosphate from the topsoil in the upper filter layer, as confirmed by the adsorption test results (Table 2).

Figurer 8 shows that dissolved phosphorous removal performance for C, Z+GAC, and zeolite configurations was initially above 30%, but dropped for the rest of the experiments. The mean removal performance values for these configurations were not statistically different from each other. The GAC configuration was statistically different from the other configurations ($p<0.05$), with lower performance throughout the runs, and whenever the inflow concentration was lower than the previous outflow concentration the columns became a source of orthophosphate rather than a sink. A possible explanation for the GAC low performance could be the correlation with the high values of pH measured in the outflow (Table 3a). As noted by Danilo and Burnham (1973), pH influences the free aluminum ion, which decreases with increasing pH. This could lead to a decreasing removal of phosphate (Danilo and Burnham 1973).

A moderate correlation has been found between dry periods and removal performance using a non-parametric Spearman test with $r_s(102)= 0.467$ and $p=0.005$. Higher performance was measured for longer dry periods (Figure 9). This correlation has also been observed in other studies, and is thought to be due to the formation of ferric oxyhydroxides favoured by increased oxygen levels (Hatt et al., 2007).

For water with alkalinity higher than 50 mg L^{-1} $CaCO_3$ the annual mean concentration of reactive phosphorus should be less than 120 µg L^{-1} (according to The Water Framework Directive 2000/60/EC). It has to be noted that these findings were as expected and other researchers showed low removal performance for configurations that did not include a specific substrate for phosphate removal (Weiss et al., 2016).

Recent research has suggested that iron-enhanced sands or aluminium oxidecoated media could be used for this purpose (Berretta & Sansalone, 2011; Glaister et al., 2014; Li & Davis, 2015; Weiss et al., 2016). Also plants, which were not included in this first series of tests, have a key role in phosphorus removal through root uptake processes (Read et al., 2010).

[Approximate location of Figure 7]

[Approximate location of Figure 8]

[Approximate location of Figure 9]

4 CONCLUSIONS

Biofilters are a promising retrofit option for site constrained urban areas due to the vertical arrangement of treatment stages that leads to a relatively compact footprint. Existing knowledge about the influence of their design and configuration on hydrological, contaminant removal, and long term performance is limited and this has been identified as a barrier to their widespread uptake.

A hydrological flux model was used to predict the long-term water budget of these systems. The model has been applied to 30 year hourly climate projections corresponding to four UK contrasting climates (Sheffield, East Midlands, NW Scotland and Cornwall). ET was assumed equal to Potential ET. Three overflow designs allowing ponding depths of 100 mm, 200 mm or 400 mm were tested. Two infiltration/drainage rates were evaluated: a restricted outflow equivalent to a greenfield runoff rate of 2 L/s/ha for lined biofilters, and the recommended minimum infiltration rate equivalent of 1 x 10^{-7} m/s in the case of unlined systems. Two contrasting biofilter to impermeable catchment area scenarios were tested: 10% and 3%.

The results of the simulations highlighted that local differences in climate significantly impact on performance. Infiltration rates as low as 0.36 mm/hr are not suitable for high rainfall locations in the UK, unless the biofilter to drainage area ratio is greater than 10%. Ponding depth – and by extension substrate depth and drainage layer depth – has a relatively small influence on performance, whereas the drainage/infiltration rate is far more critical.

The scenario of a reduced biofilter to impermeable catchment area equal to 3% showed reduced ET due to the smaller bioretention surface area. Similarly, removal due to drainage/infiltration will be reduced and overflow will be greater – equivalent to over 85% of rainfall – with low infiltration capacity. While the tested infiltration capacity is conservative, the model showed that satisfactory performance, i.e. reduced overflows, could be achievable with higher drainage/infiltration rates.

A laboratory study aimed at investigating the influence of amendments on the treatment performance, and ultimately at identifying biofilter configurations to undergo field testing in a commercial car park, has been performed. In particular, the effectiveness of Zeolite and Granular Activated Carbon on dissolved metals and phosphorus removal was investigated. Amongst the tested metals, only the results regarding dissolved copper have been presented here. A control column (C), which consisted of a filter media of sand, a saturation zone and a drainage layer at the bottom, was tested against three configurations that included an amendment layer: zeolite layer (Z), granular activated carbon layer (GAC), and a combination (Z+GAC). The columns were dosed with a volume equivalent to the average annual runoff for Leeds.

