LONGEVITY OF BIORETENTION DEPTHS FOR PREVENTING ACUTE TOXICITY FROM URBAN STORMWATER RUNOFF

Project Final Report

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Related Information

- Stormwater Action Monitoring projects aim to improve stormwater management. Learn about current and past studies at the SAM website[1].
- The Stormwater Work Group (SWG) oversees the SAM Program. Visit the SWG website.[2] to learn more.

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[1] www.ecology.wa.gov/sam

[2] https://ecology.wa.gov/about-us/accountability-transparency/partnerships-committees/stormwater-work-group

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Executive Summary

Urbanization poses increasing threats to aquatic ecosystems including increased chemical loading. The impacts of urbanization on biological integrity are especially evident in the lowland, urban streams of western North America, where adult coho salmon (*Oncorhynchus kisutch*) returning to spawn in the fall have been prematurely dying at high rates. Previous studies have demonstrated the effectiveness of bioretention treatment systems in treating urban stormwater runoff, thereby reducing chemical loading into surface waters, and preventing acutely lethal and sublethal effects to aquatic organisms. The current study aims to determine the effectiveness and longevity of the Washington State 60:40 sand:compost (% by volume) bioretention soil media (BSM) at various infiltration depths, including those shallower than the depth currently required by the Washington Department of Ecology (18”). Experimental columns, containing five different BSM depths, were dosed with roadway runoff at an accelerated rate in order to simulate 13 water years in approximately 4 calendar years. The chemical and biological effectiveness of the columns in treating runoff was assessed using analytical chemistry and the health of two fish species; a common laboratory mode (zebrafish; *Danio rerio*) and a native salmonid species of concern (coho salmon; *Oncorhynchus kisutch*).

Bioretention treatment efficiently removed total lead, zinc, PAHs, and suspended solids (> 90% removal). Total copper (73%), nickel (59%), and arsenic (37%) were more modestly removed, in part due to leaching from the organic fraction of the media. Dissolved organic carbon and nutrients were generally exported from the BSM. Although all treatments continued to export nutrients after 13 accelerated years, the export was greatly reduced by the end of the first accelerated year. Across the study, the shallowest media depth (6”) released lower concentrations of nutrients, DOC, TSS, arsenic or nickel than the deeper media treatments, but was less effective at capturing copper than the 12” or 18” depth. Depth did not impact effluent concentrations of total PAHs, Zn, or Pb.

No trend towards contaminant breakthrough was observed across the course of the 13 accelerated treatment years. Instead, the media began to be unable to infiltrate stormwater beginning in water year 7. This was demonstrated by influent water exceeding the ponding depth (3-in) and also by saturated hydraulic conductivity reduced from >400 cm/hr at study initiation to <100 cm/hr. Surface samples removed at this time showed that the standard bioretention depth of 18” retained a higher porosity than the shallower media and also contained lower concentrations of metals and PAHs than the 6” depth.

Influent stormwater runoff induced morphometric changes and detoxifying enzymes in zebrafish and was acutely lethal to juvenile coho salmon (56–100% mortality). Bioretention treatment tended to improve morphometric impairment and reduced/prevented induction of *cyp1a*. Bioretention treatment of all depths continued to prevent coho mortality throughout the study, associated with high rates of 6PPD-quinone removal (>90%). Morphometric and molecular indicators of toxicity to zebrafish were improved by bioretention treatment and tended to be more pronounced in the deeper media.

The trade-offs evident in contaminant release from BSM, removal of toxicity, and longevity of hydraulic conductivity between shallower and deeper media suggest that an intermediate depth of media such as 12” could be considered in place of the current standard of 18”.

1. Introduction

1.1. Context

Urbanization poses increasing threats to aquatic ecosystems (Miltner et al., 2004; Walsh et al., 2005) including increased chemical loading. In urbanized environments pervaded with impervious roads, parking lots, and sidewalks, infiltration of rainfall is greatly reduced. Instead, a majority of rainfall runs untreated into surface waters, collecting a concoction of contaminants including nutrients, metals, and synthetic organic compounds.

Degraded aquatic ecosystems in urbanized areas are often characterized by an absence or decreased abundance of sensitive species (Walsh et al., 2005). The impacts of urbanization on biological integrity are especially evident in the lowland, urban streams of western North America, where adult coho salmon (*Oncorhynchus kisutch*) returning to spawn in the fall have been prematurely dying at high rates (Feist et al., 2017; Scholz et al., 2011). Coho salmon serve as a sentinel species for the impacts of stormwater runoff, in part because of their sensitivity to water quality (McIntyre et al., 2018), but also because they prefer low-order streams that are most vulnerable to impacts associated with development (Feist et al., 2017). Threats to wild coho salmon are exacerbated by increasing urbanization and population growth, which increase the loading of runoff pollution to water bodies (Spromberg & Scholz, 2011).

Stormwater runoff was defined as a point source of downstream water quality impairment in the 1987 amendments to the U.S. Clean Water Act (National Research Council, 2009). These amendments require that municipalities receive authorization for stormwater discharges under the National Pollutant Discharge Elimination System (NPDES) permit, develop a plan for stormwater management, and implement control measures that improve runoff quality. A group of emerging technologies intended as control measures to alleviate threats to surface waters are called Green Stormwater Infrastructure (GSI). One such GSI is bioretention, a technology that simultaneously addresses issues of stormwater quantity (through infiltration) and quality (through the physical, chemical, and biological removal of pollutants). Bioretention cells are shallow depressions of engineered porous media (Davis, 2005). The goal of these systems is to mimic the ability of undeveloped landscapes to capture and filter runoff.

Bioretention systems have demonstrated high removal efficiency for polycyclic aromatic hydrocarbon (PAHs), metals, and nutrients (Davis et al., 2009; Hunt et al., 2012). Previous studies have additionally established the effectiveness of bioretention treatment systems in preventing acutely lethal and sublethal toxicity of urban runoff exposure to aquatic organisms (McIntyre, Edmunds, Redig, et al., 2016; McIntyre et al., 2014, 2015; Spromberg et al., 2016). Spromberg et al. (2016) found that adult coho salmon exposed to untreated highway runoff displayed symptoms of an acute spawner mortality syndrome, followed by 100% mortality. Treatment of runoff with bioretention prevented symptoms and mortality (Spromberg et al., 2016). Soil bioretention is also protective of juvenile coho salmon and their prey. While untreated highway runoff was acutely lethal to juvenile coho salmon (100% mortality within 12 hours of exposure), complete protection was conferred by treatment of the runoff with bioretention soil media (McIntyre et al., 2015). Filtration through bioretention soil media also ameliorated the sublethal effects of stormwater runoff to the invertebrate prey of juvenile coho.

1.2. Problem and significance

Bioretention systems using native soils or sand/compost mixes are appealing because of their demonstrated effectiveness, but also because they are simple and relatively inexpensive relative to other control measures. However, there remains uncertainty about how to optimize bioretention design. Few field or laboratory studies have evaluated bioretention systems greater than two years old and there is concern about their long-term performance. The long-term accumulation of pollutants in bioretention soil media (BSM) could exhaust the system's capacity for removal and lead to a buildup (potentially to hazardous levels) and subsequent breakthrough of contaminants, deleteriously affecting the quality of receiving waters (Hatt et al., 2011). An accumulation of particulates associated with stormwater runoff in BSM may also lead to physical clogging, reducing the system's hydraulic capacity and related pollutant reduction capabilities (Li & Davis, 2008a). In either instance, the bioretention system is rendered ineffective and media replacement is necessary.

Current understanding of bioretention lifespan generally follows three stages. In a model outlined by Guo et al. (2018), one- to three-year-old bioretention cells demonstrate increasingly high rates of contaminant reduction. At this stage, the rate of adsorption is greater than rates of microbial degradation or plant uptake. Within five to eight years, pollutant reduction rates decline until bioretention cells reach a stable equilibrium period, during which rates of contaminant removal remain relatively constant. Within 10-15 years, sorption sites are nearly completely saturated and bioretention cells rely mainly on plant uptake and microbial degradation to remove pollutants. This is considered the end of life for a bioretention cell, at which point media replacement is recommended (Guo et al., 2018). However, this simplified trend is not necessarily representative of all bioretention cells in all environments.

The lifespan of bioretention systems can be affected by site-specific factors, including filter media, confluence area, and influent water quality and quantity. Bioretention soil media depth is one such factor. However, like many site-specific factors, BSM depth is not consistent across studies or municipalities. A BSM depth of 18 inches is required by the Washington State Department of Ecology Stormwater Management Manual for Western Washington (SWMMWW) (Ecology, 2012). Some studies have suggested that a shallower design may be adequate for improving water quality (DiBlasi et al., 2009; Winogradoff & Coffman, 1999). Shallower media depths would reduce installation and maintenance costs and potentially reduce the export of pollutants from bioretention media. For example, BSM containing a high compost fraction is known to export nutrients. Additionally, deep media designs may not be feasible in some areas, such as those with a high water table. Shallower media designs, therefore, are preferable to deeper designs, provided they are able to deliver comparable reductions in pollutant loading and toxicity to aquatic organisms. The potential redundancy of deep media designs is supported by multiple studies showing high removal of metals within the uppermost layer of BSM, with significant metal uptake within the surface mulch layer (Blecken, 2009; Davis et al., 2001, 2003; Hatt et al., 2008; Li & Davis, 2008b; Poor & Wagner, 2017;

Winogradoff & Coffman, 1999). Similarly, the majority of PAHs appear to be captured in the top few centimeters of soil media (DiBlasi et al. 2009).

However, other studies maintain the importance of greater media depth designs, especially in increasing residence time, sorption site availability, and surface area for microbial attachment and growth (LeFevre et al., 2015). Davis et al. (2006) reported that total phosphorous removal, most likely through adsorption, increased with depth up to approximately 60-80 cm (24-31 in). Depth may also be important in increasing runoff storage capacity and reducing runoff volume. A large storage capacity slows peak flows, and allows for greater contact time between the soil media and runoff (Li et al., 2009; Li & Davis, 2009; Willard et al., 2017). Deeper media depths also reduce outflow volumes by promoting greater evapotranspiration and exfiltration (R. A. Brown & Hunt, 2011; Li et al., 2009). This reduction in outflow volume then corresponds to a reduction in pollutant loads to receiving water bodies. In this way, the hydrologic performance and water quality benefits of bioretention systems are intrinsically linked (Li & Davis, 2009). This implies that soil media depth and water quality benefits are also linked. However, the extent of this relationship is not clear. Additional studies are needed to further investigate the uncertainty that surrounds the long-term performance and sizing requirements of bioretention systems.

1.3. Response

The goal of this study was to explore the life expectancy of bioretention of various depths. We constructed replicated bioretention modules with five depths (6-18 inches) of the Washington State standard BSM containing 60:40 sand:compost. The modules were dosed with field-collected stormwater runoff at an accelerated rate to simulate 10 water years of treatment. The chemical and biological effectiveness of the modules in treating urban stormwater runoff was assessed using analytical chemistry and toxicology parameters for two fish species: zebrafish (*Danio rerio*) and coho salmon. Zebrafish are small, highly studied freshwater fish that have been used extensively in environmental toxicological research (Scholz et al., 2008), including studies monitoring gene expression to quantify contaminant exposure from stormwater runoff (McIntyre et al., 2016). Coho salmon were used because of their sensitivity to stormwater runoff, and previous use in testing the success or failure of bioretention treatment of stormwater runoff (McIntyre et al., 2015). Compared with adults, juvenile coho are easier to handle, more readily available, and can be exposed to smaller volumes of test waters. Importantly, juveniles have a similar response as adults to stormwater runoff exposure (Chow et al., 2019).

