Contaminant removal and hydraulic conductivity of laboratory rain garden systems for stormwater treatment

J. F. Good, A. D. O'Sullivan, D. Wicke and T. A. Cochrane

ABSTRACT

In order to evaluate the influence of substrate composition on stormwater treatment and hydraulic effectiveness, mesocosm-scale (180 L, 0.17 m²) laboratory rain gardens were established. Saturated (constant head) hydraulic conductivity was determined before and after contaminant (Cu, Zn, Pb and nutrients) removal experiments on three rain garden systems with various proportions of organic topsoil. The system with only topsoil had the lowest saturated hydraulic conductivity (160–164 mm/h) and poorest metal removal efficiency (Cu 69.0% and Zn 71.4%). Systems with sand and a sand–topsoil mix demonstrated good metal removal (Cu up to 83.3%, Zn up to 94.5%, Pb up to 97.3%) with adequate hydraulic conductivity (sand: 800–805 mm/h, sand–topsoil: 290–302 mm/h). Total metal amounts in the effluent were <50% of influent amounts for all experiments, with the exception of Cu removal in the topsoil-only system, which was negligible due to high dissolved fraction. Metal removal was greater when effluent pH was elevated (up to 7.38) provided by the calcareous sand in two of the systems, whereas the topsoil-only system lacked an alkaline source. Organic topsoil, a typical component in rain garden systems, influenced pH, resulting in poorer treatment due to higher dissolved metal fractions.

Key words | hydraulic conductivity, metal speciation, rain garden, treatment efficiency

INTRODUCTION

Urbanisation enhances stormwater runoff, resulting in increased metal, sediment, and nutrient pollutant loads, decreased local infiltration and greater peak flow intensities (Palhegyi 2010). Heavy metal contaminants of concern, primarily Cu, Pb, and Zn, originate from wear-and-tear of vehicle parts including brake linings (e.g. Cu) and tyre fillers (e.g. Zn) as well as additives in oil and petrol (Ward 1990; Davis et al. 2000). These contaminants accumulate on impermeable surfaces and are transported via stormwater networks, often untreated, to local and downstream aquatic ecosystems (Davis et al. 2001; Moore et al. 2005). Different technologies have been used to mitigate stormwater runoff, including traditional drainage networks fitted with concrete proprietary devices (e.g. vortex separators and filters) and large detention systems such as infiltration basins. These are primarily designed to remove suspended solids and reduce flood risk (Wanielista & Yousef 1993).

In 2003, Christchurch (population 369,000), New Zealand, underwent a paradigm shift in urban water management towards implementing ecologically-integrated drainage infrastructure in new and retrofitted urban developments. The Christchurch Councils aim to replace traditional piped structures, which incur inevitable maintenance and offer minimal benefits besides drainage, with natural treatment systems (CCC 2005), classified as Sustainable Urban Drainage Systems (SUDS) or Water Sensitive Urban Designs (WSUD) elsewhere. These natural systems inevitably require periodic maintenance, but are considered to be appreciating systems over time, compared with deprecating piped systems. The Auckland region (population 1.3 million) adopted a similar approach by spending >NZ $5 billion (2005–2014) to replace deteriorating pipe networks with natural low-impact (i.e. rain garden) designs servicing stormwater demands from new developments (Pandey et al. 2005).

Rain gardens are gaining popularity as a bioinfiltrative SUDS (e.g. Fletcher et al. 2004; Dietz & Clausen 2005), but large differences in their design criteria are apparent.
from the limited guidelines available (e.g. New Zealand systems propose a 13 mm/h infiltration rate and >100 cm topsoil (ARC 2003) compared with 13–130 mm/h and >45 cm topsoil recommended in Californian systems in the USA (SFPUC 2009)). SUDS have demonstrated high metal and TSS removal, but results of nutrient removal are variable and complicated by leaching of system substrate (Hunt et al. 2006; Davis et al. 2009). Organic material, a key component in rain garden design, reduces the overall hydraulic throughput but supports vegetation growth and is believed to play an important role in contaminant removal (ARC 2003; Muthanna et al. 2007). However, rain garden design recommendations are not informed by performance data as there is a dearth of information on treatment and hydraulic responses of bioinfiltrative rain garden systems (Fletcher et al. 2004; Dietz & Clausen 2005; Henderson et al. 2007). Furthermore, such data are required for understanding the long-term responses of bioinfiltrative treatment systems and for modelling efforts aiming to predict their mitigation behaviour.