The laboratory tests revealed that the four tested configurations were able to control dissolved Cu. With a median inflow concentration of 256 μgL^{-1}, the median removal performance for the C, Z+GAC, Z, and GAC configurations was 91% (±1.34%), 93% (±1.05%), 96% (±0.89%), and 96% (±0.49%) respectively. By the end of the experiment a slightly decreased performance of the control configuration was observed in comparison to the amended columns.

For dissolved phosphorus, with a median inflow concentration of 2.59 mg L^{-1}, the removal performance for C, Z+ GAC, Z, and GAC was 14% (±22.77%), 14% (±17.93%), 18% (±19.73%), and 4.63% (±11.09%) respectively. Low retention of dissolved phosphorus is consistent with other studies. The tested columns did not include vegetation, which is known to play a significant role in P uptake. An adsorption test revealed leaching of orthophosphate by the topsoil in the upper filter layer. Zeolite and GAC have a higher adsorption capacity than sand, respectively with 0.044, 0.055, and 0.012 mg/g. Despite this, during column experiments GAC performed the worst, suggesting that the higher outflow measured pH could have influenced the adsorption process. Longer dry periods between runs were shown to correlate with higher removal performance which is thought to be due to the formation of ferric oxyhydroxides favoured by increased oxygen levels.

Measured dissolved oxygen showed a statistical difference between inflow and outflow ($p<0.001$) and correlated with dry periods between two runs for all the tested configurations. This is probably due to the saturation anoxic zone oxidation and microbial processes.

The results in this paper are preliminary and further testing is required. The next phase of laboratory work will include tests of amended media targeting phosphorus, extending the experiments with flow volumes representative of more than one year of runoff and tests of different plant species. Also, the path of pollutants within the biofilter will be investigated through destructive tests to reveal the effective contribution of each media layer to the biofilter performance and allow costs of different configuration to be assessed. Finally, the optimum configuration will be field tested in a commercial carpark within a long term monitoring programme that will permit the measurement of key hydrological and pollutant removal processes in time, provide essential information for maintenance and support the implementation, calibration and validation of biofilter modelling approaches.

