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Stormwater Sediment and Bioturbation Influences on Hydraulic Functioning, Biogeochemical Processes, and Pollutant Dynamics in Laboratory Infiltration Systems

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Stormwater sediments that accumulate at the surface of infiltration basins reduce infiltration efficiencies by physical clogging and produce anoxification in the subsurface. The present study aimed to quantify the influence of stormwater sediment origin (urban vs industrial catchments) and the occurrence of bioturbators (tubificid worms) on the hydraulic functioning, aerobic/anaerobic processes, and pollutant dynamics in stormwater infiltration systems. In laboratory sediment columns, effects of stormwater sediments and tubificids were examined on hydraulic conductivity, microbial processes, and pollutant releases. Significant differences in physical (particle size distribution) and chemical characteristics between the two stormwater sediments led to distinct effects of these sediments on hydraulic and biogeochemical processes. Bioturbation by tubificid worms could increase the hydraulic conductivity in stormwater infiltration columns, but this effect depended on the characteristics of the stormwater sediments. Bioturbation-driven increases in hydraulic conductivity stimulated aerobic microbial processes and enhanced vertical fluxes of pollutants in the sediment layer. Our results showed that control of hydraulic functioning by stormwater sediment characteristics and/or biological activities (such as bioturbation) determined the dynamics of organic matter and pollutants in stormwater infiltration devices.

Introduction

In many countries, environmental protection and waste management are priorities for maintaining the quality of water resources. Wastewater is usually treated using water treatment plants, and stormwater management usually consists of various BMPs (best management practices) such as retention and infiltration systems used for collection, infiltration, and transport of stormwater to groundwater (*1*). Such BMPs are important for controlling the stormwater runoff volume in urban areas (*2*). However, stormwater runoff on sealed surfaces can lead to contamination of aquatic systems with soluble and particulate nutrients, metals, and hydrocarbons (*3, 4*). These contaminants are concentrated on stormwater sediment deposits which can affect the biogeochemical processes and the hydraulic functioning of infiltration basins (*5, 6*). The clogging of infiltration systems by stormwater sediments may limit the pollutant contamination of receiving waters (*7, 8*). However, it also reduces the infiltration of the stormwater in groundwaters (*9*) increasing the risk of flooding during heavy rain events.

Characteristics and pollutant loads of stormwaters depend in part on the area drained by the stormwater system (*4*). Different drained areas (i.e., industrial, urban) may thus induce deposition of different stormwater sediments which may affect hydraulic functioning and contaminant retention in infiltration devices differently. Stormwater sediments also harbor high densities of invertebrates adapted to polluted environments such as tubificid worms (*6*). Since bioturbation activities of tubificids can reduce the sediment clogging (*10*) and increase nutrient and pollutant releases from sediments to water (*11–13*), the presence of tubificid worms may increase infiltration system permeability and the contamination risk for groundwater resources.

The aim of the present study was to quantify the effects of stormwater sediments from urban and industrial systems and bioturbation activities of tubificid worms on the hydraulic functioning, microbial processes, and dynamics of nutrients and pollutants in stormwater infiltration systems. Laboratory experiments using infiltration sediment columns were performed to determine how changes in permeability of stormwater sediments affect biogeochemical processes and pollutant releases in the interstitial water. We expected that effects of polluted stormwater sediments on experimental column functioning (hydraulic conductivity, microbial processes, pollutant dynamics) would be driven by the characteristics (particle size distributions, organic matter content, pollutant content) of stormwater sediments. We also expected that a bioturbation-driven increase in hydraulic conductivity would favor aerobic microbial processes in the columns impacted by stormwater sediments. However, the effects of invertebrate bioturbation on hydraulic conductivity would depend on the physical (particle size distribution) and chemical characteristics (nutrient and pollutant contents) of stormwater sediments.

Materials and Methods

Sediment Infiltration Columns and Sediment Preparation. Experiments took place in 12 sediment infiltration columns (25 cm in height and inside diameter of 7 cm) which were filled with sand and gravel onto which stormwater sediment was deposited. Sand and gravel were used to simulate a fluvioglacial deposit on which several infiltration basins have been installed in the French Rhône Alpes Region (*14*). Gravel (grain size of 5–8 mm) and sand (grain size of 100–1000 μm) for the columns were collected from the Rhône River near Lyon, France. The gravel and sand were cleaned with deionized water and dried at 60 °C. Before the columns were filled, 18 kg of dry sand was mixed with 90 g of fibrous cellulose powder (0.5% of the sediment weight), 10 L of artificial water (prepared after (*10*)), and an extract of natural bacteria to stimulate microbial growth. Two different types of stormwater (STORM) sediments were used in our experiment: STORM-DR sediment collected from the "Django Reinhardt" infiltration basin in the industrial zone of Chassieu, France (*15*), and STORM-IUT sediment collected from the infiltration basin on the Campus of Lyon I University, Lyon, France (*5*).