The study aimed to answer the following questions:

When do improvements in water quality from bioretention treatment decline?

Hypothesis 1: Chemical or hydraulic performance will decline within five to eight years of operation.

Rationale: Few studies have examined the treatment performance of bioretention systems beyond five years of service. In the few studies that evaluated bioretention over a longer time frame (Hatt et al., 2011; Paus et al., 2014; Willard et al., 2017), some indicated a loss of chemical performance within 5-10 years, while others did not. For example, based on accelerated dosing of small-scale columns, Hatt et al. (2011) estimated that bioretention systems will resist breakthrough for at least 10 years. However, this estimate was for deeper filter media depths (at least 20") with high sorption site availability. Additional factors may result in an overestimation of bioretention lifespan in these studies, including the use of synthetic or low-contaminant runoff. The use of synthetic stormwater neglects the complexity of actual stormwater runoff, which contains myriad contaminants and their transformation products (Du et al., 2020; Peter et al., 2018). Because many of these chemicals have not yet been identified or characterized, we don't know how this complex mixture could affect bioretention removal mechanisms. For example, sorption sites may be filled more quickly with a complex mixture. Site- and study-specific factors make it difficult to directly compare bioretention lifespan across studies. However, a greater archive of these longevity-focused studies will help elucidate the long-term performance and maintenance requirements of bioretention systems.

Hypothesis 2: Declines in biological protection performance of bioretention will precede declines in chemical performance.

Rationale: Initial declines in pollutant reduction capabilities of bioretention may be too subtle to be detected by conventional analytical chemistry. As sorption sites begin to fill, analytically measurable contaminant breakthrough may be preceded by leaching of contaminants that are either below detectable limits or not commonly measured. For example, PAHs measured at concentrations near or below detection limits in bioretention-treated effluent still evoked a transcriptional response in zebrafish, indicating the bioavailability of these or related contaminants (McIntyre et al., 2016). Additionally, the contaminant primarily responsible for the premature mortality of coho salmon is a recently identified chemical that is not currently measured in bioretention studies (Tian et al., 2021). These observations underline the importance of biological rather than chemical markers as more sensitive indicators of water quality impairment.

Are temporal changes in bioretention treatment of stormwater runoff dependent on depth?

Hypothesis 1: Deeper media will initially leach greater concentrations of contaminants present in BSM and for a longer period of time.

Rationale: Bioretention soil media with high compost volumes are associated with undesirable leaching, especially for nutrients (Davis et al., 2009; A. Taylor et al., 2018). Taylor et al. (2018) reported substantial leaching of nutrients from a standard 60:40 sand:compost BSM mixture over 400 days. The study also reported net export of total metals from the BSM, which changed to net retention as the study progressed (A. Taylor et al., 2018). Deeper media will likely leach higher concentrations of these contaminants and for a longer period of time.

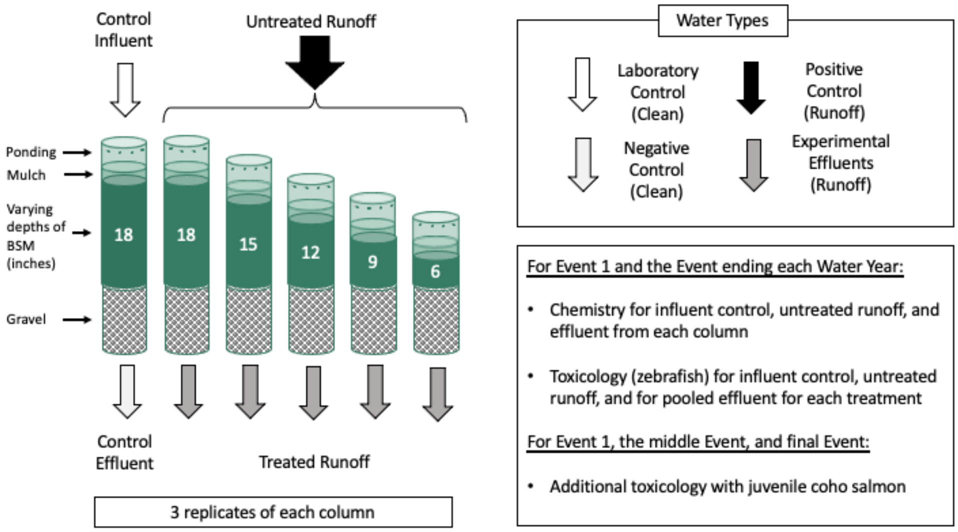
Hypothesis 2: Deeper media will delay chemical and toxicological declines in bioretention performance.

Rationale: The upper horizon of bioretention is typically the area with the greatest pollutant accumulation (Davis et al., 2003; DiBlasi et al., 2009). However, as available sorption sites of the upper horizon fill, the active loading zone of the column will work downward (Davis et al., 2010). Shallower media depths are more limited in how far down this loading zone can extend before sorption capacity is exceeded and breakthrough occurs.

2. Materials and Methods

2.1. Treatments

Eighteen experimental columns for treatment of stormwater runoff were installed in a temperature-controlled chamber (Environmental Growth Chambers, Chagrin Falls, OH) at the Puyallup Research & Extension Center of Washington State University (WSU-P). Fifteen of these modules comprised five incremental BSM depths (18”, 15”, 12”, 9”, and 6”) in triplicate (Figure 1). Three additional columns containing 18” of BSM were used as a clean water control to test the effect on effluent of changes in the BSM over time. After two accelerated water years, the number of treatments was reduced and the study continued with the depths: 18”, 12”, and 6”.



For Event 1 and the Event ending each Water Year:

- Chemistry for influent control, untreated runoff, and effluent from each column
- Toxicology (zebrafish) for influent control, untreated runoff, and for pooled effluent for each treatment

For Event 1, the middle Event, and final Event:

- Additional toxicology with juvenile coho salmon

Figure 1. Study design.

2.2. Endpoints and Frequency of Measurement

Bioretention effectiveness for treating stormwater runoff was determined by comparing the chemistry and toxicity of influent stormwater to effluent waters (filtered through bioretention). Bioretention modules were dosed with runoff from 78 storm events across the approximately four-year study. For each storm event, basic water quality parameters (pH, conductivity, and turbidity) were measured. For the first event and the events ending each simulated water year, influent and effluent waters were sub-sampled for full chemistry analysis and toxicology testing. Saturated hydraulic conductivity was additionally measured at the start of the experiment and at the end of each water year.

Prior to the start of the experiment, triplicate samples of sand and compost were analyzed for chemical composition, including metals, nutrients, organic matter, total organic carbon, total solids, and PAHs. Sand and compost samples were submitted in triplicate to Spectra Laboratories for quantification of total metals (EPA 200.7), ammonia (SM 4500 NH3 D), total nitrogen (summation), nitrate-nitrite (Easy (1-Reagment)), total phosphorus (SM 4500 P E), organic matter (ASTM D2974-13), total organic carbon (EPA 9060), and % solids (SM 2540 G). BSM components were additionally sent to Analytical Resources, Inc. for analysis of PAHs (EPA 8270D-SIM). Given the low surface-to-volume ratio for mulch and gravel, these components were not analyzed.

Additionally, triplicates of each of the BSM components (sand, compost, gravel, and mulch) were leached according to EPA method 1312 (EPA, 1994) to determine the extraction potential of the BSM components. Briefly, 5-L polypropylene beakers were filled with 3L reverse osmosis (RO) water and 150g of one of four bioretention media components. Prior to the addition of the BSM component, the pH of the extraction fluid (RO water) was adjusted with sulfuric acid/nitric acid (60/40 weight percent mixture) to a pH of 5.00 ± 0.05. Beakers were placed on an orbital shaker (rpm=9.2) and agitated for 18 hours. Following the 18-hour extraction, the liquid and solid phases were separated by filtering through acid-washed 0.7 µm glass fiber filters (Whatman, Cat No 1825-047). Prior to filtration, the extract was centrifuged at 14,000 rpm for 20 minutes to promote the settling of larger particles in the leachate and aid filtration. The collected extract was then assessed for PAHs at Analytical Resources, Inc. (ARI) and for metals (total and dissolved), fecal coliform, total suspended solids, dissolved organic carbon, pH, orthophosphate, and nitrite+nitrate (referred to as NOx) at Spectra Laboratories. Alkalinity was determined by Standard Method 2320 B (Titration Method) at the WSU-P Aquatic Toxicology Laboratory. Additional samples were leached with fish rearing water (clean water) and assessed for acute toxicity using the zebrafish embryo model at WSU-P. Clean water was de-chlorinated municipal water treated by RO and reconstituted with salts for rearing fish at WSU-Puyallup. A 1M bicarbonate solution was added to samples prior to toxicological testing to obtain a pH within the range used for zebrafish rearing (7.5-7.8).

Prior to the application of stormwater runoff, experimental modules were flushed with clean water to characterize the chemical composition of water passing through the BSM. A volume of three times the pore volume of the BSM for each module was applied to each column at 29-50 mL/min (avg = 38). The effluent from these three pore volumes was discarded and an additional three pore volumes of clean water was applied. Influent and effluent waters from these next three pore volumes were sampled for chemical analyses. The three pore volumes were pooled to obtain a sufficient volume for all chemical analyses. PAHs were not analyzed during column conditioning because they were expected to be very low based on previous conditioning experiments with the SWMMWW BSM (McIntyre et al., 2020).

2.3. Bioretention Module Construction

Bioretention modules were constructed from pipe (6” diameter) of polyvinyl chloride (PVC), cut to five different lengths to accommodate the initial five different BSM depths (6-18”) plus a gravel drainage layer (12”), a mulch layer (2”), a zone for ponding of water (3”), and a zone for air flow (1”) containing a row of eight ¼” holes (Figure 2). Prior to packing with BSM components, the inside surface of the pipes was scoured using a wire brush to deter preferential flow and lightly washed with a dilute acetone mixture (approximately 90% water, 10% acetone) to remove organic contaminants.

The base of each pipe was fit with a PVC reducer, 45-degree elbow slip, and ball valve slip. All PVC components were joined using PVC primer and cement. To achieve a consistent cross-sectional area of BSM within the columns, the reducer at the base of the columns was filled with food-safe glass marbles (12-mm diameter) after acid-washing in hydrochloric acid and oven-drying at 490 °C for six hours. Marbles were placed on top of and then covered with stainless-steel mesh (#20; 0.9-mm holes) to retain the bioretention system components.

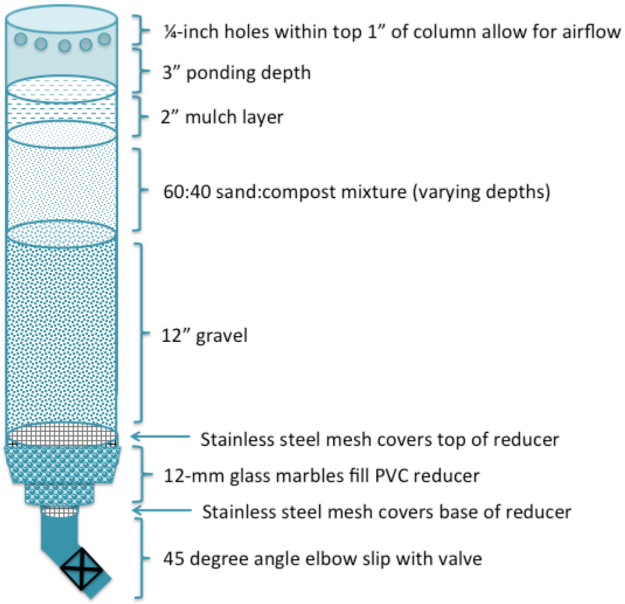


Figure 2. Bioretention module construction.

