Efficiencies of multilayer infiltration systems for the removal of urban runoff pollutants
Lizhu Hou, Fang Liu, Chuanping Feng and Li Wan

ABSTRACT
Current rates of urban development will result in water runoff becoming a major complication of urban water pollution. To address the worsening situation regarding water resource shortage and pollution, novel multilayer infiltration systems were designed and their effectiveness for removing pollutants in urban runoff tested experimentally. The multilayer infiltration systems effectively removed most pollutants, including organic matter (chemical oxygen demand (CODCr)), total nitrogen (TN), ammonia-nitrogen (NH4+-N) and total phosphorus (TP). CODCr, TN, NH4+-N, and TP were reduced by 68.67, 23.98, 82.66 and 92.11%, respectively. The main mechanism for nitrogen removal was biological nitrogen removal through nitrification and denitrification. Phosphorus in the urban runoff was removed mainly by fixation processes in the soil, such as adsorption and chemical precipitation. The results indicate that the proposed novel system has potential for removal of pollutants from urban runoff and subsequent reuse of the treated water.

Key words | adsorption, multilayer infiltration systems, urban runoff pollution

INTRODUCTION
To ensure a sustainable urban water cycle, it is necessary to establish a self-regulated city water recycle and reuse system. In particular, subsequent reuse of treated water or groundwater replenishment has been proposed as a tactic for water management, for which infiltration of urban runoff and wastewater is a potential option.

Infrastructure (roads, sidewalks, commercial and residential structures) added during land development and urbanization is designed to collect precipitation and carry it away from the watershed, typically in existing surface water channels, such as streams and rivers. Urban stormwater runoff contains pollutants that can impact the quality of surface, seepage and ground waters (Taebi & Droste 2004; Scholz 2006; Göbel et al. 2007; Vijayaraghavan et al. 2009) and are a major barrier to the reuse of road runoff. One approach to addressing this problem is to capture and treat urban runoff before it reaches the receiving watercourses.

Infiltration facilities have potential as systems for groundwater replenishment, wherein urban runoff can be infiltrated to groundwater (Furumai et al. 2005). On the other hand, the seepage water through such facilities can provide subsequent reuse of urban runoff (Pratt 1999; Hou et al. 2009). Porous pavements are one form of multilayer infiltration system that can be used to mitigate the problem of stormwater runoff. Porous pavement systems commonly consist of a matrix of concrete bricks or a plastic web-type structure, with voids filled with sand, gravel or soil. Because the brick layers are permeable, they allow water to percolate into the sub-base (filled with washed aggregates), where it is either stored for subsequent recycling, contained, and released at a managed rate or directly released to the ground (Pratt 1999; Hou et al. 2009). In China, a SAT (soil aquifer treatment) system is often used for municipal wastewater treatment but rarely used for the treatment of urban runoff. However, the potential role of the SAT system for urban runoff purification in groundwater recharge should be studied. With regard to infiltration to groundwater, the infiltration facilities constructed in Tokyo continue to contribute efficiently to inundation control and groundwater replenishment even after two decades of operation (Furumai et al. 2005). Further, it has been reported that some non-point pollutants can be trapped during infiltration (Dechesne et al. 2004) and that accumulation on surface sediments/soils occurs within or below infiltration facilities (Dechesne et al. 2004). On the other hand, a potential concern related
to groundwater contamination through infiltration facilities has also been pointed out, based on field investigations (Mason et al. 1999).

Such field investigations have disadvantages with regard to evaluation of changes in the removal efficiencies of pollutants in long-term operations. Compared with field research, bench-scale experiments such as soil infiltration column tests are useful for evaluating the removal efficiencies and mobility of pollutants. Using artificial road runoff, Kimura et al. (2005) evaluated a porous polypropylene treatment for highway runoff and attained approximately 50% removal efficiencies for suspended solids and chemical oxygen demand (COD$_{Cr}$). They found that pollutant concentrations in leachates from batch and column tests in the late stage were generally below detection limits and European Community (EC) limits for drinking water.