This study quantified the treatment efficacies and hydraulic performance of laboratory mesocosm-scale rain gardens as a function of their substrate makeup. It was hypothesised that greater organic topsoil amounts would improve the treatment capacity due to more opportunities for biotreatment, but compromise the hydraulic throughput compared to sand-only systems.

**METHODOLOGY**

**Experimental design**

Mesocosm-scale (180 L cylindrical, 0.17 m² surface area) rain gardens were established in a laboratory setup (Figure 1). Substrate makeup was different in each of the three systems to investigate the effect(s) of organic topsoil on heavy metal (Zn, Cu and Pb), nitrate removal and hydraulic throughput under simulated rain events. A small (20 mm) layer of bark mulch was applied on top of each system in order to help diffuse stormwater across the column as practised in actual rain garden construction practices. The volume of bark (upper surface) and under-drainage gravel (at exit) remained constant across the three systems. Sand (AP-20, well-graded coarse sand) and topsoil (sandy loam, 5% organic content) volumes were varied for all three systems (system 1: 500 mm sand; system 2: 500 mm topsoil; system 3: 250 mm of both sand and topsoil). A total rain garden depth of 670 mm (with 520 mm of ‘reactive’ substrate) was maintained at the onset of the experiment. Substrate media were analysed prior to all experiments to confirm substrates were not a source of the contaminants of concern (Cu, Zn, Pb, nitrate). Total metal contaminant concentrations were consistent with background soil levels (Landner & Reuther 2004; Moore et al. 2005), and TCLP results indicated that contaminant leaching from virgin substrates was primarily below detection limits.

**Constant head hydraulic conductivity**

All three systems were initially flushed with tap water filtered through a 10 μm inline cartridge. They were then saturated by filling the columns from the bottom upwards to allow entrapped air to escape through the top, ensuring system saturation and reducing the possibility for preferential flow paths to occur. The inflow rate was then adjusted to maintain a constant head over the substrate. Flow rates through the saturated columns were determined using a stopwatch and a 100 mL graduated cylinder (n = 25 measurements per column). A derivation of Darcy’s equation (Equation (1)) was used to calculate the saturated hydraulic conductivity for each system. Hydraulic conductivity tests were later repeated in the same manner following the completion of the contaminant removal
efficiency experiments (see further below).

\[
K_{\text{sat}} = \frac{Q 	imes L}{A \times (L + P)}
\]  

(1)

where \(K_{\text{sat}}\) is the saturated hydraulic conductivity [m/s], \(Q\) is the flow [m³/s], \(L\) is the depth of substrate [m], \(A\) is the cross sectional area [m²], and \(P\) is the water over substrate [m].

### Treatment efficiency

#### Experimental operation

Raw stormwater runoff from a neighbouring Christchurch city catchment (where an operational rain garden is being monitored) was collected during three storm events and kept refrigerated at <4 °C until the experiment commenced. Despite refrigeration, stormwater partitioning will progress towards equilibrium during holding; however, representative samples were collected immediately prior to use in experiments to accurately quantify metal speciation. Metal speciation change was minimum for Cu and Pb (5.2 and 2.2%, respectively) and moderate for Zn (18.6%). A simulated speciation change was minimum for Cu and Pb (5.2 and 6.1%, respectively) and moderate for Zn (18.6%).

#### Sampling and chemical analysis

Water was manually sampled following the contextual Australian and New Zealand Environment and Conservation Council (ANZECC) guidelines (ANZECC 2000). In compliance with these guidelines, at least 10% of the samples were duplicated for quality assurance/quality control (QA/QC) purposes. Samples were collected in high-density polyethylene (HDPE) sampling bottles. Raw stormwater from the influent supply tank was sampled at the onset of each experimental run. Effluent stormwater (post rain garden) was sampled every 5 min for the first 50 min and every 20 min thereafter. Effluent measurements were continuously logged for pH, temperature and nitrate using an YSI Professional Plus water quality meter. Flow measurements were conducted at five minute-intervals throughout experimental runs. [Samples for TSS were also collected; however they were not analysed because of the extended holding times (exceeding the QA/QC procedures per APHA Method 2540) resulting from the major earthquake in Christchurch on 22 February at the time of the experiment, which forced closure of the University for 5 weeks.]