PVC modules were filled with the bioretention system components. Compost, sand, and arborist mulch were donated by Cedar Grove Composting, Inc. (Maple Valley, WA, USA). Gravel aggregate (3/4" washed sandy gravel; City of Seattle type 26) for drainage beneath the bioretention media was donated by Washington Rock Quarries, Inc (Orting, WA, USA). Each of these materials met specifications for use in bioretention systems defined by the Stormwater Management Manual of Western Washington (Ecology, 2012).

The gravel drainage layer (12 inches) was placed above the meshed marbles and the BSM above the gravel. To obtain a consistent and well-mixed BSM, 10-L batches were individually prepared. Compost was sifted through a 1.3-cm screen (100% passing) to achieve a relatively even density. Sand and compost were then proportioned into each batch by volume to achieve 60:40 sand:compost by volume. Each batch was prepared in a contractor bag and shaken until well homogenized. The wet weight of each batch was recorded, and sub-samples from each batch collected for moisture content analysis (ASTM D2216-10). Moisture content data was used to estimate the dry mass of each BSM batch. The dry mass of each batch was then used to determine the wet mass to be added to each column to achieve the target bulk density. A saturated hydraulic conductivity (K_{sat}) rate of >50 cm/hour (> 20 inches/hr) was targeted. To achieve this desired K_{sat}, a bulk density of 1.4 g/cm^3 was targeted based on a plot of saturated hydraulic conductivity as a function of soil bulk density experimentally derived for the same BSM formulation by Taylor et al. (2018). As BSM was added to the columns, it was compacted every 3” using a 5.5” tamper. Following compaction, columns were covered with 2” of arborist mulch. Bioretention modules were leveled with wooden shims on a wood frame above stainless-steel containers (14” diameter, 25.25” height) to collect effluent.

2.4. Hydraulic Conductivity Testing and Maintenance Actions

2.4.1. K_{sat} Measurements

Saturated hydraulic conductivity through the BSM was assessed at the end of every water year using the falling head method (Klute and Dirksen 1986). Saturation via water applied at the surface instead of through the bottom of the column was used to minimize the problematic bulk movement of fines that was observed in a previous SAM bioretention project (SAM project: “Field Test of Plants and Fungi on Bioretention Performance Over Time”). Once the columns were filled to a marked level 10 cm above the mulch layer, they were left to saturate for 24 hours after which columns for which the water level had dropped as a result of air escaping from the media were topped off with additional water. The valves at the base of each column were then opened and water was allowed to drain from the columns. The time required for the water to drain from the marked level to the soil surface was recorded. Values per column were averaged for each treatment. The test was also used to assess the effect of intermittent maintenance activities.

2.4.2. Surface Bioretention Media Removal and Backflushing

Clogging began to occur in some replicates during WY7. During WY8 we develop a maintenance intervention plan with Ecology based on recommendations from the SWMMWW for addressing excessive ponding or overflow in bioretention systems. Prior to WY9, the top mulch layer was removed and placed in a gallon Ziploc bag. Three undisturbed soil cores (diameter = 1.25”; height = 1”) were removed from the top inch of each column for porosity measurements. From each column, the remaining top inch of BSM was removed and homogenized in a plastic gallon Ziploc bag. Subsamples of BSM were collected from these bags for chemical analyses; PAH analysis by Analytical Resources Inc., and total metals (Cu, Zn, Cd, Pb, As, Ni) analysis by Spectra Laboratories. Remaining soil was aliquoted into 250 mL amber glass jars (stored at -20°C) and 1.5-mL microcentrifuge tubes (-80°C) for potential future tire marker and qPCR analyses, respectively. The mulch layer was then replaced in each column. Prior to doing so, plastic bags containing the mulch were shaken to break up potential biofilm formation. Saturated hydraulic conductivity of each replicate column was measured following the maintenance intervention. When clogging again developed, backflushing was used. Columns were backflushed for 2-3 hours with clean water at 43 rpm (SD=8) of the peristaltic pump used to dose the columns.

2.4.2.1. Porosity and Bulk Density Analysis

For porosity and bulk density measurements, undisturbed soil cores were saturated in a pan of water, ensuring that the water level was even with the level of soil in the core. Cores were saturated from the bottom to remove entrapped air. Once fully saturated, cores were transferred to weigh boats, and the combined wet weights recorded. The cores and weigh boats were then dried in an oven at 105°C for 24 hours. After drying, samples were immediately reweighed. The difference between the saturated and dry weights was used to calculate pore volume. Soil porosity (%) was determined by dividing the pore volume by the total volume of the core. To determine soil bulk density (g/cm^3), soils from dried cores were transferred into (tared) weigh boats to obtain the mass of the soil alone. Bulk density was then calculated as the oven-dry mass of soil divided by the core volume (Hao et al., 2019).

2.5. Urban Stormwater Runoff Collection

Stormwater runoff was collected from two sequential elevated sites on west-bound State Route (SR) 16 (Figure 3A); just west of the I-5/SR 16 interchange in Tacoma, WA (2019-2022) and from the eastern end of the Tacoma Narrows Bridge (2022-2024) under a use permit approved by the Washington State Department of Transportation. The location change, during WY10 (August, between Event 57 and 58), was necessary because vagrant activities at Site 1 had compromised our equipment and posed a risk to the safety of our collection team. At each location, runoff was collected in a 500-gallon high density polyethylene tank from downspouts draining a section of the highway (Figure 3B).

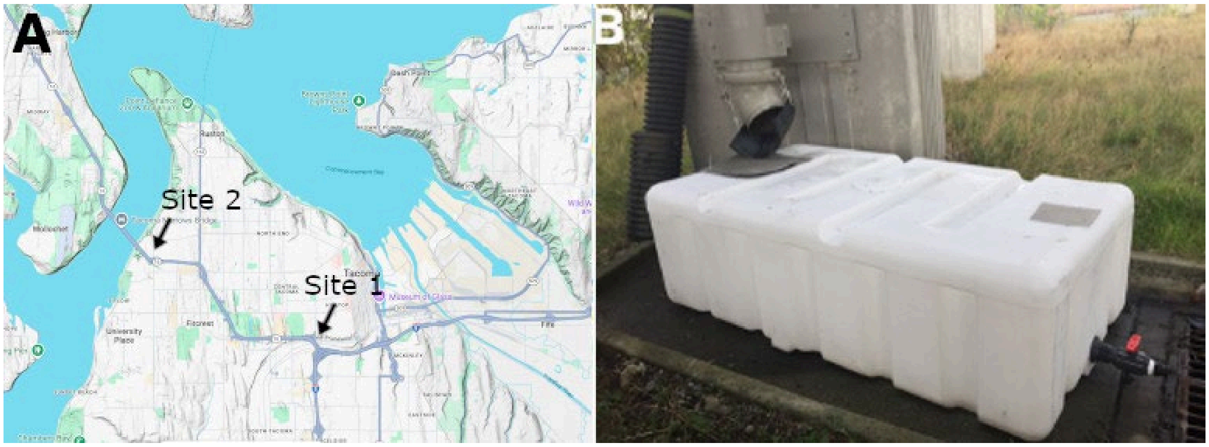


Figure 3. Stormwater runoff collection sites. A) Approximate collection sites on west-bound SR16. Site 1 was used for Events 1-57 (47° 14′ 4.47″ N, 122° 27′ 48.05″ W). Site 2 was used for Events 58-78 (47° 15′ 48.98″ N, 122° 32′ 38.89″ W). B) 500-gallon polyethylene collection tank used to collect urban stormwater runoff.

Runoff accumulated in the collection tanks was transported to WSU-P in a stainless-steel tank (250-gallon) secured to a truck bed and/or trailer (Figure 4A). Before stormwater was transferred from the collection tank to the transportation tank, the water was recirculated within the collection tank for 10 minutes to evenly disperse particulates that may have settled to the base of the tank. Once the stormwater was well-mixed, it was transferred using a gas-powered pump and food-grade hose (Figure 4B). A 400-micron mesh sock was placed over the receiving end of the hose to filter out larger particles that had the potential to clog the tubing system used to deliver stormwater runoff to the experimental columns. Filtering stormwater at this size did not exclude particles most commonly generated on roads from tires and other traffic-related sources. Roadway particles (which include contributions from sources including tires, fuel, brakes, pavement, and atmospheric deposition) collected in a study by Kreider et al. (2010) were unimodally distributed from 4 µm to 280 µm.

Figure 4. Stormwater runoff collection process. A) Stainless-steel transportation tanks at the WSU-Puyallup campus ready to transfer stormwater from the collection site. B) Stormwater runoff pumped from the polyethylene collection tanks to the stainless-steel transportation tanks using a gas-powered pump and food-grade hosing.

At WSU-P, the transportation tank was transferred by forklift and pallet jack inside the environmental growth chamber where the bioretention modules are located (Figure 5). During dosing, the stormwater transportation tank was fitted with a recirculating pump to keep stormwater particulates suspended. A 300-gallon polyethylene tank filled with clean water, used to dose the clean water control experimental modules, was also placed in the chamber. The transportation and clean water tanks were placed inside the environmental chamber so that the temperatures of the influent waters matched that of the ambient chamber temperature.

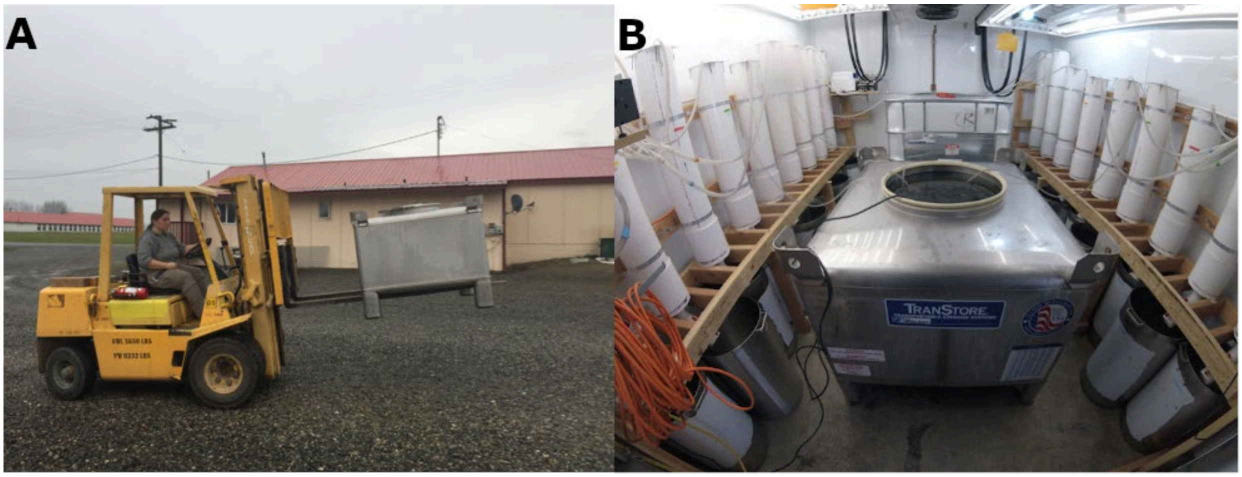


Figure 5. Stormwater runoff transportation. A) Transportation tanks filled with stormwater runoff are moved into the temperature-controlled chamber. B) The clean water tank (rear) and stormwater transportation tank (front) in the temperature-controlled chamber with the experimental bioretention modules.