However, limited research has been carried out on multiple evaluation of pollutant removal using comprehensive chemical analyses. Soil column experiments can be useful for this (Drewes & Fox 1999), so in this study artificial urban runoff was prepared and passed through soil columns. The aim was to estimate the removal efficiencies of pollutants in urban runoff during multilayer infiltration using multiple chemical analyses, and to determine whether the seepage water can be used or whether pollutants possibly reach the groundwater. The other aim of this study was to identify the mechanisms for multilayer infiltration systems and influencing factors for pollutant removal from polluted road runoff.

If a large volume of urban runoff were not treated in a timely fashion, non-point source pollution may be caused by urban runoff pollution, which would flow directly into the watercourse, leading to eutrophication, for example in Dianchi Lake, Kunming City, China. Therefore, COD, phosphates and nitrates and other parameters relating to eutrophication were the focus of this study.

**METHODS**

**Experiment apparatus**

Figure 1 shows the soil columns for a porous pavement system using mixtures designed recently: a conventional porous pavement system, a natural soil infiltration system (a type of SAT) and a soil infiltration system covered with lawn. The glass columns were 50 cm in length, with each square column measuring 20 x 30 cm. All soils were packed to field density at field moisture content. Each column comprised 5 cm of headspace above the surface, different thicknesses of soil, and 5 cm of quartz sand at the bottom. Each flow in the column was generalized into vertical flows.

**Experiment treatments**

The four columns were filled with commercially available porous paving materials that are generally used in Kunming City, Yunnan Provence, China. The treatments are shown in Figure 1:

- **Treatment 1**, porous pavement brick (6 cm) paving with a padding of 9 cm gravel soil, 5 cm of mixed padding #1 of soil, quartz sand, active carbon with a volume ratio of 1:2:0.2, 5 cm of mixed padding #2 of soil, quartz sand,
active carbon, scrap iron and wood sawdust with a volume ratio of 1:2:0.2:0.2:0.2, 15 cm of thick soil, and 5 cm of quartz sand.

- Treatment 2, porous pavement brick (6 cm) paving with a padding of 9 cm of gravel soil, 25 cm of thick soil, and 5 cm of quartz sand.
- Treatment 3, 40 cm of thick soil, 5 cm of quartz sand.
- Treatment 4, 35 cm of thick soil covered with 5 cm of lawn, and 5 cm of quartz sand.

**Soil characterization**

Soil samples (Table 1) were taken from the Milesi Garden site in Xishan District, Kunming City, China. Soil texture (particle size) was analysed using a Malvern laser particle size analyser (MS2000). According to the US system of classification division systems (De Latt 2010), the samples were clay loam.

Porous pavement bricks designed and manufactured by Yatai Company (Beijing, China) were used for the construction of experimental columns. This pavement brick is suitable for car parks, driveways, and moderately trafficked areas. Each brick had a dimension of 100 mm (width) × 200 mm (length) × 60 mm (depth), as shown in Figure 1. The pavement brick had a pressure intensity 44.3 MPa, a porosity of 7.45% and an infiltration coefficient of 0.789 mm s⁻¹. The pavement surfaces were graded to a slope between 1 and 2% (National Development and Reform Commission 2005). The mixture of zeolite, scrap iron and sands was made in the laboratory.

**Influent characterization**

Table 2 lists the primary categories of urban runoff pollutants, pollutants associated with each category, typical concentrations, likely runoff sources, and potential effects. A runoff supply system was prepared in a water tank with 1.5 times the average urban runoff concentration of Kunming conditions (Table 2) (Yang & Zhao 2007; Liu et al. 2009). The runoff influent was obtained by the following method. A pharmaceutical influent was prepared in the

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**Table 1** | Physical properties of soils