Total (Σ particulate + dissolved) metal samples were preserved with concentrated nitric acid (HNO₃, Fisher, trace analysis grade) to reduce the pH to less than 2.0 (APHA 2005). All metals (Cu, Zn, Pb) were analysed by ICP-MS (Agilent) following Method 3125B (APHA 2005). Total metal samples for digestion were mixed thoroughly on a magnetic stir plate while 25 mL of sample were transferred to a 50 mL polypropylene centrifuge tube. After the addition of 5 mL concentrated HNO₃, tubes were placed in a heating block and samples were boiled for 1 h. Cooled samples were then filtered through an encapsulated 0.45 μm PVDF filter (47 mm, Labserve) directly into the analysis tube and analysed via ICP-MS. Dissolved metal samples were pre-filtered through disposable 0.45 μm filters before HNO₃ acidification. Nitrate-nitrogen (NO₃-N) was analysed according to Method 4500-NO₃, based on the cadmium reduction method (APHA 2005).

### RESULTS AND DISCUSSION

#### Stormwater characterisation

Median total metal concentrations in untreated mixed stormwater collected from the header tank immediately

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prior to experimental runs (Cu = 16.6 μg/L, Zn = 168.0 μg/L, Pb = 35.1 μg/L) greatly exceeded the recommended 90% ANZECC median guidelines by a factor of 9.2 (Cu), 11.2 (Zn) and 6.3 (Pb) (Table 1). First flush concentrations (first 15 min of appreciable runoff collected in the field) are presented to show contrast to homogenised feed tank samples (Table 1). It was anticipated that nitrate concentrations might be of concern in stormwater as reported elsewhere (Taylor et al. 2005; Henderson et al. 2007); however, median nitrate concentrations measured in the stormwater (700.0 μg/L) were less than the 90% ecotoxicological threshold value of 3,400 μg/L (Table 1).

The ANZECC effects-based guidelines are the main thresholds adopted in New Zealand for estimating likely ecotoxicity arising from discharges into surface water bodies (ANZECC 2000). The 90% threshold applies to urban areas, stipulating that at these (total, median) concentrations, 90% of the species are likely to be unaffected. The guidelines are also adjusted for hardness (but hardness levels in the stormwater in this study were low with 20–28 mg/L as CaCO₃, thus no adjustments to the values were required). While these guidelines are not legally-binding, they are typically adopted in consenting processes, so in effect become compliance targets. Rain gardens in Christchurch currently discharge to surface waters due to high water tables and low permeability of local soils. Therefore, discharges are subject to the ANZECC guidelines to control ecotoxicological effects.

**Hydraulic performance**

Saturated hydraulic conductivity, a measure of the infiltrative capacity, was quite different for each of the three different systems. Measurements collected before (initial) and after (final) the treatment experiments were similar, with no apparent indication of system clogging. Hydraulic conductivity ranged from 805 mm/h (initial) in the sand-only system to 160 mm/h (final) in the topsoil-only system (Table 2). These hydraulic conductivities equate to an order of magnitude greater than the minimum allowable

### Table 1

<table>
<thead>
<tr>
<th>Contaminant</th>
<th>Untreated stormwater (μg/L)</th>
<th>90% ANZECC guidance (μg/L)</th>
<th>90% ANZECC exceedance factor</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>First flush</td>
<td>Median</td>
<td>First flush</td>
</tr>
<tr>
<td>Copper</td>
<td>25.0</td>
<td>16.6</td>
<td>1.8</td>
</tr>
<tr>
<td>Zinc</td>
<td>310.0</td>
<td>168.0</td>
<td>15.0</td>
</tr>
<tr>
<td>Lead</td>
<td>87.0</td>
<td>35.1</td>
<td>5.6</td>
</tr>
<tr>
<td>Nitrate</td>
<td>2,540.0</td>
<td>700.0</td>
<td>3,400.0</td>
</tr>
</tbody>
</table>