2.6. Accelerated Schedule

Experimental bioretention modules were dosed with roadway runoff at an accelerated rate that simulated 13 water years. Runoff was collected from 78 storm events, with each collection designated as a separate ‘Event’, and six events completing one water year (WY). Each WY spanned 2-6 months, depending on runoff availability, and was defined as the period over which experimental columns were treated with the equivalent of one year of runoff (Figure 6). Over the duration of the study there were some unplanned extended periods between events. These were due to a variety of conditions that interfered with runoff collection including seasonal drought, vandalism to the collection tanks, and the need to relocate collection to a site protected from vagrants. During these periods, the thermal regime of the simulated season was continued until runoff could again be collected.

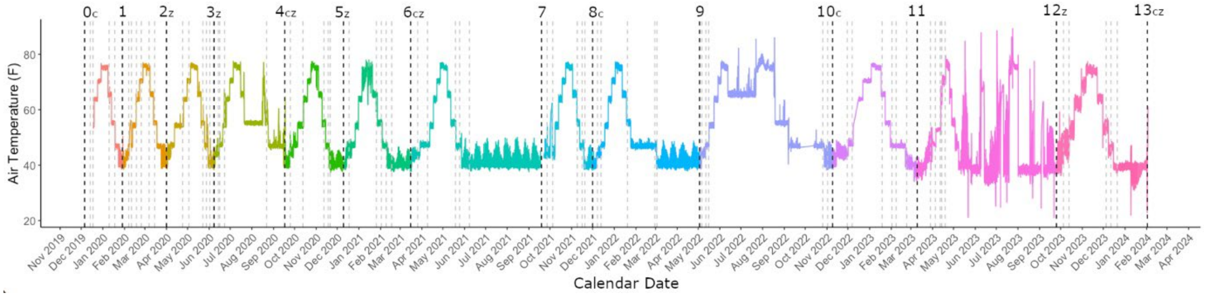


Figure 6. Accelerated schedule for treatment of stormwater by the experimental bioretention columns. Temperature in the environmental chamber reflects seasonal temperatures of the accelerated water year (WY). Each WY consisted of approximately six events indicated by vertical dashed lines. Full water quality sampling coincided with the end of each water year, indicated by bold dashed lines numbered for the water year they represent. Toxicity testing with coho indicated by ‘c’ and with zebrafish by ‘z’.

To account for the discrepancy in thermal conditions between the simulated water year and the actual calendar year, a thermal regime was established to more closely align the climatic and microbial conditions of the simulated water year with those of the natural environment in a typical calendar year. Temperature and relative humidity were controlled in the environmental growth chamber where the columns were housed with a programmable electronic system. Temperature was set to approximate temperatures expected from in-ground installations of bioretention cells at an average 8” soil depth of the simulated month in Puyallup, WA. Relative humidity was similarly set to match the average air relative humidity of the simulated month. The environmental growth chamber was equipped with data logging capabilities. RealTerm Serial Capture software was used to capture temperature and relative humidity data hourly.

2.7. Stormwater Dosing

Stormwater and clean water were delivered to the experimental columns using three peristaltic pumps (MasterFlex L/S 6-600 RPM), each equipped with a three-roller six-channel pump head. A submersible pump within the stormwater tank ensured that stormwater remained well mixed during dosing while peristaltic pumps drew stormwater from the collection tank through one of three PVC manifolds with three to six individual outlets. One additional three-outlet manifold drew water from the clean water tank to dose the three clean water control columns. Two-stop tubing in each channel of the peristaltic pump delivered individual aliquots of the stormwater (or clean water) to each column. Two-stop tubing was 3/16” platinum-cured silicone (Cole-Parmer, Masterflex L/S). Stormwater and clean water were evenly distributed across each column using 3-D printed rainheads affixed to the top of each column. Each rainhead was four inches in diameter and dispensed water via 19 holes 0.8-mm in diameter (see Appendix 1 for diagram). Rainheads were designed using parametric 3D modelling software (SolidWorks 2017). They were printed using a Fused Deposition Modeling (FDM) 3D printer (Prusa i3 MK3S) and HatchBox polyethylene terephthalate glycol (PETG) filament (1.75 mm) at 0.1 mm layer height. Printed rainheads were rinsed with deionized water prior to use.

Per the accelerated schedule, the goal was to simulate five water years in one calendar year. We anticipated being able to collect stormwater runoff from 30 individual storm events in one year. The treatment volume per storm event was 55.6 L/replicate. This value was calculated based on average annual precipitation for Seattle, WA (annual precipitation = 0.9144 m) for a 20:1 contributing area to treatment area ratio multiplied by five WYs. The treatment volume was applied over approximately 24 hours in order to avoid an unrealistically high application rate for a typical Pacific Northwest storm duration. The target application rate (38.6 mL/min or 12.7 cm/hr) was within the infiltration capacity of bioretention according to the Western Washington Hydrology Model (0.5-30.5 cm/hr) and is just twice the rate of a 1” rain event on the 20:1 area over an eight-hour storm (Ecology, 2016).

At the conclusion of WY2, the number of BSM depths being tested was reduced from five to three (18”, 12”, and 6”) for the remainder of the experiment. We chose to continue with these three depths because, by the end of WY2, all five BSM depths were performing equally well in terms of preventing acute toxicity while still performing well in terms of chemistry and hydraulic conductivity.

2.8. Bioretention Stormwater Sampling

Influent waters (runoff and clean water as a control) and effluent waters (filtered through bioretention) were sub-sampled for chemistry and toxicology following the first event and the events ending each WY. For these events, effluent samples were collected from each of the experimental columns (n = 3 per treatment) and were compared with influent samples (n = 1 for each of runoff and control waters). Effluents were sampled from solvent-rinsed stainless-steel pots collecting effluent below the valve draining each experimental column. Stainless-steel paddles were used to mix effluents thoroughly before sampling. Stainless-steel pitchers were then used to grab sub-samples from each of the pots. The influent stormwater sample was collected in an additional stainless-steel pot fed directly by an individual line of stormwater peristaltic pump tubing. The influent clean water sample was collected directly from a line of the clean water tubing.

2.8.1. Chemistry

Water samples for chemical analysis were stored at 4 °C. Water samples were analyzed by Spectra Laboratories (Tacoma, WA) for pH, conductivity, turbidity (ISO 7027 compliant nephelometric method), fecal coliform (SM 9222 D MF), total suspended solids (SM 2540D), dissolved organic carbon (SM 5310 B), alkalinity (SM 2320 B), nitrate + nitrite (Easy (1-Reagent), orthophosphate (EPA 365.3), and total and dissolved arsenic, cadmium, copper, nickel, lead, zinc (EPA 200.8) and dissolved sodium, calcium, and magnesium (EPA 6010). Fecal coliform was discontinued after WY6 because the 6-h holding time was not compatible with our sampling schedule. Water samples for analysis of polycyclic aromatic hydrocarbons (PAH) were field preserved in 10% analysis-grade methylene chloride to prevent degradation prior to analysis by Analytical Resources, Inc. (ARI, Tukwila, WA) for 24 PAH congeners (EPA 8270D - SIM). Spectra did not alert us that they were no longer conducting their own analytical chemistry and were instead sending all samples to ARI for analysis. For WY 12 and WY13, we directly submitted samples to ARI to save the overhead costs charged by Spectra.

Samples for analysis of 6PPD-quinone were submitted to the laboratory of Dr. Kolodziej at the University of Washington. Sample were extracted and analyzed using published methods (Tian et al., 2022). Briefly, 200 mL water samples were spiked with D5-6PPDQ (25 ng) as an internal standard (HPC Standards Inc., Atlanta, Georgia, USA) and concentrated using SPE cartridges (200mg/ 6mL Oasis HLB, Waters, MA). Cartridges were conditioned with 10 mL of methanol and 25 mL of DI water. Cartridges were rinsed with 10mL of water and dried for 10 minutes and then eluted with 10 mL of methanol. Eluates were evaporated under nitrogen to 1 mL and transferred into autosampler vials.

6PPDQ was quantified using an Agilent 1290 infinity HPLC coupled with Agilent 6430A triple quadrupole MS/MS (LC-MS/MS). An Agilent Poroshell HPH-C18 column (2.1×100 mm, 2.7 µm) was equipped with a C18 guard column (2.0 × 4 mm, Phenomenex, Torrance, California) at 45 °C, injection volume 5 µL, and binary gradient of DI water (A) and methanol (B) both with 0.1% formic acid at a 0.2 mL/min flow rate: 50% B 0-0.5 min, 100% B at 10.5-12 min, 50% B at 13-16 min. Detection used electrospray ionization (ESI+) and multi reaction monitoring (MRM) modes with a 300 °C gas temperature, 5 L/min gas flow, 400°C sheath gas temperature, 11 L/min sheath gas flow, 500V nozzle voltage, 3000V capillary voltage, 45 psi nebulizer pressure, and 110 fragmentor voltage.

6PPDQ was quantified using a 7- point calibration curve (0.025-100 ng/mL) with determination coefficient (R^2) value of >0.99. Method detection limit (MDL, 2.5 ng/L) and method quantification limits (MQL, 5.1 ng/L) were determined by signal to noise (S/N) ratios of 3 and 10, respectively.

2.8.2. Toxicity Testing

Toxicity testing was conducted for BSM components prior to the construction of the bioretention modules and for influent and effluent waters over the course of stormwater treatment. Effluent samples from each of the triplicate bioretention modules were composited for toxicological analysis to assess the average performance of each treatment. Biological replicates were used to capture the variability in test conditions. Toxicity testing used two fish species: zebrafish (*Danio rerio*) embryos and coho salmon (*Oncorhynchus kisutch*) juveniles. For assays with zebrafish, water samples for each treatment were frozen at -20 °C in amber glass bottles. For assays with coho salmon, water samples for each treatment were transported from the environmental chamber to the WSU-P Aquatic Toxicology Laboratory in 10-gallon stainless-steel drums. Polypropylene beakers (5 L) were then used to dispense water from the drums into 35-L glass aquaria as described below (Section 2.9.2).

2.9. Toxicological Analyses

2.9.1. Zebrafish

Zebrafish were analyzed for survival and sublethal changes in morphometric endpoints (WY2-3) following previously published methods for urban runoff toxicity to zebrafish embryos (McIntyre et al. 2014). Morphometric analysis was discontinued in favor of transcriptional responses which would be more sensitive (McIntyre et al. 2016). Treatment effects on embryos were analyzed for differential transcription of *cyp1a* using quantitative polymerase chain reaction (qPCR). This gene is transcribed in response to contaminants binding the aryl hydrocarbon receptor. This non-specific response integrates exposure to a variety of contaminants known to be toxic to aquatic life. Due to a freezer malfunction, only samples from WYs 4, 5, 6, 12 and 13 were able to be analyzed.

2.9.1.1. Morphometric Analysis

For each treatment, water toxicity was assessed using 32 individual embryos (2-4 hours post-fertilization; hpf) placed in individual wells of a 96-well glass-lined microplate. Using an autopipette, 250 µL of treatment or control water were added to each well of the microplate in a randomized order. The microplates were placed in an incubator set at 28.5 °C. At 24 hpf, treatment water was replaced, and notes made of any mortalities or obvious developmental delays. At test termination at 48 hours, embryos were dechorionated (if unhatched), mounted in 3% methylcellulose, and imaged with a digital camera mounted on a stereomicroscope (Nikon SMZ800). Images were analyzed using the open-source software Image J (http://rsbweb.nih.gov/ij/) to assess sublethal morphometric endpoints.