<table>
<thead>
<tr>
<th>Sampling layer (cm)</th>
<th>( \rho_d (g \text{ cm}^{-3}) )</th>
<th>TN (g kg⁻¹)</th>
<th>TP (g kg⁻¹)</th>
<th>Organic matter (g kg⁻¹)</th>
<th>( \text{NH}_4^+ - \text{N} ) (mg kg⁻¹)</th>
<th>( \text{NO}_3^- - \text{N} ) (mg kg⁻¹)</th>
<th>( \text{NO}_2^- - \text{N} ) (mg kg⁻¹)</th>
<th>pH</th>
</tr>
</thead>
<tbody>
<tr>
<td>1–10</td>
<td>1.41</td>
<td>0.741</td>
<td>0.454</td>
<td>11.5</td>
<td>0.88</td>
<td>28.68</td>
<td>0.15</td>
<td>7.57</td>
</tr>
<tr>
<td>10–20</td>
<td>1.22</td>
<td>0.513</td>
<td>0.454</td>
<td>5.02</td>
<td>0.28</td>
<td>36.98</td>
<td>0.13</td>
<td>7.58</td>
</tr>
<tr>
<td>20–30</td>
<td>1.52</td>
<td>1.08</td>
<td>0.881</td>
<td>21.8</td>
<td>0.59</td>
<td>36.44</td>
<td>0.14</td>
<td>7.67</td>
</tr>
<tr>
<td>30–40</td>
<td>1.59</td>
<td>1.54</td>
<td>1.07</td>
<td>28.2</td>
<td>1.75</td>
<td>35.66</td>
<td>0.15</td>
<td>7.67</td>
</tr>
<tr>
<td>1–10</td>
<td>1.41</td>
<td>0.741</td>
<td>1.01</td>
<td>12.4</td>
<td>4.749</td>
<td>29.86</td>
<td>0.00</td>
<td>7.75</td>
</tr>
<tr>
<td>10–20</td>
<td>1.22</td>
<td>0.855</td>
<td>0.959</td>
<td>12.9</td>
<td>1.5456</td>
<td>33.91</td>
<td>0.00</td>
<td>7.84</td>
</tr>
<tr>
<td>20–30</td>
<td>1.52</td>
<td>0.684</td>
<td>0.873</td>
<td>11.5</td>
<td>2.426</td>
<td>34.67</td>
<td>0.00</td>
<td>7.93</td>
</tr>
<tr>
<td>30–40</td>
<td>1.59</td>
<td>0.741</td>
<td>0.731</td>
<td>13.7</td>
<td>1.8097</td>
<td>33.18</td>
<td>0.00</td>
<td>8.2</td>
</tr>
</tbody>
</table>

\( \rho_d \): particle density.

**Table 2** | Summary of urban runoff pollutants relevant to this study

<table>
<thead>
<tr>
<th>Category</th>
<th>Parameters</th>
<th>Average concentration (Yang &amp; Zhao 2007; Liu et al. 2009) (mg L⁻¹)</th>
<th>Possible sources</th>
<th>Possible effects</th>
<th>Designed solution concentration (mg L⁻¹)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Organic enrichment</td>
<td>COD(_{Cr})</td>
<td>100</td>
<td>Urban runoff; agricultural runoff; landfills; septic tanks; atmospheric deposition; soil erosion</td>
<td>Dissolved oxygen depletion; odours; toxicity levels for fish and other aquatic life</td>
<td>150</td>
</tr>
<tr>
<td>Nutrients</td>
<td>TN</td>
<td>25</td>
<td>Urban runoff; agricultural runoff; fertilizers; landfills; septic tanks; atmospheric deposition; soil erosion</td>
<td>Surface waters: algal overgrowth and blooms; ammonia toxicity</td>
<td>37.5</td>
</tr>
<tr>
<td></td>
<td>( \text{NH}_4^+ - \text{N} )</td>
<td>15</td>
<td></td>
<td></td>
<td>22.5</td>
</tr>
<tr>
<td></td>
<td>( \text{NO}_3^- - \text{N} )</td>
<td>1.15</td>
<td></td>
<td></td>
<td>2.25</td>
</tr>
<tr>
<td></td>
<td>TP</td>
<td>2</td>
<td></td>
<td></td>
<td>3</td>
</tr>
</tbody>
</table>
laboratory. Six litres of solution containing 1 g of dextrose, 0.64 g of ammonium sulphate, 0.1 g of potassium nitrate, and 0.08 g of potassium dihydrogen phosphate was used in each column for every experimental cycle.