### Table 2

<table>
<thead>
<tr>
<th>Average values</th>
<th>Inflow</th>
<th>Outflow</th>
</tr>
</thead>
<tbody>
<tr>
<td>Sat. hydraulic cond. (mm/hr)</td>
<td>Sand</td>
<td>Sand/topsoil</td>
</tr>
<tr>
<td>Initial</td>
<td>NA</td>
<td>805 ± 4</td>
</tr>
<tr>
<td>Final</td>
<td>NA</td>
<td>800 ± 5</td>
</tr>
<tr>
<td>pH</td>
<td>6.23 ± 0.08</td>
<td>7.38 ± 0.03</td>
</tr>
<tr>
<td>Cu concentration (μg/L)</td>
<td>17.1 ± 3.6</td>
<td>12.0 ± 11.1</td>
</tr>
<tr>
<td>% dissolved (%)</td>
<td>16</td>
<td>24</td>
</tr>
<tr>
<td>Zn concentration (μg/L)</td>
<td>162.8 ± 28.1</td>
<td>76.6 ± 125.6</td>
</tr>
<tr>
<td>% dissolved (%)</td>
<td>51</td>
<td>30</td>
</tr>
<tr>
<td>Pb concentration (μg/L)</td>
<td>39.8 ± 11.3</td>
<td>11.6 ± 14.8</td>
</tr>
<tr>
<td>% dissolved (%)</td>
<td>4</td>
<td>1</td>
</tr>
<tr>
<td>Nitrate (μg/L)</td>
<td>883 ± 126</td>
<td>3,613 ± 1,602</td>
</tr>
</tbody>
</table>
conductivity of 13 mm/h stipulated in California, Maryland, and New Zealand design guidelines (ARC 2003; MDE 2009; SFPUC 2009). All systems maintained an adequate hydraulic throughput across experimental runs. The supply tank functioned as a pre-treatment sump to reduce the clogging risk, which is typically recommended in stormwater design manuals internationally.

**Treatment efficiency**

Average effluent metal concentrations from each of the different systems at the standard loading rate were lower than the untreated influent levels (Table 2). An exception was for Cu from the topsoil-only system, which had an average effluent concentration of 37.9 ± 34.8 μg/L, compared with the raw influent of 17.1 ± 3.6 μg/L. This higher average outflow concentration is explained by the fact that the average values represent instantaneous contaminant concentrations which increased for Cu at the end of the run when discharge from the system decreased. Other contaminant concentrations exhibited similar trends across experimental runs evidenced by high standard deviation relative to contaminant values (Table 2). Individual (time-step) contaminant loading calculations (a product of concentration and flow) confirmed that copper did not leach from the substrate on any particular occasion.

Total metal loadings are shown in Figure 2 for the entire standard loading experimental run. Removal of total Cu, Zn and Pb was greater than 50% for all systems, except Cu in the topsoil-only system that had a negligible reduction due to a high dissolved fraction. It was originally hypothesised that the topsoil-only system would demonstrate optimal contaminant removal due to greater organic content; however, the converse was observed for Cu and Zn (Table 2, Figure 2). For instance, effluent discharged from the topsoil system contained 4,973 μg Cu/m³ (0.3% removed) and 18,785 μg Zn/m³ (60.5% removed), while effluent from the sand system contained 2,174 μg Cu/m³ (56.4% removed) and 12,589 μg Zn/m³ (73.5% removed). Furthermore, the topsoil-only system had the largest proportion of dissolved metals (80% Cu and 94% Zn) compared with the sand-only system (24% Cu and 30% Zn) or sand–topsoil mix (56% Cu and 54% Zn) (Figure 2), highlighting the effect of topsoil on metal speciation in stormwater (bio)infiltration systems. To investigate this further, regression...
relationships were derived for each of the dissolved metals with effluent pH (Figure 3). It is well reported that pH influences metal speciation (Pitcher et al. 2004; Sansalone & Glenn 2007) and this study demonstrated this principle in stormwater treatment systems where pH is never considered in their design criteria.