2.9.1.2. qPCR Analysis

Groups of zebrafish embryos (30 per replicate x 3 replicates per treatment) were exposed to treatment waters in glass petri dishes (15 mm diameter). Dishes were randomly sorted onto glass trays and moved into a Precision™ Low Temperature BOD Refrigerated Incubator (Thermo Scientific™) maintained at 28.5°C. At 24 h, mortalities were removed from dishes and water was replaced with treatment water acclimated to 28.5°C. At 48 h, dead or severely deformed embryos were removed. The remaining embryos were transferred into 2 mL PCR clean microcentrifuge tubes with excess water removed and flash frozen in an acetone and dry ice bath maintained at –78°C.

Frozen embryos were homogenized in TRIzol™ reagent (5% embryos in Trizol v/v; Invitrogen™) with a 5 mm stainless steel grinding bead on a Bead Ruptor Elite (Omni, Inc.) for one minute at 1.95 m/s. Homogenized samples were stored at -80°C until RNA extraction. BCP (1-Bromo-3-chloropropane, Sigma Aldrich) was added to thawed, homogenized samples at a ratio of 1:10 BCP: TRIzol™. Samples were centrifuged (at 4°C) for 15 minutes at 12,000 rcf, and the aqueous phase containing RNA was transferred to a new 2 mL PCR clean tube with an equal volume of 95% ethanol. RNA purification was carried out using a Zymo Direct-zol RNA miniprep kit (Zymo Research) following the manufacturer’s protocol, including a DNA treatment step with DNAase I. Following RNA purification, RNA concentration (ng/µg L) and quality was quantified using an N60 Implen NanoPhotometer®. Extracted RNA was stored at -80°C until cDNA synthesis. First strand cDNA was synthesized via reverse transcription from 2 µg RNA using Superscript IV VILO Master Mix (Invitrogen™) following manufacturer instructions.

Quantitative Polymerase Chain Reaction (RT-qPCR) was carried out in triplicate 10 µL reactions using PowerTrack SYBR Green reagent (Applied Biosystems, Inc.), 10 ng of template cDNA, and 500 nM gene-specific primers on a QuantStudio5 Real-time PCR System (Applied Biosystems, Inc.). Fast cycling conditions followed manufacturer instruction for the SYBR Green reagent (95°C for 2 min, followed by 40 cycles of 95°C for 5 s and 60°C for 30 s). To verify single product amplification, dissociation curves were generated at the end of all qPCR reactions. Standard curves from a 2X dilution series were run in triplicate for each primer during each assay to estimate amplification efficiency. Four reference genes (*wdtc1*, *ef1a*, *mtm1*, and *rxrba*) were amplified and selected from based on their stability using RefFinder (Xie et al. 2012). Stable reference gene expression was used to normalize *CYP1A* expression. Primers (ordered from Integrated DNA Technologies, Inc.) and their sequences and NCBI accession numbers are shown in Table 1.

Table 1. Primer sequences and NCBI accession number for target and reference genes used in this study.

Gene		Sequence (5'-3')	NCBI accession	Source
cyp1a	F:	GGGAAAGAGTCCCAAATATTCC	NM_131879.1	(McIntyre et al. 2016)
	R:	CTCATATTAACCAGTCGCACCA		
wdtc1	F:	GCAGCGCTCTTCTCCAAAAC	NM_001130606.1	(McIntyre et al. 2016)
	R:	CGACTCCTTCCGGCTGAAAT		
ef1a	F:	CTTCTCAGGCTGACTGTGC	NM_131263	
	R:	CCGCTAGCATTACCCTCC		
mtm1	F:	GAGTCCAGTCGGGTGCTGTA	NM_001037684	
	R:	AGCTCTTTGTAGCGCTGCTC		
rxrba	F:	TGTCAAGCCCAAGTGAAGTG	NM_131275.1	
	R:	CAACCGGAGAAGAAGCTTTG		

2.9.2. Salmon Survival

Juvenile coho salmon survival was assessed for water samples collected from WYs 0, 4 6, 8, 10 and 13. Replicate glass aquaria were filled with influent or effluent waters and maintained at 13 °C in water baths. An airstone was placed in each aquarium to maintain dissolved oxygen levels at 6 mg/L. Prior to beginning the exposure, water quality parameters (temperature, dissolved oxygen, pH, conductivity) were measured for each aquarium. Test chamber size varied across the study to accommodate the size and number of coho used. Following guidelines established for acute toxicity testing by the U.S. Environmental Protection Agency (EPA, 2002), a minimum of 20 fish were used across at least 2 replicates per treatment. At test termination (24 hours), surviving juvenile coho were euthanized in MS-222 (500 mg/L), and their total length and weight recorded.

2.10. Data Analysis Methods

Data organization and analysis were conducted using the statistical computing language R (version 4.0.4) implemented in the RStudio (version 1.4.1103) software environment (R Core Team, 2021; RStudio Team, 2021). Significance level was generally set at $\alpha=0.05$, but trends were at times also interpreted for $0.05<p<0.1$. Differences among treatments were often evaluated with linear mixed effects models (LMM) fitted using the lmer() function in *LmerTest* package (Kuznetsova et al., 2017). Pairwise differences among treatments were evaluated from the estimated marginal means using the emmeans() function in the *emmeans* package (Lenth, 2025), which includes a Tukey p-value adjustment for multiple comparisons. Models specific to each analysis are described below.

2.10.1 Non-detects.

For data analysis purposes, one-half of the value of the method detection limit (MDL) was substituted for the value of the non-detect when there was at least one replicate with a value >MDL. A value of zero was used when there were no detections within a treatment.

2.10.2 Data completeness and detectability

Before beginning statistical analyses, the dataset was assessed for completeness and detectability. We excluded parameters with influent runoff detections below 50% to ensure sufficient detections for a robust analysis (Table 2). Parameters included in the final statistical analysis were total and dissolved arsenic, copper, nickel, zinc, nitrates, orthophosphate (oP), total suspended solids (TSS), dissolved organic carbon (DOC), and total PAHs.

2.10.3 Outliers

We removed eight outliers from the dataset that were presumed detection contamination or transcription error (Table 3).

Table 2. Frequency of detections for measured parameters. Detections below 50% are in italics.

Compound	Detections in Runoff (%)	
	Influent (n=14)	**Effluent (n=126)**
Total Metals		
As	93	74
Cd	*14*	*2*
Cu	100	100
Pb	93	*33*
Ni	100	72
Zn	100	98
Dissolved Metals		
As	9	59
Cd	*7*	*0*
Cu	100	100
Pb	*21*	*16*
Ni	71	70
Zn	100	91
Nutrients		
Nitrate/Nitrite	100	100
Orthophosphate, as P	*43*	98
Conventional		
DOC	100	100
TSS	100	80
Dissolved Ca	100	100
Dissolved Mg	100	100
Dissolved Na	100	100
Microbiology		
Fecal Coliform by MF	71	82 (n=57)
Aromatics		
Total PAHs	79 (n = 315)	*36* (n = 1857)

MF = membrane filtration; through WY6

Table 3. Outliers removed for data analysis.

WY	Compound	Treatment (replicate)	Reason for Removal	Solution
0	Ni	R9 (1) & R12 (2)	Contamination	Replaced with average of other two replicates
0	dNi	R9 (1) & R12 (2)	Contamination	Replaced with average of other two replicates
1	Zn	R6 (2)	Contamination	Replaced with average of other two replicates
1	dCu	R	Transcription error	Increased value by factor of 10
1	DOC	R	Transcription error	Assigned median value
1	Ortho	C18 (1)	Contamination	Replaced with average of other two replicates
6	As	R	Transcription error	Increased value by factor of 10
13	Pb	R	Transcription error	Increased value by factor of 10

2.10.4 Chemical performance by depth and time

Changes in concentration of measured parameters a result of BSM depth across the study were explored by LMM with effluent concentrations from the three retained treatment depths (6”, 12”, and 18”) as response variables and categorical treatment (6”, 12”, 18”) as explanatory variables. Water year was included as a random factor to reduce unexplained variability introduced by the grouping of observations during distinct sampling events. This grouping was observed because effluent contaminant concentrations were strongly influenced by influent concentrations that varied between different stormwater sampling events. Response variables were log-transformed to correct for right-skewed data. R18 was set as the reference treatment for pairwise comparisons. Model: lmer(Analyte ~ Depth +(1|WY)). Fecal coliform was only measured in seven of the 13 events and was therefore not included in the analysis of effectiveness over time.

2.10.5 Chemical removal efficiency/net export.

For each sampled event, removal efficiency or net export for bioretention media treating runoff was calculated based on the concentration of each analyte present in the influent sample. For contaminants with higher concentrations in the influent than the effluent, percent removal efficiency was calculated as: $\frac{\text{Influent}-\text{Effluent}}{\text{Influent}} \text{ x } 100$

where influent is the chemical concentration in stormwater and effluent is the concentration in effluents from each of the bioretention depths. For contaminants with higher concentrations in the effluent than in influent, the net concentration was calculated by subtracting the influent from the effluent concentration. A LMM was used to analyze the data with WY as a random variable and bioretention depth as the fixed variable. R18 was the reference treatment for pairwise comparisons. Model: lmer(Analyte ~ Depth + (1|WY)).

2.10.6 Biological Assays

2.10.6.1 Zebrafish morphometrics

Treatment effects on zebrafish morphometrics were assessed for WY2 and WY3 by linear models for each endpoint (length, eye area, pericardial area, periventral area) and then across the two events by LMM with WY as a random factor. Model: lmer(endpoint ~ depth + (1|WY). Post-hoc analysis tested for differences in endpoint for the three depths of BSM compared with the influent runoff.

2.10.6.2 Zebrafish transcriptional response

Cycling time (Ct) was averaged across three technical replicates. Delta (Δ) Ct was the difference between the mean Ct value of the target gene and the reference gene(s). The $\Delta\Delta$Ct was the difference between the ΔCt of each sample and the average ΔCt of the clean water control replicates. Fold-change (FC) from controls was calculated as 2^-$\Delta\Delta$Ct for visualizing trends relative to controls. Statistical analysis was conducted on the $\Delta\Delta$Ct values for each water year as an analysis of variance using the aov() function followed by a Tukey's posthoc test using the function TukeyHSD(). Model: aov($\Delta\Delta$Ct ~ treatment). To assess effects of treatment across the five water years for which we were able to conduct exposures for qPCR, we used a LMM with $\Delta\Delta$Ct as the dependent variable and water year as a random variable. Model: lmer($\Delta\Delta$Ct ~ treatment + (1|WY)).

2.10.6.3 Coho survival

For most tests of coho survival, mortality was only observed in the runoff exposure. Survival rate was compared with the control survival of 100% using a Fisher's Exact Test, fisher.test() function, in base R. Mortality rates in runoff were always significantly different from controls ($p < 0.001$). For WY 8, in which overflowing water was pooled with the bioretention-treated effluent, there was also some mortality in the R18 treatment, which was not significantly different from controls ($p = 0.107$).

2.10.7 Hydraulic conductivity calculation

Saturated hydraulic conductivity was calculated by the falling head method (Klute & Dirksen 1986):

$$Ksat = \left(\frac{Lc}{t}\right)\ln\left(\frac{Lc + H1}{Lc + H2}\right),$$

where L_c = length of the porous medium in the column, t = time for water to fall from the rim of the ponding zone (H_1) to the surface of the mulch (H_2).

Differences in K_{sat} values among treatments over time were assessed with a LMM. Because we expected K_{sat} to decline over time we used bioretention treatment and water year (WY) as the explanatory variables and replicate as a random effect. Model: KSAT ~ TRMT * WY + (1 | REP). The effect of WY was centered on WY13 to determine effects relative to the end of the study.