During the course of the experiment, the mean entering-water temperatures were from 12.6 to 28.8 °C; the mean effluent temperatures were from 11.5 to 29.5 °C. Mean pH values remained constant over the course of the experiment and can be described as slightly alkaline for all treatments.

Experimental process

Tests were conducted to mimic pollutant removal by using multilayer infiltration systems receiving urban road runoff (Figure 1). The wastewater was pumped to the surface of different multilayer infiltration systems by a water pump. When the depth of the water layer exceeded 1–2 cm, a relief valve was opened to maintain a depth of 1–2 cm. The hydraulic conductivities of treatments 1, 2, 3 and 4 were 0.037755, 0.002599, 0.002736 and 0.000159 cm s\(^{-1}\), respectively, as measured according to the method of De Latt (2010). In the late phase of the experiments, the corresponding hydraulic conductivity for treatments 1, 2 and 3 remained relatively stable, with high values throughout the experiment.

In the formal running stage, columns were operated for 26 cycles (short-term SAT) (Drewes & Fox 1999). Inflow and outflow samples were taken at regular intervals and subsequently analysed directly after sampling in the Groundwater Circulation Remediation Experimental and Training Center of the China University of Geosciences (Beijing).

All liquid samples were analysed for COD\(_{\text{cr}}\), total nitrogen (TN), ammonia-nitrogen (NH\(_4^+\)-N), NO\(_3^−\)-N, NO\(_2^−\)-N and total phosphorus (TP). The COD\(_{\text{cr}}\) of water was determined by the potassium dichromate method (State Environmental Protection Administration of China 2002). All nutrients were analysed using traditional chemical methods (State Environmental Protection Administration of China 2002). Nitrate was reduced to nitrite by hydrazine in alkaline solution and determined at multi-wavelengths by a colorimetric method using a HACH DR 5000 flow injection analyser according to the manufacturer’s instructions. Blue-coloured ammonia compounds were measured at 420 nm wavelength. Salicylate, dichloroisocyanuric acid, sodium chloride and nitroprusside (catalyst) were also used. TP was measured with the ammonium molybdate method, using ammonia molybdate, ascorbic acid, and sulphuric acid. Ammonium molybdate and sodium bicarbonate were also used in the analyses performed by automated colorimetry for all water samples at 700 nm.

The experimental study ran from August to November 2010. A comparison of the results is given in Table 3.

RESULTS AND DISCUSSION

COD removal

While a variety of removal mechanisms, including sedimentation, filtration, precipitation, volatilization, adsorption and plant uptake, are well documented (Kadlec & Knight 1996), recognition that removal of most pollutants in multilayer infiltration systems is due primarily to microbial activity has been a cornerstone of the technology almost from the beginning (Hou et al. 2009). Organic matter contained in urban runoff entering the multilayer infiltration system must be filtered, adsorbed, and then retained for oxidative degradation by microorganisms.

COD\(_{\text{cr}}\) removal efficiencies are illustrated in Figure 2. With regard to treatment 1, the average removal efficiency of COD\(_{\text{cr}}\) was about 24.93 ± 6.62%, and the COD\(_{\text{cr}}\) concentration exceeded 100 mg L\(^{-1}\), which satisfies the national quality standard of China (GB 3838-2002; Chinese National Environmental Bureau 2002) for inferior class V surface water (see Table 4). This may be explained by the thickness of the mixed padding layer of column (treatments) 1 not being thick enough and the hydraulic retention time (HRT) values being lower. In comparison, the corresponding values were 82.48 ± 5.57, 82.04 ± 6.94 and 85.22 ± 7.48% for treatments 2, 3 and 4, respectively. The final effluent COD\(_{\text{cr}}\) from treatments 2, 3 and 4 remained relatively stable below 15 mg L\(^{-1}\), with variations in influent COD\(_{\text{cr}}\). The COD\(_{\text{cr}}\) concentration in the effluent decreased through the experimental process. COD\(_{\text{cr}}\) effluent concentrations of treatments 2, 3 and 4 were reduced to less than 15, 5.0 and 5.0 mg L\(^{-1}\) (Figure 2), respectively. These values all satisfy the national quality standard of China (GB 3838-2002) for class I surface water.