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The stormwater sediments were sieved through a 1 mm mesh aperture net to remove macrofauna and coarse particles.

Experimental Design. The experiment was designed as a factorial two-way ANOVA with equal cell sizes. Main factors were stormwater sediment type (STORM-DR and STORM-IUT) and bioturbation influence (control with no tubificid and added tubificids), with 3 columns per cell (i.e., $n = 3$ per treatment). On day −12 (12 days before the tubificid addition), each column was filled with the sand and gravel mixture to a height of 18 cm to simulate the heterogeneous and porous granulometry of infiltration basins. One day later, each column was supplied with aerated artificial water with a constant pressure head ($\Delta H = 2.5$ cm). Use of well-oxygenated (7.8 ± 0.5 mg L^{-1} dissolved oxygen) artificial water allowed a constant water quality at the inlet of infiltration columns throughout the experiment with heavy metal and hydrocarbon concentrations below the quantification limits (<4.6 μg L^{-1} and 10 ng L^{-1}, respectively), low concentrations (<10 μg L^{-1}) of $N\text{-}NH_4^+$ and $P\text{-}PO_4^{3-}$, and constant concentrations of $N\text{-}NO_3^-$ (2.6 ± 0.1 mg L^{-1}) and dissolved organic carbon (0.5 ± 0.1 mg L^{-1}). On day −8, a layer of 2 cm of stormwater sediments (about 250 g) was deposited at the sediment surface of the 12 columns; 6 columns had STORM-IUT sediment added and 6 columns had STORM-DR sediment added. Eight days after stormwater sediment addition (day 0), 50 tubificid worms (*Tubifex tubifex*) were introduced in 3 of the columns from each stormwater sediment treatment. Tubificids were collected from the infiltration basin of Django Reinhardt and acclimated to experimental conditions (particle size and temperature) in the laboratory for more than 15 days before use in infiltration columns. The tubificid density used (about 13 000 individuals m^{-2}) is typical for organic sediments (*16*). Experiments were performed at constant water temperature (15 ± 0.5 °C) and light (12 h light, 12 h dark cycle).

Sediment, Microbial, And Chemical Analyses. *Characteristics of Stormwater Sediments.* Prior to the column experiments, concentrations of particulate organic carbon (POC), particulate nitrogen (PN), particulate phosphorus (PP), and particle size distributions were measured in stormwater sediments. Analyses of heavy metals (Pb, Zn, Cu, Cd) and polycyclic aromatic hydrocarbons (PAHs) in stormwater sediments were determined before and at the end of the columns experiments. PN, PP, metal, and PAH analyses were performed by the Health and Environmental Laboratory of Lyon following standard methods (*17*). Particle sizes were determined by a laser diffraction granulometer (Mastersizer 2000, Malvern Instrument, UK).

Vertical Distribution of Tubificids. The living invertebrates were recovered at the end of the experiment to estimate mortality and vertical distribution in each treatment. The top 10 cm of sediments were sliced (the top 3 cm in slices of 0.5 cm thickness and the lower 7 cm in slices of 1 cm thickness), sieved through a 500 μm mesh and preserved in alcohol (96%) before counting. Tubificid mortality was low; 92.2 ± 5.8% and 97.2 ± 3.7% of the tubificids added on day 0 were recovered at the end of the experiment in columns with STORM-DR and STORM-IUT, respectively. The abundance of living tubificids in each sediment slice were expressed as percentages of the total tubificids found in each column.

Hydraulic Conductivity. Water flow rates were measured on day −10 (before stormwater sediment and tubificid additions), on days −7 and 0 (after stormwater sediment addition and before tubificid addition), and on days 6 and 12 (after stormwater sediment and tubificid additions) at the outlet of each column to quantify the effects of stormwater sediments and bioturbation on sediment clogging. We determined the hydraulic conductivity (K) of the sediment in the infiltration columns using Darcy's formula: $q = K(\Delta H)/z$, where q is the specific discharge (cm h^{-1}), K is the hydraulic conductivity (cm h^{-1}), ΔH is the hydraulic head (cm), and z is the thickness of the sediment section (cm). In our experiment, ΔH was 2.5 cm and z was 18 cm (at day −10, before the addition of the 2 cm layer of stormwater sediment) or 20 cm (at days −7, 0, 6, and 12, after the stormwater sediment addition).