3. Results

3.1. Baseline Conditions

3.1.1. Bioretention Components Analysis

Compost and sand used in the bioretention soil media (BSM) were analyzed for a suite of metals (arsenic, cadmium, copper, lead, nickel, zinc), ammonia, total nitrogen, nitrates (the sum of nitrate and nitrite), total phosphorus, organic matter, TOC, and percent solids. Total metal concentrations ranged from < 0.03 mg cadmium per kg of dry sand to 54 mg zinc per kg of dry compost (Table 4). Metal concentrations in compost and sand were compared to EPA's Ecological Soil Screening Levels (Eco-SSLs). Eco-SSLs are threshold contaminant concentrations in soil intended to protect terrestrial ecosystems including criteria for terrestrial plants, soil invertebrates, birds, and mammals. These values are intended to identify concerns during the screening stage of ecological risk assessment (EPA, 2018).

Table 4. Concentrations of total metals in triplicate samples of sand and compost and EPA Ecological Soil Screening Levels. Values presented are mean (standard deviation).

Compound	Detection Limit (mg/kg)	Result (mg/kg dry weight) Compost	Result (mg/kg dry weight) Sand	EPA Ecological Soil Screening Levels
As	0.4	<0.4	<0.4	18-46
Cd	0.03	0.19 (0.02)	<0.03	0.8-140
Cu	0.03	21 (3)	16.6 (0.5)	28-80
Pb	0.03	6.8 (0.6)	1.13 (0.06)	11-1700
Ni	0.5	3.9 (0.5)	40* (4)	38-280
Zn	0.2	54* (4)	29.1 (0.8)	46-160

* exceeds one category of the screening levels

All compost and sand samples were below Eco-SSLs for arsenic, cadmium, copper, and lead. Compost samples were below the Eco-SSLs for nickel and sand samples were below the Eco-SSLs for zinc. Our samples for compost and sand exceeded just two criteria: sand contained higher nickel concentrations than the Eco-SSL value for terrestrial plants (38 mg/kg), and compost exceeded the zinc Eco-SSL for avian wildlife (46 mg/kg). However, compared with western U.S. soils, nickel in sand was less than the maximum reported background concentrations (EPA, 2007a) and zinc in compost was similar to the median background concentration (EPA, 2007b). Additionally, dilution of sand and compost in the 60:40 mixture would result in none of the criteria being exceeded.

Total PAH concentrations in compost and sand samples (Table 5) were much higher in compost (2.03 mg/kg) than in sand (0.013 mg/kg) and were well below most PAH Eco-SSLs. The mean value of high molecular weight (HMW) PAHs in compost (1.6 mg/kg) exceeded the mammalian

Eco-SSL (1.1 mg/kg) but was less than the mean of HMW PAHs for urban soils in the U.S. (2.5 mg/kg; Mauro & Roush, 2008). The distribution of PAHs in compost were similar to those in urban soils with the most abundant PAHs being fluoranthene > pyrene > phenanthrene.

As expected, nutrient and organic matter concentrations were greater in compost than in sand samples (Table 6).

Table 5. Average PAH concentrations (mg/kg) (standard deviation) in triplicate samples of compost and sand. One-half of the value of the detection limit was substituted for the value of non-detects in calculating means; used when the compound was detected in at least one replicate for the treatment. Values following '<' are equal to the detection limit.

Compound	Result (mg/kg dry weight)	
	Compost	Sand
1-Methylnaphthalene	0.009 (0.001)	0.0003 (0.0002)
2-Methylnaphthalene	0.013 (0.002)	<0.00105
Naphthalene	0.016 (0.003)	<0.00121
Acenaphthene	0.02 (0.005)	0.0008 (0.0005)
Acenaphthylene	0.008 (0.002)	<0.00103
Anthracene	0.047 (0.007)	<0.00083
Dibenzofuran	0.03 (0.004)	<0.00131
Fluorene	0.027 (0.006)	<0.0006
Phenanthrene	0.265 (0.03)	0.0011 (0.0009)
Benz[a]anthracene	0.125 (0.02)	0.0005 (0.0003)
Chrysene	0.189 (0.04)	0.0011 (0.00006)
Fluoranthene	0.555 (0.06)	0.0016 (0.0009)
Pyrene	0.353 (0.04)	0.0017 (0.0007)
Benzo(a)pyrene	0.064 (0.01)	<0.00058
Benzo(b)fluoranthene	0.107 (0.02)	0.0009 (0.0004)
Benzo(j)fluoranthene	0.061 (0.02)	<0.00065
Benzo(k)fluoranthene	0.05 (0.009)	<0.00072
Dibenzo(a, h)anthracene	0.01 (0.002)	<0.00085
Benzo(ghi)perylene	0.044 (0.005)	<0.00101
Indeno(1,2,3-cd)pyrene	0.036 (0.006)	<0.001
Total PAHs	2.030 (0.09)	0.013 (0.002)
Sum High Molecular Weight (HMW)	1.596 (0.09)	0.008 (0.001)
Sum Low Molecular Weight (LMW)	0.435 (0.03)	0.005 (0.001)

Table 6. Nutrient concentrations in triplicate samples of sand and compost. Values presented are mean (standard deviation).

Compound	Reporting Limit (mg/kg)	Result (mg/kg dry weight)	
		Compost	Sand
Nitrates (mg/kg-N)	0.1	9 (5)	0.3 (0.1)
Ammonia (mg/kg-N)	4	85 (37)	16 (12)
Total Phosphorous (mg/kg)	0.1	45 (36)	1.6 (0.5)
Total Nitrogen (mg/kg-N)	10,000	94 (35)	16 (13)
Organic Matter (wt. % dry)	n/a	57 (2)	1.17 (0.06)
Total Organic Carbon (mg/kg)	50,000	247,000 (16,462)	17,333 (2,517)
Total Solids (wt. %)	n/a	42.5 (0.8)	95.4 (0.2)

3.1.2. Bioretention component extraction analysis

Bioretention soil media components (sand, compost, gravel, and mulch) were extracted with a mildly acidic solution (pH: 5.00 ± 0.05) to determine their leaching potential.

3.1.2.1 Metals

Overall, greater concentrations of metals were extracted from compost and mulch than from the other BSM components. Zinc was the dominant metal that was extracted from all the BSM components in the order of mulch > compost > gravel > sand (Table 7).

3.1.2.2 PAHs

PAHs that were extracted from the bioretention components were predominantly low molecular weight, dominated by phenanthrene (Table 8). Mulch extract contained the highest concentrations of PAHs (TPAH = 0.986 µg/L) whereas in extracts of compost, sand, and gravel TPAH was 0.05-0.11 µg/L. Among the PAHs in the mulch extract, phenanthrene concentrations were greater even than concentrations measured in road runoff (McIntyre et al., 2014; Spromberg et al., 2016).

Table 7. Summary of total and dissolved metal concentrations in extract of bioretention components. Values presented are mean (standard deviation).

Compound	Detection Limit (µg/L)	Leachate (µg/L)			
		Compost	Sand	Gravel	Mulch
Total					
As	0.05	11 (1)	0.3 (0.3)	0.4 (0.4)	1.6 (0.2)
Cd	0.05	<0.05	<0.05	<0.05	<0.05
Cu	0.05	23 (21)	0.6 (0.3)	0.4 (0.5)	5.4 (0.4)
Pb	0.079	3 (3)	<0.079	<0.079	0.5 (0.1)
Ni	0.2	12 (5)	1.5 (0.3)	0.7 (0.6)	2.4 (0.5)
Zn	0.19	161 (31)	55 (13)	96 (18)	199 (50)
Dissolved					
As	0.05	11 (1)	0.2 (0.2)	<0.05	1.4 (0.2)
Cd	0.05	<0.05	<0.05	<0.05	<0.05
Cu	0.05	11 (2)	0.3 (0.4)	0.2 (0.3)	3 (3)
Pb	0.079	0.8 (0.2)	<0.079	<0.079	0.6 (0.4)
Ni	0.2	8.1 (0.5)	1.2 (0.3)	0.4 (0.5)	2.1 (0.3)
Zn	0.19	140 (40)	51 (10)	89 (10)	181 (40)

Table 8. Average PAH concentrations (µg/L) (standard deviation) in extract from each bioretention component. One-half of the value of the detection limit was substituted for the value of non-detects in calculating means; used when the compound was detected in at least one replicate for the treatment. Averages following '<' are equal to the detection limit.

Compound	Extract (µg/L)			
	Compost	Sand	Gravel	Mulch
1-Methylnaphthalene	0.001 (0.001)	0.003 (<0.001)	0.003 (<0.001)	0.05 (0.01)
2-Chloronaphthalene	<0.002	<0.001	<0.002	<0.001
2-Methylnaphthalene	0.005 (0.001)	0.004 (<0.001)	0.006 (0.002)	0.002 (0.002)
Naphthalene	0.014 (0.003)	0.01 (0.001)	0.015 (0.005)	0.1 (0.003)
Acenaphthene	<0.004	<0.004	<0.004	0.009 (<0.001)
Acenaphthylene	<0.003	<0.003	<0.003	<0.003
Anthracene	0.001 (<0.001)	<0.002	0.002 (<0.001)	<0.002
Carbazole	0.003 (0)	<0.002	<0.002	<0.002
Dibenzofuran	0.007 (0.002)	0.003 (0)	0.008 (0.004)	0.039 (0.002)
Fluorene	0.008 (0.002)	0.003 (0)	0.011 (0.006)	0.12 (0.01)
Phenanthrene	0.023 (0.003)	0.011 (0.002)	0.022 (0.006)	0.62 (0.09)
Benz[a]anthracene	0.004 (0)	<0.001	<0.001	<0.001
Chrysene	0.005 (<0.001)	<0.001	<0.001	<0.001
Fluoranthene	0.015 (0.002)	<0.002	0.003 (0.003)	0.006 (0.002)
Pyrene	0.005 (0.002)	0.001 (<0.001)	0.01 (0.01)	0.017 (0.004)
Benzo(a)pyrene	<0.004	<0.003	<0.004	<0.003
Benzo(b)fluoranthene	0.002 (<0.001)	<0.001	<0.001	<0.001
Benzo(j)fluoranthene	<0.003	<0.003	<0.003	<0.003
Benzo(k)fluoranthene	<0.005	<0.004	<0.005	<0.004
Dibenzo(a, h)anthracene	<0.002	<0.002	<0.002	0.004 (0.003)
Perylene	<0.008	<0.008	<0.008	<0.008
Benzo(ghi)perylene	0.001 (<0.001)	<0.002	<0.002	0.005 (0.001)
Indeno(1, 2, 3-cd)pyrene	0.001 (<0.001)	<0.001	<0.001	0.004 (0.002)
Total PAHs	0.111 (0.006)	0.054 (0.002)	0.1 (0.02)	0.99 (0.09)
Sum High Molecular Weight*	0.044 (0.003)	0.015 (<0.001)	0.02 (0.01)	0.046 (0.006)
Sum Low Molecular Weight**	0.066 (0.005)	0.039 (0.002)	0.07 (0.01)	0.94 (0.09)

*Compounds composed of four or more rings

**Compounds composed of fewer than four rings

3.1.2.3 Nutrients & Conventional Chemistry

Compost extract had the highest alkalinity, hardness, and concentration of nitrates, whereas mulch extract contained the highest concentration of orthophosphate and DOC (Table 9). Both compost and mulch were sources of TSS.

Table 9. Summary of nutrient and conventional parameters in extract of bioretention components. Values presented are mean (standard deviation).