The average HRT values of columns (treatments) 1, 2, 3 and 4 were 6, 72, and 82 min and 41 h for the whole experimental phase, respectively. It is apparent from Figure 2 that the COD\(_{\text{cr}}\) removal efficiencies of treatment 1 were the lowest. This can be explained by the soil void in column 1 being larger, and therefore the seepage discharge velocity would be higher and the HRT shorter, so the degradation process would be inadequate. The COD\(_{\text{cr}}\) removal efficiencies of treatments 2 and 3 were higher than in treatment 1,
because their soil columns had a thicker soil layer, relatively more microbes and longer HRTs throughout the whole experimental phase. Therefore, the reaction was more complete, and the retention, adsorption and degradation of organic matter in soil columns 2 and 3 were obvious. The COD$_{Cr}$ removal efficiencies of treatment 4 were the highest of the four soil columns. This may be explained by the fact that the column was prone to compression and blockage in the late experimental stages and therefore the water flow velocity was the lowest of the four columns, giving the highest HRT. At the same time, the absorption of organic matter by the roots of the lawn reduced the organic content of urban runoff. Therefore, the COD$_{Cr}$ removal efficiencies of different treatments decreased, in order, for columns 4, 2, 3 and 1. The long HRT resulted in high COD$_{Cr}$ removal efficiencies.

Figure 2 gives the COD$_{Cr}$ levels of a typical effluent that had penetrated through the four columns under different influent concentrations. It can be seen from Figure 2 that COD$_{Cr}$ removal for each treatment was lower during the initial stage, owing to the impact of the organic matter being higher (see Table 1). It is apparent that influent concentrations affected the effluent concentrations. Effluent followed the same variation trends as the influent, but at steady state influent conditions the COD$_{Cr}$ levels of the effluent were also stable.

**Nitrogen removal**

During the initial phase, the removal of TN by the multilayer infiltration system was through ammonia adsorption. With further operation of the system, the microorganisms reproduced and gradually grew, so soil organic matter decreased. Therefore, the growth of heterotrophic microbes was limited, and nitrifying microorganisms gradually increased.

Table 3 illustrates nitrogen removal due to biogeochemical degradation and nutrient transformation processes, such as nitrification and denitrification (Scholz 2006) that occur predominantly within the wet, lower sub-base of multilayer infiltration systems.

The average TN removal efficiencies of treatments 1, 2, 3 and 4 were $18.72 \pm 3.59$, $13.70 \pm 2.80$, $22.64 \pm 8.57$ and $40.84 \pm 5.95\%$, respectively. Also, the corresponding TN effluent concentrations all exceeded 2 mg L$^{-1}$, which satisfies the national quality standard of China (GB 3838-2002)

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**Table 3** Average concentrations and removal efficiencies for COD$_{Cr}$, TN, NH$_4$-N, NO$_3$-N, NO$_2$-N and TP between August 2010 and November 2010 ($n = 26$, 95% confidence)

<table>
<thead>
<tr>
<th>Parameter</th>
<th>COD</th>
<th>TN</th>
<th>NH$_4$-N</th>
<th>NO$_3$-N</th>
<th>NO$_2$-N</th>
<th>TP</th>
</tr>
</thead>
<tbody>
<tr>
<td>COD$_{Cr}$ removal for each treatment (%)</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Treatment 1</td>
<td>151.23 ± 8.62</td>
<td>23.81 ± 2.48</td>
<td>24.04 ± 2.44</td>
<td>4.39 ± 0.16</td>
<td>0.0066 ± 0.0058</td>
<td>3.25 ± 0.03</td>
</tr>
<tr>
<td>Treatment 2</td>
<td>147.6 ± 13.04</td>
<td>24.17 ± 4.96</td>
<td>25.33 ± 0.97</td>
<td>4.57 ± 0.21</td>
<td>0.0062 ± 0.0055</td>
<td>3.35 ± 0.08</td>
</tr>
<tr>
<td>Treatment 3</td>
<td>147.14 ± 5.31</td>
<td>24.74 ± 1.80</td>
<td>25.12 ± 0.99</td>
<td>4.35 ± 0.19</td>
<td>0.0081 ± 0.0105</td>
<td>1.52 ± 0.02</td>
</tr>
<tr>
<td>Treatment 4</td>
<td>150.12 ± 12.96</td>
<td>21.53 ± 3.52</td>
<td>24.37 ± 0.95</td>
<td>4.45 ± 0.19</td>
<td>0.0093 ± 0.0053</td>
<td>3.26 ± 0.11</td>
</tr>
</tbody>
</table>