Physicochemical Analyses. Dissolved oxygen (DO), nitrate ($N\text{-}NO_3^-$), ammonium ($N\text{-}NH_4^+$), orthophosphate ($P\text{-}PO_4^{3-}$), and dissolved organic carbon (DOC) concentrations were measured in water at days −10, −7, 0, 2, 6, and 12 to assess biogeochemical processes in sediment columns. We measured DO concentrations using microsensors (UNISENSE, Denmark) fitted to the tubes at the inlet or the outlet of each column. Water samples were collected in the overlying water and at the outlet of each column for chemical analyses. These samples were filtered through Whatman GF/F filters and stored at 4 °C until analysis. Analyses of $N\text{-}NO_3^-$, $N\text{-}NH_4^+$, and PO_4^{3-} in water were made with an automatic analyzer (Easychem Plus, Systea, Italy) using standard colorimetric methods (*18*). DOC concentration was measured with an Analytik Jena total carbon analyzer using high temperature combustion after removing inorganic C with hydrochloric acid and CO_2 stripping under O_2 flow. The fluxes of DO, $N\text{-}NO_3^-$, $N\text{-}NH_4^+$, $P\text{-}PO_4^{3-}$, and DOC (mg d^{-1} m^{-2} of infiltrating surface) in the columns were calculated using changes in the concentration of each species between the inlet and the outlet of the columns reported to measured water flow rates.

Microbial Analyses. At the end of the experiment, 10 g of wet sediment were collected from the stormwater sediment surface (0−1 cm) and in the gravel/sand substrate (15−18 cm) to assess the effects of stormwater sediments and bioturbation on bacterial abundances and hydrolytic activity. The DNA intercalating dye (DAPI) and a Cy3-probe (EUB 338, eubacteria) were used to determine total bacteria abundance (bacteria g^{-1} of sediment dry weight (DW)), and the percentage of active eubacteria (*13*). Hydrolytic activity was measured using fluorescein diacetate (FDA) following Mermillod-Blondin et al. (*13*). Results were expressed as μmoles of hydrolyzed FDA h^{-1} g^{-1} sediment DW.

Pollutant Releases. Dissolved metal and hydrocarbon concentrations were measured to quantify the effects of stormwater sediments and bioturbation on pollutant releases from sediment. Concentrations of heavy metals (Pb, Zn, Cu, and Cd) and of PAHs commonly found in stormwater sediments (*5*) were measured in water at the outlet of the columns from day 0 to day 3 of the experiment. Analyses of dissolved metals and PAHs were performed by the Health and Environmental Laboratory of Lyon following standard methods (*17*). The fluxes of pollutants (mg d^{-1} m^{-2} of infiltrating surface) in the columns were calculated using changes in the concentration of each metal and hydrocarbon between the inlet and the outlet of the columns reported to measured water flow rates.

Data Analysis. Differences in nutrient and pollutant concentrations in the two stormwater sediments (STORM-DR and STORM-IUT) were tested by one-way analysis of variance (ANOVA). Hydraulic conductivity and physicochemical variables were tested among columns on day −10 (before stormwater sediment and tubificid additions) and on days −7 and 0 (after stormwater sediment addition and before tubificid addition) using a two-way ANOVA with sediment and tubificid treatments as main effects. After tubificid addition, a two-way repeated measures ANOVA (RM-ANOVA) was used to compare hydraulic conductivity and physicochemical variables among sediment and tubificid treatments, using time as the repeated factor (days 6 and 12). For bacterial measurements and pollutant releases from stormwater sediments to water, we tested the effects of

TABLE 1. Characteristics of the Two Stormwater Deposits (mean ± SD, $n = 3$)[a]