Compound	Detection limit (mg/L)	Leachate (mg/L)			
		Compost	Sand	Gravel	Mulch
Nitrates	0.003	79 (60)	1.0 (0.2)	5 (3)	0.3 (0.1)
Orthophosphate	0.01	1.5 (0.1)	<0.01	0.02 (0.02)	9.8 (0.1)
DOC	0.08	32.3 (0.3)	<0.08	<0.08	147 (9)
Alkalinity*	n.a.	39 (13)	0.8 (0.6)	1 (1)	3.88 (0.06)
Hardness*	calculated	11.1 (0.6)	1.11 (0.07)	1.5 (0.2)	0.015 (0.002)
TSS	0.5	1 (1)	0.4 (0.2)	0.6 (0.3)	1.3 (0.3)
Sodium	27	29500 (2000)	800 (90)	876 (100)	4250 (670)
Magnesium	1.9	881 (50)	199 (4)	140 (8)	1443 (184)
Calcium	3.4	3010 (100)	118 (20)	358 (90)	3870 (545)

* as $CaCO_3$

3.1.3. Export of contaminants following clean water conditioning

Prior to the application of stormwater runoff, experimental columns were conditioned with three pore volumes of clean water. Influent and effluent waters from an additional three pore volumes of water were sampled for chemical analyses. Metals and conventionals were measured in the conditioning water, but not PAHs due to the high cost and low concentrations released during the leaching tests of the individual BSM components.

3.1.3.1 Metals

All metals except cadmium and dissolved lead leached from the columns during clean water conditioning. Copper was more concentrated than other tested metals in the effluent of clean water flushed through the experimental columns (Table 10). For each BSM treatment depth, metals were detected in effluent samples in the order of copper > nickel > zinc > arsenic > lead. Effluents from different treatment depths differed significantly in concentrations of arsenic, zinc, and copper. The 15” and 18” BSM treatment depths tended to leach greater concentrations of arsenic, zinc, and copper than the 6” and 9” treatment depths (Table 10). Total lead followed the same trend but was not statistically different among treatments. Higher analyte concentrations in the deeper treatments were unexpected because the different treatment depths were conditioned with a volume of water proportional to the volume of BSM, which was expected to cause a similar degree of leaching for the BSM. These differences could be due to the small sorption capacity of the gravel layer, the depth of which was constant in all of the columns regardless of treatment depth. In the columns with the lesser treatment depths, the majority of the metals leached from the BSM media may have sorbed to the gravel. In the larger columns, which were flushed with a greater volume of water, the sorption capacity of the gravel may have been exceeded before all of the water had been flushed through the columns. In the influent water sample, only copper and nickel were above the analytical detection limits. Results for dissolved metals were very similar and followed the same general patterns as for total metals.

3.1.3.2 Nutrients & Conventional Chemistry

All nutrients and conventional analytes were higher in effluent than influent water flushed through the columns during clean water conditioning. Nitrates were below the analytical detection limit in the influent water sample but were more concentrated than orthophosphate in effluent from all of the columns (Table 11). Unlike nitrates, orthophosphate concentrations were significantly greater in the 15” and 18” columns compared to the 6” and 9” treatment depths. Similar to metals, this trend may be the result of orthophosphate sorption to the gravel layer, and potential sorption site saturation in the deeper columns. However, compared to nitrates, orthophosphate concentrations were relatively low in effluent from all columns.

Table 10. Summary of total metal concentrations during clean water conditioning of bioretention soil media for influent (clean) water and triplicate effluent samples from each of the five treatment depths plus the clean water control (C18). Values presented are mean (standard deviation).

Analyte	Detection Limit (µg/L)	Effluent Water (µg/L)						
		Influent Water	R6	R9	R12	R15	R18	C18
Total								
As	0.05	<0.05	2.1 (0.5)^{a}	3 (1)^{a}	6.5 (0.4)^{ab}	7.3 (0.7)^{b}	7.2 (0.7)^{b}	7.6 (0.5)^{b}
Cd	0.05	<0.05	<0.05	<0.05	<0.05	<0.05	<0.05	<0.05
Cu	0.2	1.3	34 (2)^{ac}	41 (7)^{ac}	50 (1)^{bc}	48 (5)^{bc}	48 (2)^{bc}	47.1 (0.5)^{c}
Pb	0.079	<0.079	0.4 (0.4)	0.5 (0.5)	1.07 (0.06)	1.2 (0.2)	1.2 (0.2)	1.2 (0.4)
Ni	0.20	0.9	14 (1)	15 (1)	17.2 (0.4)	17 (2)	17.9 (0.5)	18 (1)
Zn	0.19	<0.19	4.0 (0.6)^{a}	5 (2)^{a}	8.6 (0.9)^{ab}	9.4 (0.6)^{b}	9.7 (0.6)^{b}	10 (2)^{b}
Dissolved								
As	0.05	<0.05	1.7 (0.4)^{a}	3 (1)^{ab}	5.9 (0.7)^{abc}	7 (1)^{bc}	6.9 (1.0)^{c}	7.3 (0.8)^{c}
Cd	0.05	<0.05	<0.05	<0.05	<0.05	<0.05	<0.05	<0.05
Cu	0.05	0.8	31 (3)^{a}	36 (6)^{ab}	45.1 (0.4)^{b}	44 (3)^{b}	45.1 (0.8)^{b}	44 (5)^{b}
Pb	0.079	<0.079	<0.079	<0.079	<0.079	<0.079	<0.079	<0.079
Ni	0.20	0.90	13 (2)^{a}	13.8 (0.6)^{a}	15.8 (0.2)^{ab}	15 (1)^{ab}	16.5 (0.6)^{b}	17 (2)^{b}
Zn	0.19	<DL	2.1 (0.7)^{a}	2.6 (0.6)^{a}	5 (1)^{b}	4.6 (0.8)^{b}	7 (5)^{b}	7 (4)^{b}
Ca	3.4	6.5	94 (20)^{a}	74 (8)^{ab}	55.0 (0.6)^{abc}	47 (2)^{bc}	46 (2)^{c}	44 (2)^{c}
Mg	1.9	18.9	24 (5)^{a}	19 (2)^{a}	14.2 (0.2)^{ab}	12.1 0.3)^{b}	12.6 (0.6)^{b}	12.4 (0.3)^{b}
Na	27	197	181 (10)	183 (9)	167 (6)	169 (7)	177 (9)	167 (6)

Effluent concentrations of metals were not different ($\alpha = 0.05$) for treatments sharing a superscript letter

Table 11. Summary of nutrient and conventional parameters during clean water conditioning of bioretention soil media for influent (clean) water and triplicate effluent samples from each of the five treatment depths plus the clean water control (C18). Values presented are mean (standard deviation).

Analyte	Detection Limit	Effluent Water						
		Influent Water	R6	R9	R12	R15	R18	C18
Nitrates (mg/L)	0.003	BDL	33 (20)^{a}	26 (10)^{a}	9 (4)^{ab}	5.6 (0.4)^{b}	7 (2)^{b}	8 (4)^{b}
Ortho phosphate (mg/L)	0.01	0.032	0.07 (0.03)^{a}	0.15 (0.08)^{a}	0.49 (0.05)^{ab}	0.63 (0.09)^{b}	0.6 (0.1)^{b}	0.63 (0.06)^{b}
DOC (mg/L)	0.08	0.57	38 (7)	40 (4)	47 (0.00)	44 (5)	47 (4)	46 (3)
Alkalinity (as $CaCO_3$	0.3	80	56 (8)^{a}	80 (5)^{ab}	101 (5)^{abc}	118 (13)^{bc}	133 (10)^{c}	120 (10)^{bc}
pH	n.a.	7.70	7.59 (0.07)	7.60 (0.06)	7.66 (0.02)	7.69 (0.03)	7.60 (0.02)	7.62 (0.02)
Conductivity (mS/cm)		1525	1893 (200)	1851 (70)	1717 (70)	1712 (30)	1767 (20)	1783 (70)
Hardness (as $CaCO_3$	calculated	0.094	0.34 (0.07)^{a}	0.26 (0.03)^{ab}	0.196 (0.002)^{abc}	0.167 (0.006)^{bc}	0.166 (0.006)^{c}	0.162 (0.005)^{bc}
TSS (mg/L)	0.5	0.25	7 (2)	11 (6)	20 (4)	18 (2)	19 (2)	18 (2)
Turbidity (NTU)	n.a.	0.07	11 (5)^{a}	24 (1)^{ab}	54 (9)^{b}	52 (6)^{b}	56 (7)^{b}	52 (9)^{b}
Fecal coliform (CFU/100 mL)	n.a.	<2	>1300	>1300	>6000	>6000	>6000	>6000

Note: Treatments with different superscript group labels (a, b, c) show significance at α = 0.05 (Kruskal-Wallis with post-hoc Dunn Test). BDL = Below Detection Limit; NTU = Nephelometric Turbidity Units

3.1.4. Comparison of BSM solid, extractable, and leachable fractions

Concentrations of metals, nutrients and PAHs in the bioretention soil components were compared with concentrations in the acid leachate experiment and the clean water conditioning to better understand leaching potential of the various pollutants during stormwater treatment.

3.1.4.1 Metals

Metal concentrations in the solid BSM, BSM extract, and leachate from clean water conditioning were not consistent. In both the solid BSM and BSM extract, zinc was the most concentrated of the measured metals, with higher concentrations in the compost compared to the sand component. However, in effluents from column conditioning with clean water, copper was detected at the highest concentrations of the measured metals. For each treatment depth, metals were detected in effluent samples in the order of copper > nickel > zinc > arsenic > lead. Aside from zinc, this order corresponds with the order of metal concentrations from individually leached bioretention components, although it differs from the order of metals detected in the solid BSM.

Metal concentrations from the solid BSM versus the BSM extract indicate that zinc was more mobile than the other metals, as it appeared at much higher concentrations in the compost and sand extracts relative to the solid media (Figure 7). This is supported by research indicating that zinc activity in soil is high (Rutkowska et al., 2015). However, concentrations of zinc and other metals in effluents from column conditioning with clean water were less than concentrations measured in the extraction tests (Figure 7). This was expected because there was less contact between water and the BSM components during column conditioning than during the extraction test. Additionally, the water source varied between the two tests. During column conditioning, WSU-Puyallup lab water (pH = 7.70) flowed through the BSM whereas during the extraction test BSM components were completely submerged in and agitated with an acidic extraction fluid (RO water adjusted to a pH of 5.00). The solubility of metals generally increases with decreasing pH. Differences in pH between the solutions used in the extraction test and column conditioning therefore likely explain differences in metal concentrations between the two tests.