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TABLES AND FIGURES

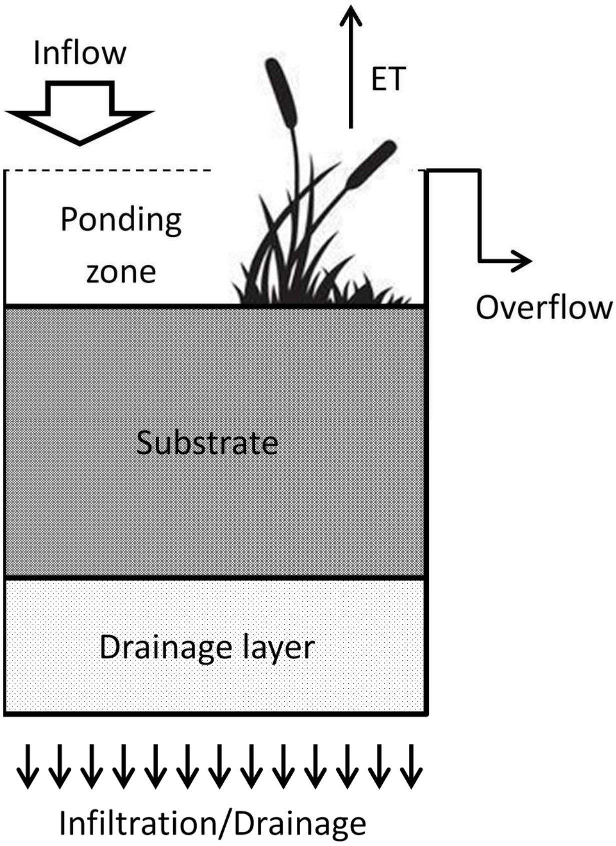

Figure 1. Conceptual biofilter model

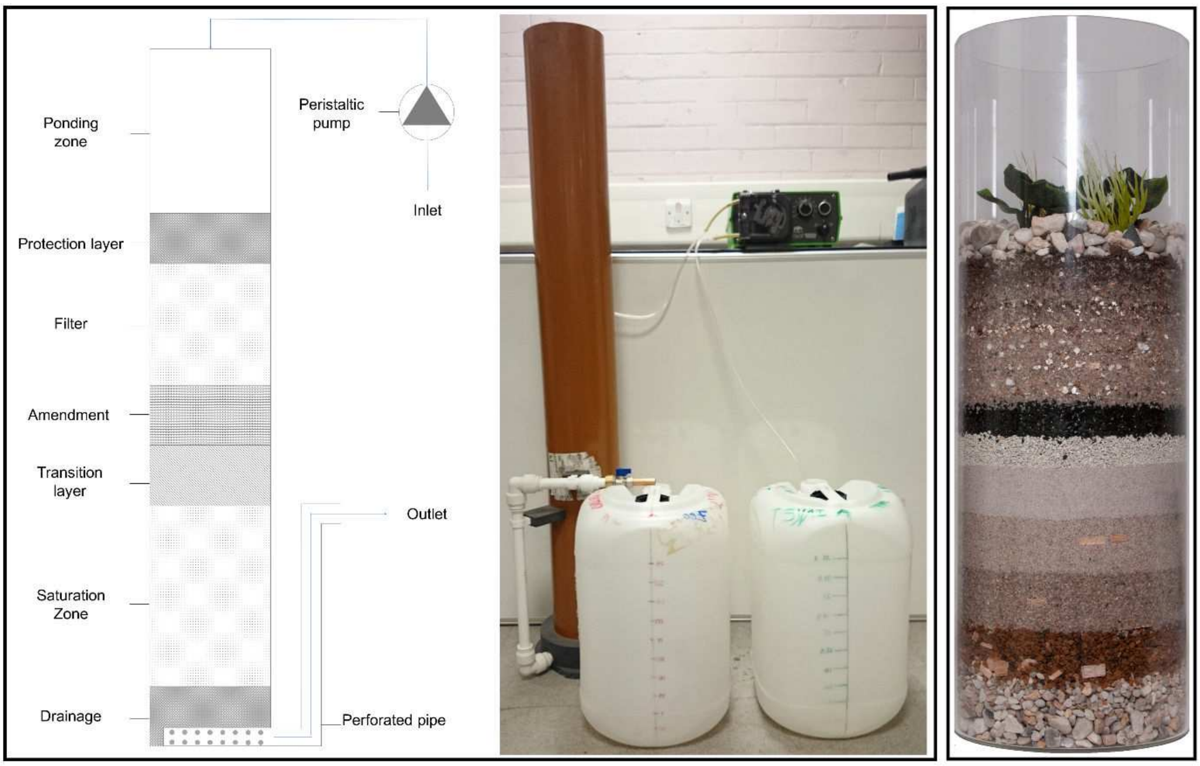

Figure 2. (a) Experimental setup; (b) Scale model of the tested biofilter column

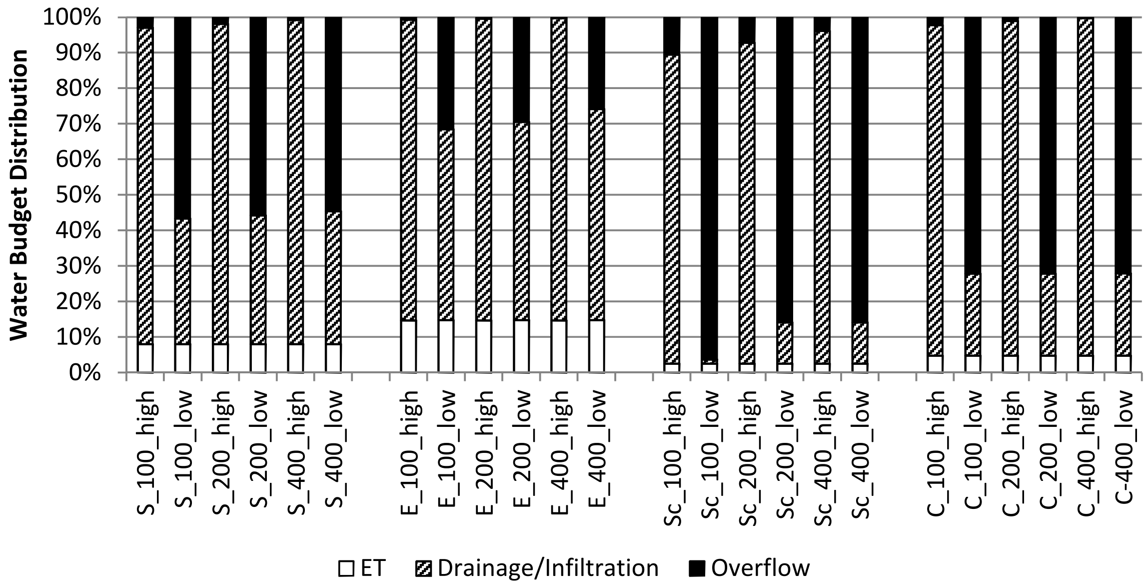

Figure 3. Long-term water budgets associated with a 10 m^2 bioretention device serving an adjacent 90 m^2 drainage area. Labels are in the format: Location_Ponding depth_drainage/infiltration rate. The four locations are: S – Sheffield; E – east Midlands; Sc – NW Scotland; C – Cornwall. Ponding depths are in mm. The two infiltration drainage rates are: high – 7.2 mm/hr; low – 0.36 mm/hr.

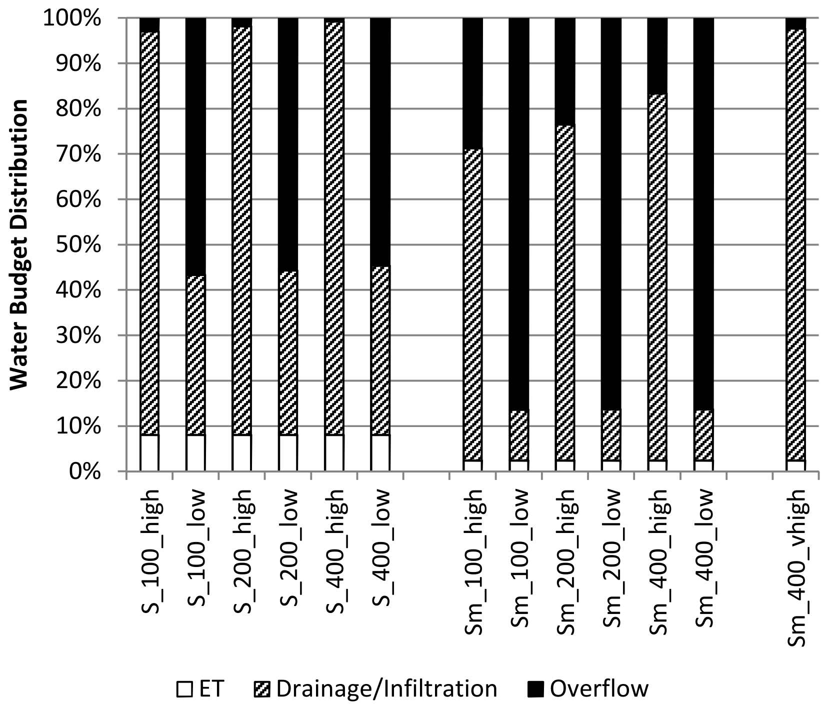

Figure 4. Long-term water budgets associated with a bioretention device. Labels are in the format: Location/ratio_Ponding depth_drainage/infiltration rate. The two location/ratio scenarios are: S – Sheffield 10% and Sm – Sheffield 3%. Ponding depths are in mm. Drainage/infiltration rates are: high – 7.2 mm/hr; low – 0.36 mm/hr; and vhigh – 72 mm/hr.