	STORM-DR	STORM-IUT
Nutrients (g kg^{-1} sed. DW)		
particulate organic carbon	198 ± 8.1^{A}	56.2 ± 1.94^{B}
particulate nitrogen	5.67 ± 0.26^{A}	2.24 ± 0.09^{B}
particulate phosphorus	3.53 ± 0.06^{A}	2.74 ± 0.22^{B}
Pollutant Contents (mg kg^{-1} sed. DW)		
Metals		
Cd	19.2 ± 0.4^{A}	3.15 ± 0.05^{B}
Cu	258 ± 4.2^{A}	152.3 ± 12.5^{B}
Ld	533 ± 15^{A}	216.8 ± 17.6^{B}
Zn	1073 ± 14^{A}	489.6 ± 59.2^{B}
Polycyclic Aromatic Hydrocarbons		
fluoranthene	1.22 ± 0.02^{A}	1.59 ± 0.39^{A}
benzo[*b*]fluoranthene	1.01 ± 0.04^{A}	0.78 ± 0.10^{B}
benzo[*k*]fluoranthene	0.42 ± 0.02^{A}	0.34 ± 0.06^{A}
benzo[*a*]pyrene	0.70 ± 0.02^{A}	0.68 ± 0.10^{A}
benzo[*ghi*]perylene	0.95 ± 0.04^{A}	0.57 ± 0.02^{B}
indeno[1,2,3-*cd*]pyrene	0.66 ± 0.03^{A}	0.49 ± 0.02^{B}
chrysene	0.88 ± 0.03^{A}	0.94 ± 0.20^{A}
pyrene	1.45 ± 0.05^{A}	1.57 ± 0.46^{A}
phenanthrene	0.54 ± 0.02^{A}	1.10 ± 0.37^{A}
benzo[*a*]anthracene	0.49 ± 0.01^{A}	0.70 ± 0.19^{A}
anthracene	<QL	0.23 ± 0.06
acenaphthene	<QL	0.22 ± 0.05
dibenzo[*a*,*b*]anthracene	<QL	0.16 ± 0.04
fluorene	<QL	0.20 ± 0.02
naphthalene	<QL	0.15 ± 0.02
2-methyl naphthalene	<QL	<QL
2-methyl fluoranthene	<QL	<QL
Sediment Grain Sizes (% of volume)		
0−10 μm	40.1 ± 3.9^{A}	13.4 ± 0.5^{B}
10−100 μm	52.5 ± 1.1^{A}	34.5 ± 4.1^{B}
100−200 μm	3.0 ± 0.4^{A}	13.5 ± 1.5^{B}
200−500 μm	2.3 ± 0.4^{A}	29.3 ± 2.9^{B}
500−1000 μm	2.1 ± 1.2^{A}	9.3 ± 0.4^{B}

[a] Different symbols (i.e., A, B) revealed significant difference among stormwater sediment treatments (Tukey post hoc test, $p < 0.05$). Note: QL, quantification limit: $QL_{Cd} = 0.2$, $QL_{Cu} = 10$, $QL_{Pb} = 10$, $QL_{Zn} = 30$, and $QL_{PAHs} = 0.13$.

sediment (STORM-DR vs STORM-IUT) and tubificid (control vs tubificids) treatments using a two-way ANOVA. Sediment and depth effects on vertical distribution of tubificids at the end of experiment were tested using a two-way ANOVA.

Tukey post hoc tests were used in cases of significant ANOVA results to determine which treatments differed. When necessary, data were log-transformed and all percentages were arcsine-transformed before statistical analysis to meet the assumption of homoscedasticity and normality. Statistical analyses were performed using Statistica 6_{TM} (Statsoft, Tulsa, OK).

Results and Discussion

Stormwater Sediment Characteristics. The high concentrations of nutrients, metals, and organic pollutants measured in the two studied stormwater sediments (Table 1) were in accordance with studies conducted in infiltration basins located in industrialized and urbanized areas (*3, 5, 15, 19*). No significant differences in nutrient and pollutant concentrations in stormwater sediments were detected between the start and the end of the experiment for all treatments (data not shown). There were significant differences in physicochemical characteristics of the two stormwater sediments (one-way ANOVAs, sediment effect, $p < 0.01$, Table 1). STORM-DR sediment had a higher proportion of very fine sediment particles than that of STORM-IUT sediment (92.5% vs 47.9% of particles smaller than 100 μm, respectively, Table 1). Concentrations of nutrients (POC, PN, and PP), metals (Cd, Cu, Pb, and Zn), and several PAHs (benzo[*b*]-fluoranthene, benzo[*ghi*]perylene, indeno[1,2,3-*cd*]pyrene) were significantly higher in STORM-DR sediment (i.e., ×3.4 and ×6 for POC and Cd, respectively) than in STORM-IUT sediment (Table 1). However, PAHs such as anthracene, acenaphthene, dibenzo[*a*,*b*]anthracene, fluorene, and naphthalene were only detected in the STORM-IUT sediment. The differences in physicochemical characteristics of the two stormwater sediments were probably due to differences in their catchments (industrial vs urban area *4, 20*). Industrial activities account for a large proportion of the nutrient (i.e., mainly N and P), metal, and hydrocarbon loads in stormwaters in industrial catchment (*4*). In urban catchment, nutrient releases come from wastewater, metals from gasoline residues (Pb), roofs, and outdoor furniture (Zn, Cu), and PAHs from road traffic and construction (*4, 19*). Higher proportions of fine sediments and higher contaminant concentrations in the STORM-DR sediment are probably due to the combination of great industrial activities in an urbanized environment (75% of imperviousness) and larger surface catchment (195 vs 2.5 ha) than for the STORM-IUT sediment (urban area, 90% of imperviousness).