The trend lines (Figure 3) give a general indication of the relationship of dissolved metals and pH. The correlation between pH and dissolved metal fraction was highly significant (Pearson’s correlation, $p < 0.0001$, $d_1 = 74$) for Cu, Zn, and Pb. Results indicate that pH (within the range of influent stormwater (6.23) to effluent sand-only system (7.38)) significantly influences metal speciation and, hence, metal removal capacity in biofiltration systems. The effluent pH from the sand and sand/topsoil systems was somewhat buffered to 7.38 and 6.60, respectively, compared with the raw stormwater pH of 6.23 (Table 2). This probably resulted from calcium carbonate in the sand component (Plassard et al. 2000), while the topsoil-only pH was not buffered and remained at 6.24, resulting in poorer metal treatment. The organic topsoil lacked an alkaline medium necessary to elevate pH and promote metals to the particulate species, and organic acids present in the topsoil media likely resulted in lower pH and higher dissolved concentrations (organic-metal complexes) leaching from the systems (Mason et al. 1999). Both the low pH and organic acid-metal interactions explain the higher dissolved fractions in the effluent for Cu and Zn (not as applicable for Pb, which was successfully removed (>80%) in all three systems as lead is prevalent in the particulate state and thus easier to remove in infiltration systems). This phenomenon is problematic for most stormwater treatment systems in New Zealand that are designed to remove 75% of TSS (ARC 2003) on the premise that metals (including Zn and Cu) are concurrently removed (ARC 2003) – an assumption that is currently debated amongst the engineering profession and water quality scientists as metals can prevail in dissolved states (Sansalone & Glenn 2007). Without a pH amendment, rain gardens (and other filtration systems relying on particulate removal) are unlikely to provide adequate total metal removal if the pH is less than neutral in the raw stormwater. Additionally, acidic runoff (pH 3–7) which is not buffered can leach metals from treatment systems back into the environment (Sansalone & Ying 2008).

Removal efficiencies (expressed as a percentage) for each system under standard and high loading rates were also compared (Table 3). Pb removal was very good across all systems (ranging between 81.6% for ‘standard’ loading up to 98.6% for ‘high’ (i.e. double) loading). Zn and Cu were generally removed better in the sand-only system than the topsoil-only system. For instance, during application of the high loading rate 94.5% Zn was removed in the sand system but only 71.4% in the topsoil system, while Cu removal was 83.3% in the sand system and

![Figure 3](https://iwaponline.com/wst/article-pdf/65/12/2154/442150/2154.pdf)

**Figure 3** | Dissolved metal regression trends as a function of stormwater effluent pH. Pearson’s correlation $R$ and $p$ are shown ($n = 76$).

### Table 3 | Total metal removal efficiency during standard (Cu $= 6.0 \pm 0.7 \mu g/min$, Zn $= 57.9 \pm 6.1 \mu g/min$, Pb $= 13.7 \pm 2.8 \mu g/min$) and high (Cu $= 11.4 \pm 3.0 \mu g/min$, Zn $= 91.5 \pm 23.4 \mu g/min$, Pb $= 30.6 \pm 9.1 \mu g/min$) contaminant loading rates ($n = 76$ standard loading, $n = 37$ high loading per contaminant).

<table>
<thead>
<tr>
<th></th>
<th>Standard loading</th>
<th>High loading</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Sand</td>
<td>Sand/topsoil</td>
</tr>
<tr>
<td><strong>Cu</strong></td>
<td>Influent conc. for all systems (µg/L)</td>
<td>17.1 ± 3.6</td>
</tr>
<tr>
<td></td>
<td>Removal efficiency (%)</td>
<td>56.4</td>
</tr>
<tr>
<td><strong>Zn</strong></td>
<td>Influent conc. for all systems (µg/L)</td>
<td>162.8 ± 28.1</td>
</tr>
<tr>
<td></td>
<td>Removal efficiency (%)</td>
<td>73.5</td>
</tr>
<tr>
<td><strong>Pb</strong></td>
<td>Influent conc. for all systems (µg/L)</td>
<td>39.8 ± 11.3</td>
</tr>
<tr>
<td></td>
<td>Removal efficiency (%)</td>
<td>81.6</td>
</tr>
</tbody>
</table>
69.0% in the topsoil system. Overall, removal of all metals was greater at the higher loading rate indicating that the systems have not yet reached treatment capacity.