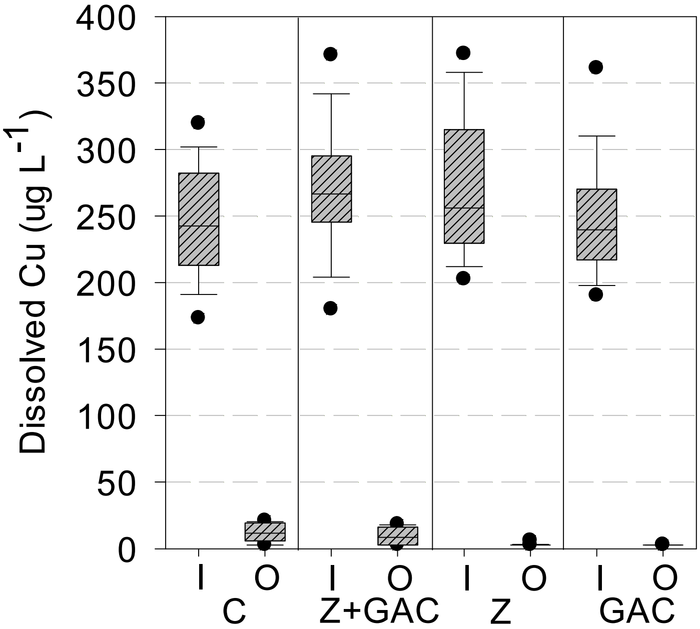

Figure 5. Inflow (I) and outflow (O) EMC of copper for the tested configurations. The box plot summarizes the statistical measures of median, upper and lower quartiles, minimum and maximum values (lines) and outliers (dots)

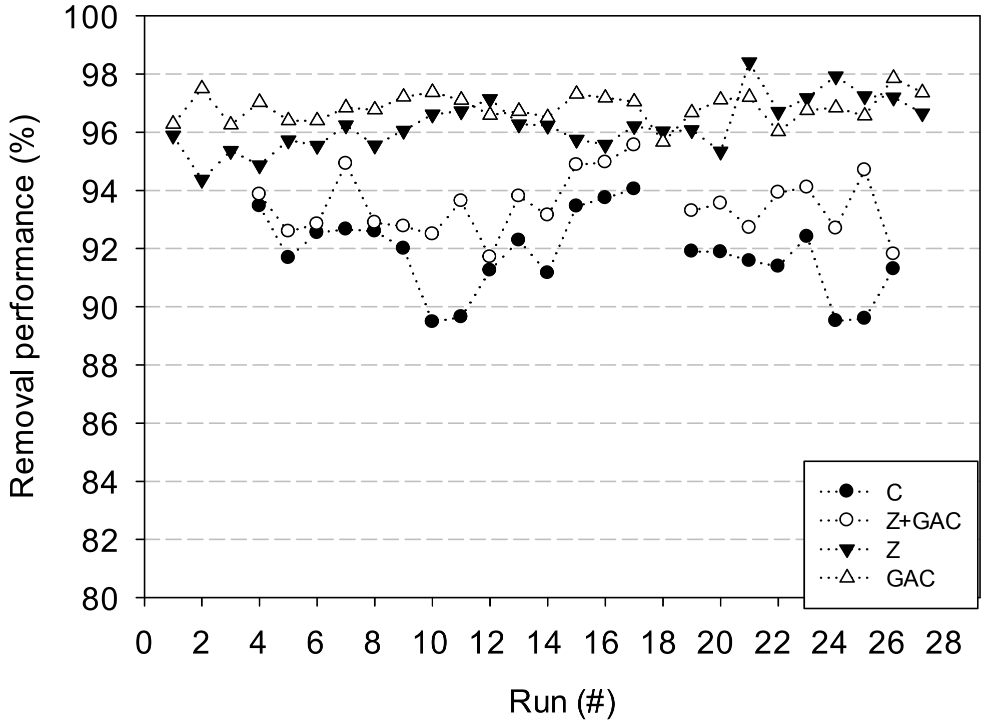

Figure 6. Removal performance (%) for copper for the tested configurations.