Influences of the Two Stormwater Sediments on Hydraulic Conductivity and Chemical Conditions in Infiltration Column. Before stormwater sediment additions (day −10), no significant differences were detected in hydraulic conductivity and physicochemical variables between the groups of columns assigned to each treatment (Figures 1 and 2, two-way ANOVAs, tubificid and sediment effects, $p > 0.05$). The addition of the sediments from infiltration basins sharply reduced the hydraulic conductivity in the columns (days −7 and 0 in Figure 1, two-way ANOVAs, sediment effect, $p < 0.01$). This reduction of hydraulic conductivity was associated with reductions of DO and N-NO_3^- concentrations and increased concentrations of N-NH_4^+, DOC, and P-PO_4^{3-} (days −7 and 0 in Figure 2, two-way ANOVAs, sediment effect, $p < 0.05$). Such results indicate a shift from aerobic to anaerobic conditions; NO_3^- decreases associated with the denitrification process and releases of NH_4^+, DOC and PO_4^{3-} linked with an anaerobic degradation of organic matter (*21*). Hydraulic conductivities were more reduced in the presence of STORM-DR sediment (e.g., 0.17 ± 0.05 cm h^{-1} in controls on day −7) than with STORM-IUT (e.g., 1.91 ± 0.41 cm h^{-1} in controls on day −7) (Tukey post hoc tests, $p < 0.01$). As a consequence, addition of STORM-DR sediment induced lower concentrations of DO and N-NO_3^- and higher concentrations of N-NH_4^+ (×12), DOC (×18), and P-PO_4^{3-} (×10) than measured concentrations with the addition of STORM-IUT sediment (Tukey post hoc tests, $p < 0.05$). These results support our hypothesis that the effects of stormwater sediment deposition in infiltration columns depend on sediment characteristics. The higher percentage of fine particles (<100 μm) in the STORM-DR sediment than in the STORM-IUT sediment (i.e., 92.5% vs 47.9%, respectively) can explain that STORM-DR sediment produced a higher reduction of hydraulic conductivity than STORM-IUT sediment. Changes in hydraulic conductivity determine the water infiltration from the surface to the interstitial water in experimental columns and act in concert with microbial activities to control aerobic−anaerobic conditions in sediments (*22, 23*). The largest reduction in hydraulic conductivity due to STORM-DR sediment addition was associated with the largest shift from aerobic to anaerobic conditions in interstitial water, indicating that biogeochemical processes were strongly influenced by the clogging potential of the stormwater sediments. Microbial analyses showed higher hydrolytic activities (×8) and bacterial abundances (×3) in STORM-DR sediment which had higher OM content than STORM-IUT sediment (Table S1, three-way ANOVA, sediment effect, $p < 0.05$). Thus, aerobic−anaerobic conditions in infiltration columns were also influenced by sedimentary

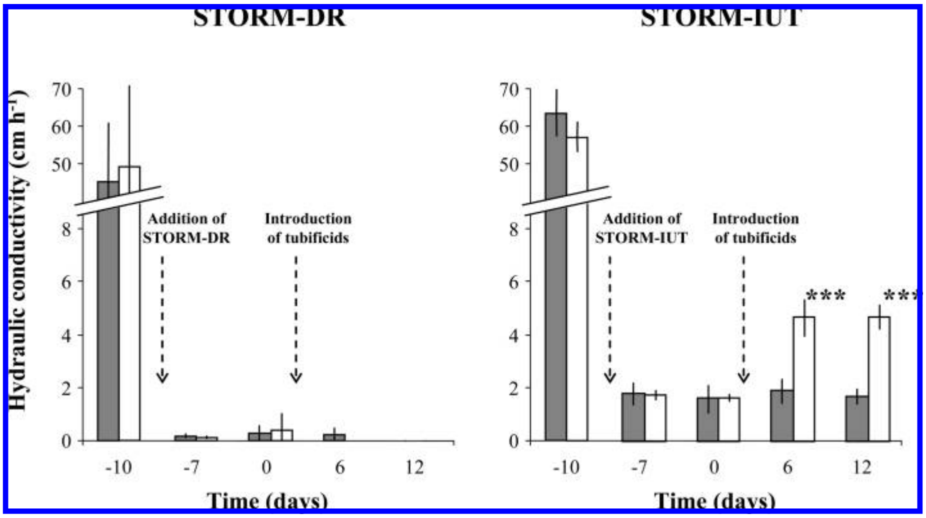

FIGURE 1. Hydraulic conductivity in control (■, gray square) and tubificid (□) columns for the STORM-DR and STORM-IUT sediment treatments (mean ± SD, $n = 3$). For a given sediment type and time, Tukey post hoc tests between control and tubificid treatments revealed significant differences at $p < 0.001$ (***).