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**Figure 2**: COD$_{Cr}$ levels of a typical effluent that had penetrated through the four columns under different influent concentrations.
for inferior class V surface water. The nitrogen removal efficiencies of treatment 1 are relatively obvious in this process and can be explained by the fact that there was a 5 cm mixed layer in column 1, which contained iron and provided anaerobic conditions to the microbial reductive environment (Jin & Roden 2014). However, the average negative TN removal efficiencies, −27.5, −61.5 and −23.6%, of treatments 2, 3 and 4 during their initial 10 cycles can be explained by the TN content being larger in the clay loam (Table 1).

The NH$_4^+$-N removal efficiencies can be seen in Table 3. The NH$_4^+$-N removal efficiencies from treatments 3 and 4 were much higher, with effluent concentrations under 1.0 mg L$^{-1}$ in the late stages, which satisfies the national

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**Table 4 | Water quality standards for various pollutants**

<table>
<thead>
<tr>
<th>Parameters/Standard value</th>
<th>I</th>
<th>II</th>
<th>III</th>
<th>IV</th>
<th>V</th>
</tr>
</thead>
<tbody>
<tr>
<td>pH $\leq$</td>
<td>6–9</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>COD$_c$ (mg L$^{-1}$) $\leq$</td>
<td>15</td>
<td>15</td>
<td>20</td>
<td>30</td>
<td>40</td>
</tr>
<tr>
<td>TP (mg L$^{-1}$) $\leq$</td>
<td>0.02</td>
<td>0.1</td>
<td>0.2</td>
<td>0.3</td>
<td>0.4</td>
</tr>
<tr>
<td>TN (mg L$^{-1}$) $\leq$</td>
<td>0.2</td>
<td>0.5</td>
<td>1.0</td>
<td>1.5</td>
<td>2.0</td>
</tr>
<tr>
<td>NH$_4^+$-N (mg L$^{-1}$) $\leq$</td>
<td>0.15</td>
<td>0.5</td>
<td>1.0</td>
<td>1.5</td>
<td>2.0</td>
</tr>
<tr>
<td>NO$_3^-$-N (mg L$^{-1}$) $\leq$</td>
<td>–</td>
<td>–</td>
<td>–</td>
<td>–</td>
<td>–</td>
</tr>
<tr>
<td>NO$_2^-$-N (mg L$^{-1}$) $\leq$</td>
<td>–</td>
<td>–</td>
<td>–</td>
<td>–</td>
<td>–</td>
</tr>
</tbody>
</table>
quality standard of China (GB 3838-2002) for class III surface water. These processes may be due to the soils in columns 3 and 4 having longer HRTs and the soil in column 4 being covered with a layer of lawn at the top of the column, where the absorption of NH$_4^+$-N by plant roots would play a role to some degree. Therefore, the microbial responses in columns 3 and 4 were sufficient (Table 3) and for NH$_4^+$-N the removal efficiencies were, in order, columns 4, 3, 2 and 1. The NH$_4^+$-N removal from multilayer systems mainly occurred through NH$_4^+$-N adsorption and microbial degradation. The other reason for the efficient NH$_4^+$-N removal was the longer HRT, which resulted in sufficient reaction. Changes in the removal efficiency showed the same trends as the influent. Therefore, the multilayer infiltration systems possessed some buffering capacity.