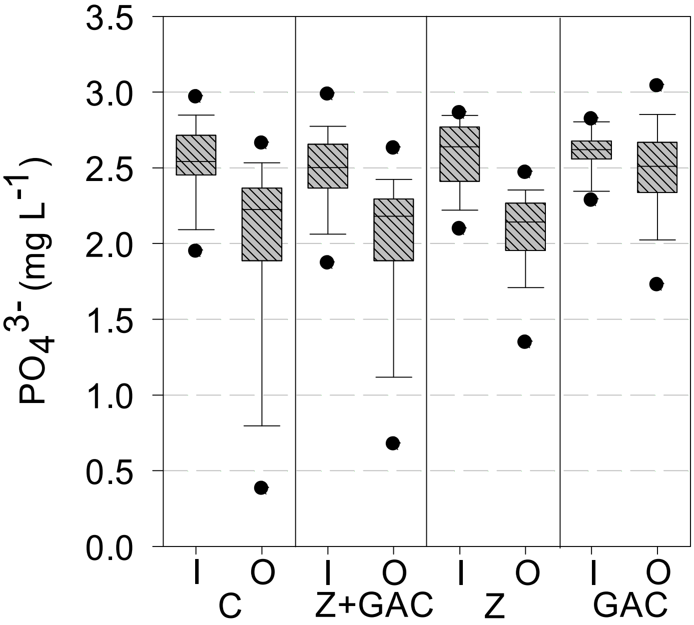

Figure 7. Inflow (I) and outflow (O) dissolved phosphorus EMC values for the tested configurations. The box plot summarizes the statistical measures of median, upper and lower quartiles, minimum and maximum values (lines) and outliers (dots)

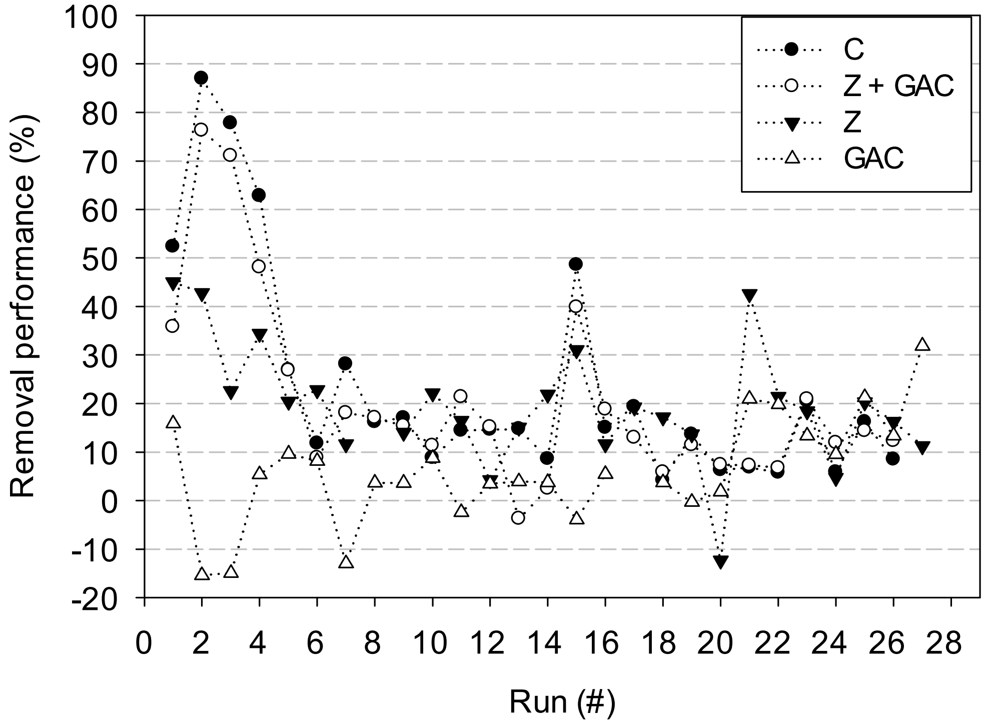

Figure 8. Removal performance (%) for dissolved phosphorus for the tested configurations.