TABLE 2. Vertical Fluxes of Oxygen, Nutrients, and Pollutants Measured between the Inlet and the Outlet of the Columns (mean ± SD, $n = 3$)[a]

	STORM-DR			STORM-IUT		
	controls	tubificids	Tukey tests	controls	tubificids	Tukey tests
Oxygen and Nutrient Fluxes(mg d^{-1} m^{-2})						
O_2 consumption	101 ± 0.3	26 ± 0.5	***	433 ± 41	649 ± 53	**
nitrate consumption	4.4 ± 1.2	0.6 ± 0.4	**	8.9 ± 0.5	0.4 ± 1.4	***
ammonium release	12 ± 5.1	3.2 ± 4.4	n.s	1.8 ± 0.2	1.1 ± 0.5	n.s
phosphate release	7.2 ± 3.4	1.5 ± 1.4	n.s	1.5 ± 0.3	1.9 ± 0.2	n.s
DOC release	169 ± 90	73 ± 103	n.s	42 ± 3.4	70 ± 14	*
Metal Releases (μg d^{-1} m^{-2})						
Ld	139 ± 39	99 ± 27	n.s.	–	–	–
Zn	769 ± 222	718 ± 274	n.s.	1634 ± 730	1052 ± 301	n.s.
PAH Releases (ng d^{-1} m^{-2})						
phenanthrene	83 ± 17	95 ± 18	n.s	–	–	–
acenaphthene	219 ± 15	126 ± 45	*	594 ± 50	1512 ± 151	***
naphthalene	435 ± 29	205 ± 107	*	2476 ± 387	5588 ± 803	**
2-methyl naphthalene	175 ± 86	89 ± 24	n.s	759 ± 29	1709 ± 317	**

[a] For a given sediment type, Tukey post hoc tests between control and tubificid treatments revealed significant differences at $p < 0.05$ (*), $p < 0.01$ (**), and $p < 0.001$ (***) or nonsignificant differences (n.s).

organic matter which controlled microbial activities in sediments (*21, 24*).

Influences of the Two Stormwater Sediments on Biogeochemical and Pollutant Fluxes. Despite greater decreases in DO and NO_3^- concentrations measured at the outlet of infiltration columns with STORM-DR sediment (Figure 2), rates of O_2 and NO_3^- consumption (negative fluxes) were significantly lower in STORM-DR sediment than in STORM-IUT sediment (Table 2, one-way ANOVAs, sediment effect, $p < 0.005$). This result indicates that hydraulic conductivity which partly controls the fluxes of electron acceptors (here, O_2 and NO_3^-) from the sediment surface to the sediment determines the ability of the sediment column to degrade organic matter. This positive relationship between hydraulic conductivity and organic matter processing also has been reported in river sediments (*25–27*).

Fluxes of N-NH_4^+ and P-PO_4^{3-} were significantly higher in columns with STORM-DR sediment than with STORM-IUT sediment (Table 2, one-way ANOVAs, sediment effect, $p < 0.005$). Accumulation of compounds such as NH_4^+ in interstitial water can be measured when POM degradation occurred under anaerobic conditions (*21*). Orthophosphate binding to sediment is mainly controlled by oxygen concentrations (*28*). Therefore, the presence of OM-rich surface sediments that reduced water infiltration and dissolved oxygen concentration in underlying porous media was likely to produce more favorable conditions for N-NH_4^+ and P-PO_4^{3-} release processes than surface sediments that did not reduce hydraulic conductivity.

Zinc and three PAHs (acenaphtene, naphthalene, and 2-methylnaphthalene) were detected at the outlet of all columns (Figure S1, Supporting Information). Lead and phenanthrene were also measured in columns containing STORM-DR sediment. Concentrations of dissolved metals and PAHs in all columns were below 200 μg L^{-1} and 100 ng L^{-1}, respectively, because most pollutants remained bound to the sediment particles in stormwater sediments, limiting their release in interstitial water (*13, 21, 29*). As reported for DO and NO_3^- fluxes, zinc fluxes were higher in columns with STORM-IUT than with STORM-DR sediments (Table 2) while higher Zn concentrations were measured at the outlet of control columns with STORM-DR than at control columns with STORM-IUT (Figure S1). Hydraulic conductivity of infiltration columns (with STORM-IUT sediment), which modulates the water flux rates and the time of contact between water and contaminated stormwater sediments,