**Nutrient export**

Nitrate concentrations, likely originating from the topsoil and/or bark that were exported from the systems during experimental runs (3,613, 4,557, and 4,330 μg/L for sand, sand/topsoil, and topsoil, respectively) greatly exceeded the ANZECC guidelines. Alternate wet/dry cycles can affect nitrate leaching from soils (Randall & Mulla 2001) and these rain gardens undergo such cycles due to the sporadic nature of stormwater. Without vegetation, which has been shown to facilitate nutrient retention in stormwater bioinfiltration systems (Read et al. 2008), nitrate-nitrogen is easily mobilised. Nitrogen dynamics in rain garden systems (including systems with vegetation) should be further investigated to more accurately assess nutrient loads. [Although samples were also taken to measure phosphate, they could not be measured during this experiment due to substantial laboratory disruption incurred by the major earthquake in Christchurch on 22 February 2011.]

**Media lifespan**

An assessment of rain garden media lifespan for the optimal sand/topsoil system was calculated by estimating the annual contaminant metal loads to the system (employing the SCS rainfall–runoff curves and catchment characteristics described earlier under experimental operation) as well as laboratory system removal efficiency under the standard loading regime and size of field systems. This resulted in net contaminant retention amounts/m$^3$ substrate media. These concentrations were compared with the Interim Sediment Quality Guidelines (ISQG) produced by ANZECC (2000), which are appropriate for these metals in New Zealand (McCready et al. 2006). Calculations indicated that (assuming pre-treatment for gross sediment removal) media should last a long time prior to requiring replacement. For instance, ISQG-low concentrations of Zn (200 mg/kg), Cu (65 mg/kg) and Pb (50 mg/kg) were not expected to be reached until 56, >100, and 23 years, respectively, indicating that the substrates would not be considered an environmental hazard before that time. In practice, life expectancy of rain gardens is highly dependent on mitigating diffuse sediment input to them. Further research should seek to understand the likely longevity of different media receiving known contaminant amounts.

**CONCLUSIONS**

Elevated metal concentrations in Christchurch stormwater far exceed the contextual ecotoxicological guidelines recommended for healthy freshwater ecosystems. Therefore, appropriate treatment should be implemented to mitigate adverse ecological impact prior to surface discharge. In line with the Christchurch City Council’s approach of implementing more ‘natural’ treatment systems, rain gardens are being adopted but their effectiveness is not well understood as performance data are extremely sparse. This study found that total metal loads were reduced by >50% in all laboratory rain gardens investigated, with the exception of copper in the topsoil-only system that demonstrated a negligible reduction due to its high dissolved fraction. Contrary to previous assumptions (ARC 2003; Muthanna et al. 2007), topsoil is not an optimal substrate to enhance metal or nutrient removal in (bio)infiltrative systems. This was attributed to its inability to buffer the pH of incoming stormwater resulting in higher dissolved metal fractions that are not conducive to being removed through settling like particulate metal phases. Metal removal efficiencies were enhanced at an effluent pH of 7.38 compared with the 6.24 pH provided in raw stormwater. Therefore, pH enhancement afforded by alkaline substrates may help improve metal removal in (bio)infiltrative systems, which are designed on filtration principles. It may also provide sufficient pH enhancement to overcome the limitations of topsoil, which is an important component for sustaining vegetation growth. The experimental systems do not yet appear to have reached treatment capacity as metal removal efficiency was strongly correlated to contaminant loading, and hydraulic conductivities did not decline following multiple treatment experiments. Nitrate export from the systems was observed and will be further investigated. Subsequent experiments are examining the treatment behaviour of similar systems that are amended with alkaline waste products to provide a pH buffering media. Overall, the rain garden and similar bioinfiltration systems require further data to optimise their designs.

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