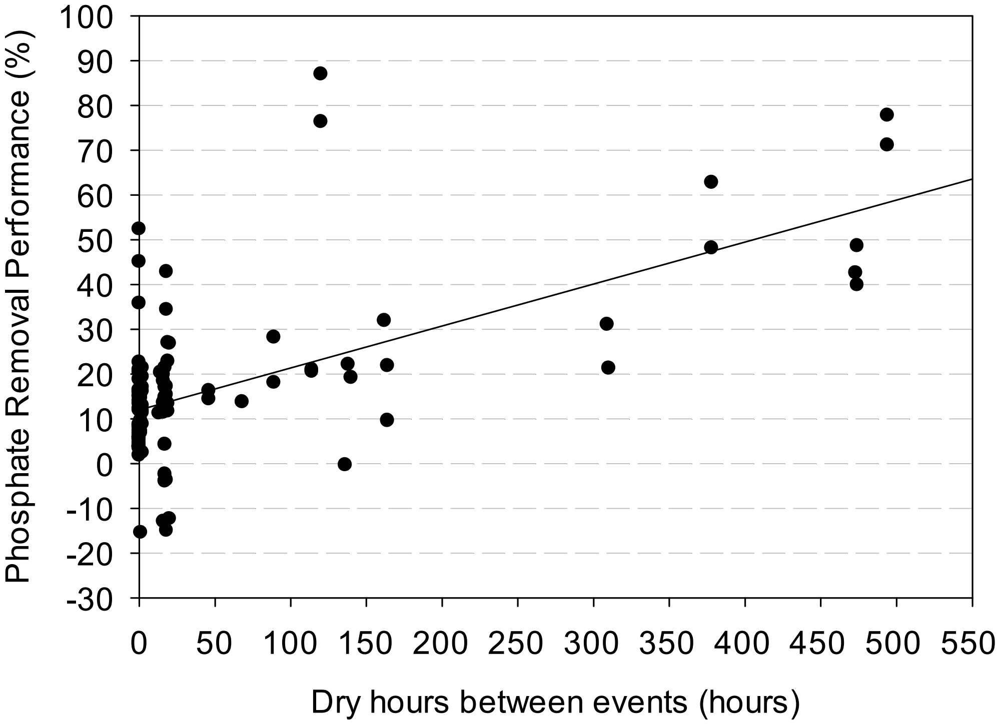

Figure 9. Phosphate removal performance against dry hours between events. A moderate correlation with rs(102)=0.467 and p=0.005 has been found using a non-parametric Spearman test.

Table 1. Synthetic stormwater characteristic (inflow)

Parameter	Median	Min	Max	SD	Source
Cu (µg/L)	250.94	61.86	596.15	73	$CuSO_4$
Zn (µg/L)	331.6	267.03	428.59	41	ZnCl
Cd (µg/L)	0.67	0.45	2.12	0.34	CdCl
PO_4^{3-} (mg/L)	2.59	1.84	3.08	0.23	Tap water
pH	7.18	6.51	7.83	0.23	Tap water
Conductivity (µS/cm)	324	183.7	507	55.33	Tap water
Temperature (˚C)	22.4	18.6	25.5	1.42	Tap water
Dissolved Oxygen (mg/L)	9.39	7.35	10.96	0.59	Tap water

Table 2. Adsorption test for the media used in the column test expressed in terms of adsorption capacity (Qe) and removal performance (RP).

	Qe (mg/g)				RP (%)			
	Phosphate		Copper		Phosphate		Copper	
	AVG	SD	AVG	SD	AVG	SD	AVG	SD
Zeolite	0.044	0.010	31.01	4.27	7.1	1.4	69.9	3.2
GAC	0.055	0.029	39.77	3.79	8.9	4.5	90.0	0.4
Woodchip	-0.071	0.016	7.26	1.72	-11.6	2.9	16.9	5.2
Perlite	0.029	0.033	25.25	2.24	4.9	5.5	54.3	6.3
Vermiculite	0.085	0.016	43.95	1.94	14.0	2.8	94.2	0.1
Top Soil	-0.180	0.030	42.73	2.30	-29.7	4.7	91.6	0.9
Sand	0.012	0.010	4.24	2.25	1.9	1.5	13.3	7.0

Table 3. Outflow EMC values of (a) pH, conductivity, (b) temperature and dissolved oxygen for the four tested configurations

	pH				Cond. (µS/cm)			
Design	Median	Min	Max	SD	Median	Min	Max	SD
C	7.16	6.6	7.5	0.2	273	147	449	105.6
Z + GAC	7.18	6.8	7.6	0.2	267	157	456	102.7
Z	7.32	6.6	7.7	0.2	332	287	414	29.93
GAC	8.06	7	9.3	0.7	316	281	392	19.28
	T (˚C)				DO (mg/L)			
Design	Median	Min	Max	SD	Median	Min	Max	SD
C	22.5	22	24	4.3	8.42	5.41	8.9	0.84
Z + GAC	22.9	20	24	1	8.17	4.96	8.7	0.98
Z	23	21	25	0.9	8.84	6.8	9.4	0.7
GAC	23.6	22	25	0.8	6.76	4.96	8.2	0.87