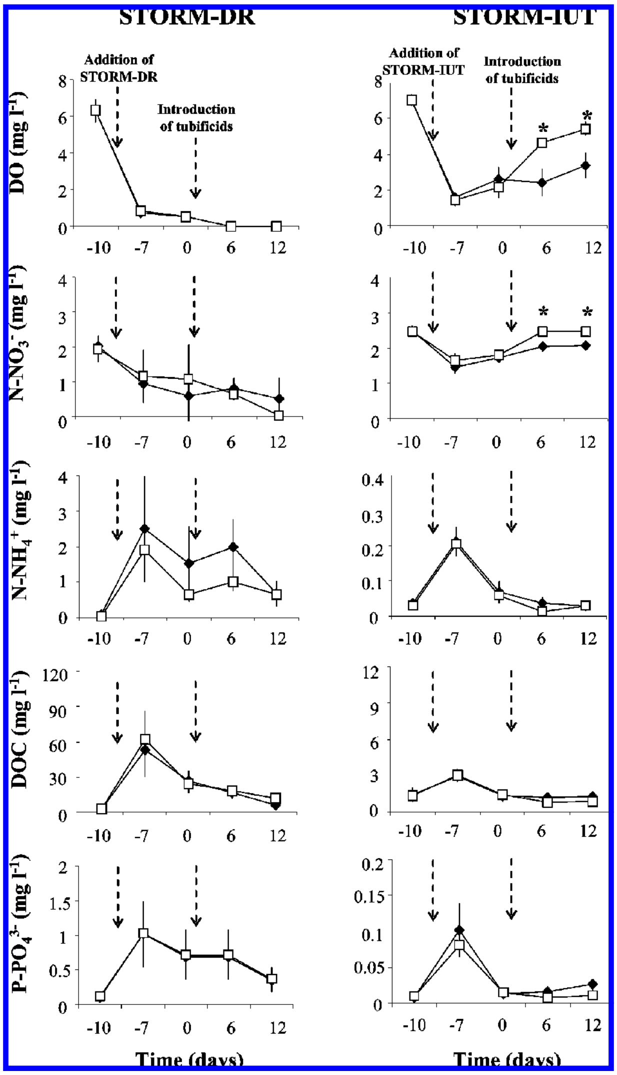

FIGURE 2. Physicochemical analyses in control (◆) and tubificid (□) columns for the STORM-DR and STORM-IUT sediment treatments (mean ± SD, $n = 3$). DO, N-NO_3^-, and DOC concentrations were 7.8 ± 0.5, 2.6 ± 0.1, and 0.5 ± 0.1 mg L^{-1}, respectively, at the inlet of the columns throughout the experiment (mean ± SD, $n = 12$). Concentrations of N-NH_4^+ and P-PO_4^{3-} were below 10 μg L^{-1} at the inlet of the columns throughout the experiment. For a given sediment type and time, Tukey post hoc tests between control and tubificid treatments revealed significant differences at $p < 0.05$ (*).

appeared to be a predominant factor on contamination potential of infiltrating water.

Influence of Tubificids in Stormwater Infiltration Columns. *Influence of Tubificids on Physical Habitat and Hydraulic Functioning.* Bioturbation activities of tubificid worms created dense gallery networks and sediment transport by ingestion of particles at depth and egestion of fecal pellets at the sediment surface (upward conveying (*16*)). Vertical distributions of tubificid worms were significantly different in the two sediment treatments (Figure S2, Supporting Information, two-way ANOVA, "depth × sediment" interaction effect, $p < 0.001$), as tubificid worms were found and burrowed deeper in columns with STORM-IUT than with STORM-DR sediment (Figure S3, Supporting Information). Peaks of tubificid abundance (mean 43.6 ± 19.4%) were at 1.5−2 cm depth in columns with STORM-IUT sediment and at 0.5−1 cm depth (mean 37.5 ± 7.3%) in columns with STORM-DR sediment (Figure S2). Hydraulic conductivity was not affected similarly by the presence of tubificids in columns with STORM-IUT and STORM-DR sediments (days 6 and 12 in Figure 1, two-way RM ANOVA, "sediment × tubificid" interaction, $p < 0.001$). Hydraulic conductivities were significantly increased by worm bioturbation (2.5 times higher than controls) in columns with STORM-IUT sediment whereas worms had no significant influence on hydraulic conductivities in columns with STORM-DR sediment. The characteristics of the stormwater sediments had a great influence on worm burrowing. Although differences in burrowing depth of tubificid worms could be linked to differences in sediment contamination of the sediments (see Figure 7 in ref *30*), we suggest that the fine texture of the STORM-DR sediment affected the vertical distribution of tubificid worms in columns (Figure S3). A recent investigation (*13*) showed that the fine texture of STORM-DR sediments produced a strong compaction of the interface with the fluvioglacial soil (composed of sand and gravel) which restricted tubificid worms to the STORM-DR sediment layer. As a consequence, tubificids did not move through the 2 cm layer of clogged STORM-DR sediments and did not affect hydraulic conductivity. In contrast, tubificids could create galleries throughout the layer clogged by STORM-IUT sediment and produced flowpaths which increased the infiltration efficiencies of the column (Figure S3).