The concentrations of NO$_3^-$-N and NO$_2^-$-N in the effluent increased with increasing influent NO$_3^-$-N loading. NO$_3^-$-N in the experiment comes from two sources, one from the nitrification reaction and the other from influent water, which is not directly involved in the reaction. The negative nitrate removal efficiency can be explained by a relatively high transformation rate from ammonia to nitrate and a relatively low denitrification rate (Scholz 2006). The multilayer infiltration system had no statistical results on the removal efficiencies for NO$_3^-$-N, so no relationships could be identified between the NO$_3^-$-N removal efficiency and multilayer infiltration. Of the four treatments, the NO$_3^-$-N effluent concentration in treatment 1 was the lowest, which can be explained by the fact that it comprised a mixture containing iron, which provided electrons to the denitrification process.

The NO$_2^-$-N effluent concentration was higher than the influent concentration, especially with respect to treatment 2 during its late stage, which indicates that the denitrification process was affected, resulting in NO$_2^-$-N accumulation. This can be explained by a lack of anaerobic conditions, or limited anaerobic space. A further reason is that shorter HRTs lead to lower NO$_3^-$-N and NO$_2^-$-N production. During the degradation process, a large amount of NO$_3^-$-N and NO$_2^-$-N was produced, but the above denitrification process was blocked. This may be explained by the compaction of material in the soil column, the decrease in porosity and the decrease in hydraulic conductivities during the treatment phase of the experiment. The NO$_2^-$-N effluent concentrations in treatment 1 were the lowest of the four treatments. This can be explained by the fact that there was only a 5 cm thick mixed layer containing iron in column 1, which provided a carbon source and a reductive environment.

**TP removal**

Phosphorus fixation is a two-step process: sorption onto the soil solids and conversion of the adsorbed phosphorus into mineraloids or minerals. If the sorption sites are filled with either phosphate anions or other ions, phosphorus sorption will be low and the sorption of phosphorus per unit of percolation liquid decreases with each year of recharge (Pitt et al. 1999).

Table 3 illustrates the total phosphorus removal efficiencies. It can be seen from Table 3 that the average TP removal efficiencies of treatments 1, 2, 3 and 4 were about 79.89 ± 3.43, 94.12 ± 1.46, 96.95 ± 0.71 and 97.46 ± 0.28%, respectively. The TP effluent concentrations of treatment 1 exceeded 0.4 mg L$^{-1}$, which satisfies the national quality standard of China (GB 3838-2002) for inferior class V surface water. The TP effluent concentrations of treatments 2, 3 and 4 were reduced to below 0.4, 0.1 and 0.1 mg L$^{-1}$, respectively, which satisfies the national quality standards of China (GB 3838-2002) for classes V, II and II surface water, respectively. With respect to TP, the corresponding removal was, in order, columns 3, 4, 2 and 1. The average content of P in the soil layer of treatment 3 was 0.715 g kg$^{-1}$ before the experiment; however, the average P content in the soil layer of treatment 3 was only 0.893 g kg$^{-1}$ after the experiment phase.

After repeating the experiment several times, the changes in the curves were reduced and the removal effects became stable. This was due to the higher phosphorus content in the soil (Table 1) and unstable phosphorus adsorption by the soil at the beginning of the experiment. With the repeated experimental cycles, the reactions became essentially stable, as the phosphate-accumulating bacteria gradually acclimated.

**CONCLUSIONS**

A range of infiltration media (soils) were selected and their effectiveness for pollutant removal from urban runoff was compared. Seepage discharge from each experimental column was analysed, and the contaminant removal mechanism was studied in depth.

COD$_{Cr}$ and TP in urban runoff were effectively removed by the multilayer infiltration systems using multiple chemical processes. NO$_3^-$ and NO$_2^-$, which were produced during nitrification by bioactivation, were not sufficiently removed.

Biological nitrification and denitrification were the main removal mechanisms for nitrogen. Phosphorus in urban
runoff was mainly removed by fixation processes in the soil, such as adsorption and chemical precipitation.

The pollutant removal effects of treatments 2, 3 and 4, being common forms of multilayer systems, were as expected. However, the pollutant efficiency of treatment 1, designed in this work, was not as expected and may be explained by the mixed padding layers not being thick enough. Hence, the mixed padding layer should be thicker in future trials.

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