Influence of Tubificids on Nutrient and Pollutant Fluxes. Tubificids had different influences on chemical conditions in columns with STORM-IUT and STORM-DR sediments (Figure 2, two-way RM ANOVAs, "sediment × tubificid" interaction, $p < 0.05$). Enhancement of hydraulic conductivity by tubificid worms increased DO and NO_3^- concentrations in infiltration columns with STORM-IUT sediment. Lack of tubificid influence on hydraulic conductivity in STORM-DR sediment columns was associated with a lack of tubificid-driven changes in physicochemical conditions. The higher aerobic conditions induced by tubificid worms with STORM-IUT sediment favored aerobic processes, which enhanced the consumption of oxygen in infiltration columns by 50% (Table 2). Such oxygenated sediments increased the percentage of active eubacteria and DOC processing and reduced anaerobic processes such as nitrate consumption by denitrification (Tables 2 and S1). We detected significant differences in DO and NO_3^- fluxes between the control and tubificid treatments in columns with STORM-DR sediment (Table 2) which were slightly (but not significantly) higher hydraulic conductivities measured on day 6 without tubificids (Figure 1).

The presence of tubificid worms did not affect pollutant concentrations at the outlets of columns with either STORM-DR or STORM-IUT sediments (Figure S1) nor the concentrations of pollutants measured on stormwater sediments at the end of the experiment (data not shown). However, higher water fluxes due to the increase in hydraulic conductivity by the presence of tubificids in STORM-IUT sediment led to an increase of acenaphthene, naphthalene, and 2-methyl naphthalene fluxes (Table 2). With STORM-DR sediments, the slightly lower hydraulic conductivities measured with tubificids on day 6 produced the opposite result, as lower flux rates of acenaphthene and naphthalene were measured with tubificids than in controls (Table 2). These results further support our prediction that nutrient and pollutant fluxes are strongly controlled by hydraulic conductivities in stormwater sediments. Moreover, our study is the first to demonstrate that a bioturbation-driven modification of hydraulic conductivity in infiltration systems (with STORM-IUT sediment deposit) may affect whole-system functioning (e.g., aerobic and anaerobic processes, pollutant fluxes).

Implications for the Management of Stormwater Infiltration Basins. In the present study, we simulated stormwater sediment deposits on a fluvioglacial soil typical of the infiltration basins of the French Rhône Alpes Region. Soil characteristics are thus specific for our case and use of different underlying soils might have modified the transfer of pollutants and polluted particles through the sediment columns. However, our experimental design clearly showed that the characteristics of accumulated stormwater sediments and the biological activities of microorganisms and invertebrates need to be considered in the management practices of stormwater infiltration systems. Stormwater sediments act as a sink for many nutrients, metals, and hydrocarbons that can be released depending on physicochemical conditions (e.g., pH (*31*), thickness of the accumulated layer of stormwater sediments (*13*)) and biological-mediated processes (*32, 33*). To reduce negative effects of stormwater sediments, Hatt et al. (*8*) suggested removing the 2−5 cm of the clogged layer every two years and using vegetation to limit sediment compaction in filtration systems. Although such a procedure might be effective in reducing clogging, our results indicated that a reduction of the clogging process (by plants or bioturbating fauna) would be associated with an increased transfer of pollutants and nutrients to ground waters. Pitt et al. (*4*) recommended a pretreatment of stormwaters before their infiltration using sedimentation processes (e.g., grass filters, wet detention ponds) to limit groundwater contamination and prolong the life of the infiltration devices. Such procedures associated with bioremediation processes by plants (*34, 35*) and bioturbating (*29*) fauna may be the best ways to protect groundwater systems from contamination.

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Supporting Information Available

Results concerning the microbial analyses, concentrations of pollutants, vertical distribution of tubificids in the sediment, and pictures of the water sediment interface are available in Table S1 and Figures S1, S2, and S3, respectively. This material is available free of charge via the Internet at http://pubs.acs.org.

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