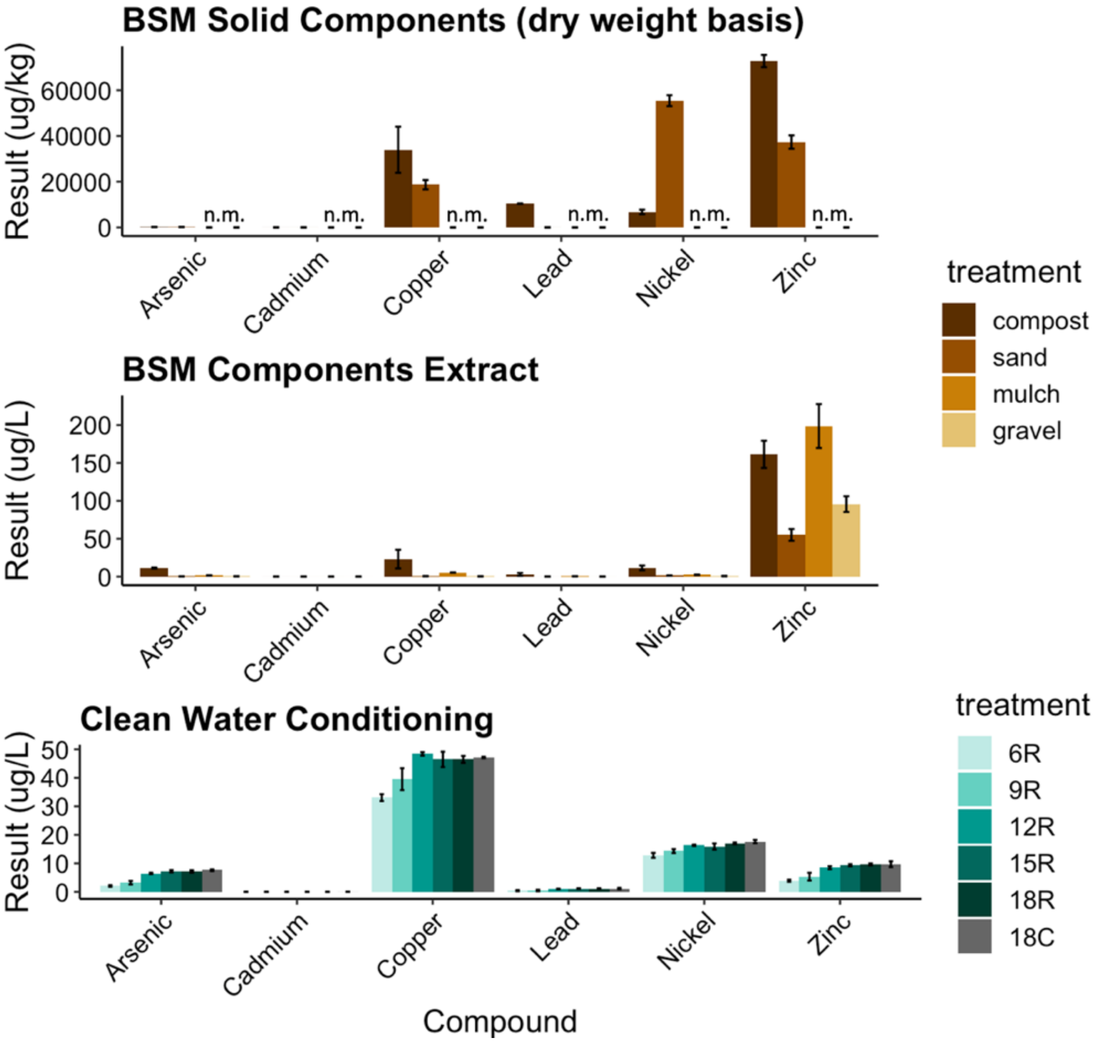


Figure 7. Comparison of metals concentrations in solid bioretention soil media (BSM), BSM leachate, and influent (clean water) and effluent waters during column conditioning. Error bars are one standard error of the mean. n.m. = not measured. R6, R9, R12, R15, and R18 = effluent from the 6”, 9”, 12”, 15”, and 18” treatment depths and C18 = export from the 18” clean water control columns.

Differences in the order of metals leaching between the two tests may also be explained by pH differences. At neutral pH, dissolved metal concentrations in soil water extracts are dominated by DOC-metal complexes. At low pH, free ionic forms of the metals dominate (Reddy et al., 1995). Dissolved organic matter (DOM) can form complexes with metals and increase their mobility (Ashworth & Alloway, 2004; Christensen et al., 1996). Christensen et al. (1996) found that the presence of DOC lowered the adsorption-desorption distribution coefficients (K_d) of the metals cadmium, nickel, and zinc. However, DOM concentration may have less of an effect on the mobility of zinc than on other heavy metals (such as copper and nickel) (Ashworth & Alloway, 2004; Christensen et al., 1996). This may explain why during column conditioning, when the pH was close to neutral, metals other than zinc dominated. During the extraction tests, in which the extraction fluid had a much lower pH, metals would have leached predominantly in free ionic forms and were dominated by zinc, which is considered a highly mobile metal (Rutkowska et al., 2015).

3.1.4.2 Nutrients

Nitrates were more concentrated than orthophosphate in the BSM extracts and effluents from clean water column conditioning (Figure 8). In both BSM extracts and solid components, compost appeared to be the primary source of nitrates in the bioretention columns. Given this, it was surprising to see that nitrates concentrations were significantly greater in effluent from the columns with the lowest sand and compost volumes (columns with 6" and 9" treatment depths) compared to those columns with a greater sand and compost volume (15" and 18" treatment depths). This observation could be the result of greater denitrification rates in the deeper columns. A deeper saturation zone, which would encourage denitrification, may have been formed in the larger columns because of the greater volume of water flushed through these columns.

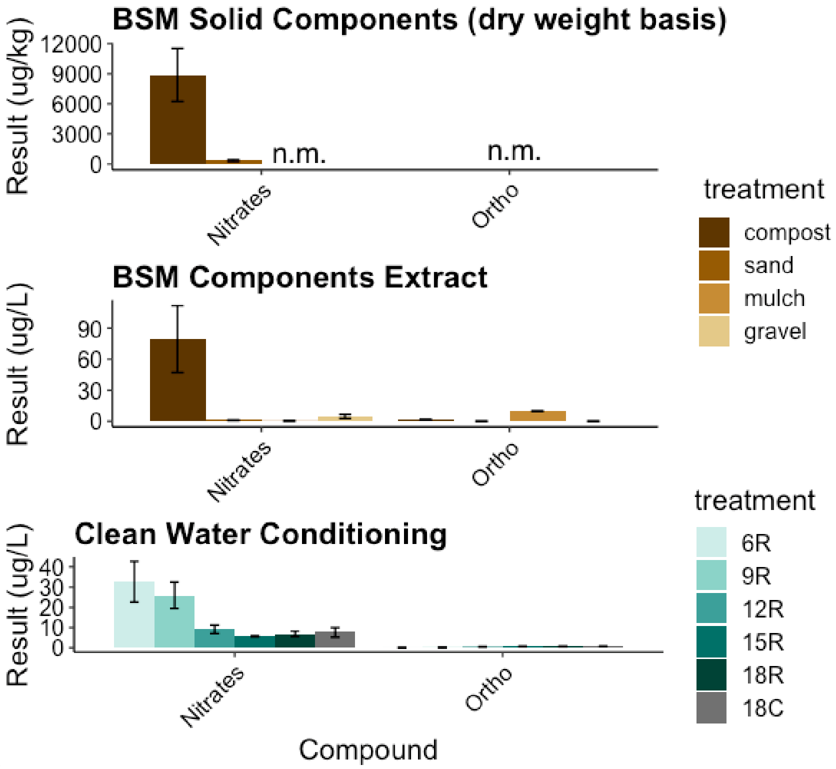


Figure 8. Comparison of nutrient concentrations in bioretention soil media (BSM), BSM leachate, and influent (clean water) and effluent waters during column conditioning. Error bars are one standard error of the mean. n.m. = not measured. R6, R9, R12, R15, and R18 = effluent from the 6", 9", 12", 15", and 18" treatment depths and C18 = export from the 18" clean water control columns.

3.1.4.3 PAHs

Comparing PAH ratios by ring number in the solid compost and sand with that in the components extract (Figure 9), we see that LMW PAHs were more likely to be extracted into water than the HMW PAHs. Whereas four-ring PAHs dominated in the solid samples, three-ring PAHs were most dominant in the extracts of compost and sand.

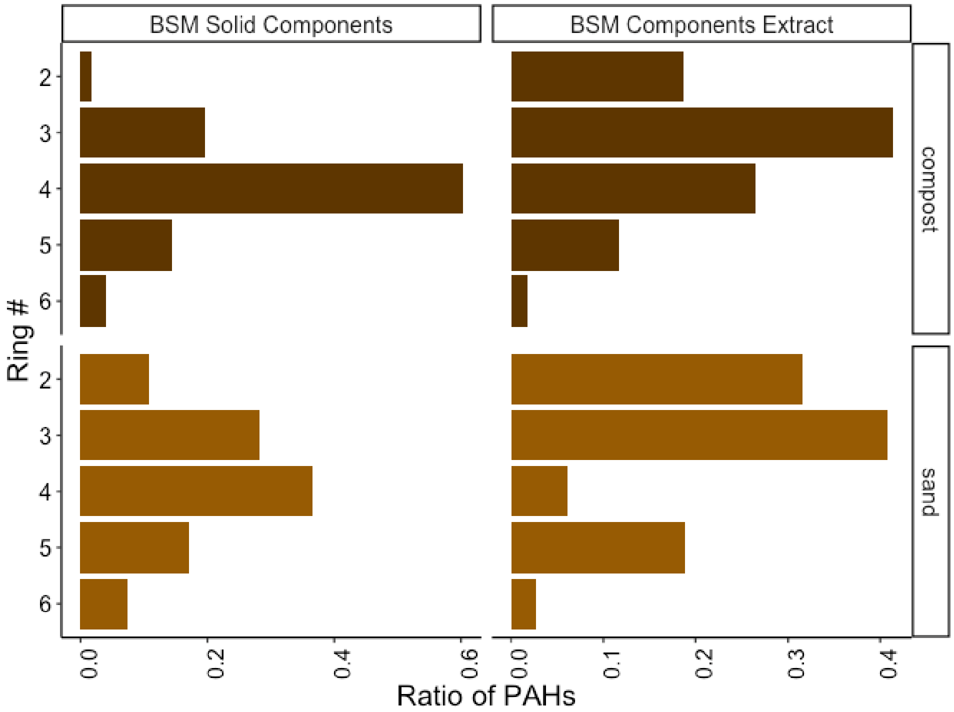


Figure 9. Comparison of PAHs (summed by ring number) in solid and extracted samples of sand and compost as a ratio of total PAHs summed by treatment (compost vs. sand) and fraction (solid vs. extracted into water).

3.1.5. Toxicity of bioretention soil media components

We tested the potential for each bioretention component to cause toxicity to zebrafish embryos. Embryo survival was high in extracts of compost, sand, and gravel, with values of 100% survival for compost and sand, 97% for gravel, and 100% for the embryo rearing medium control after a 48-hour exposure (Table 12). Embryos exposed to mulch leachate all died within 24 hours of exposure. Three outliers (two control and one compost treatment) were removed from the dataset due to extreme developmental abnormalities. Length, pericardial area (PCA), or periventral area (PVA) did not differ significantly among treatment groups (Figure 10). However, the eye area of embryos reared in compost leachate were significantly reduced compared to controls (Kruskal-Wallis with post-hoc Dunn’s test; p=0.035), although this small difference from controls (4%) may not be biologically relevant.

Table 12. Summary of effects of bioretention component extracts on zebrafish development at 48 hpf. Values presented are mean (standard deviation).

Treatment	Mortality Rate	Length (mm)	Eye Area (mm^2)	PCA (mm^2)	PVA (mm^2)
Control	0%	2.87 (0.06)	0.047 (0.003)	0.019 (0.002)	0.022 (0.003)
Mulch	100%	na	na	na	na
Gravel	3%	2.86 (0.08)	0.047 (0.002)	0.019 (0.003)	0.023 (0.004)
Sand	0%	2.90 (0.10)	0.046 (0.003)	0.019 (0.003)	0.022 (0.003)
Compost	0%	2.89 (0.08)	0.045 (0.003)*	0.019 (0.003)	0.023 (0.003)

* Significantly different from control ($p < 0.05$)

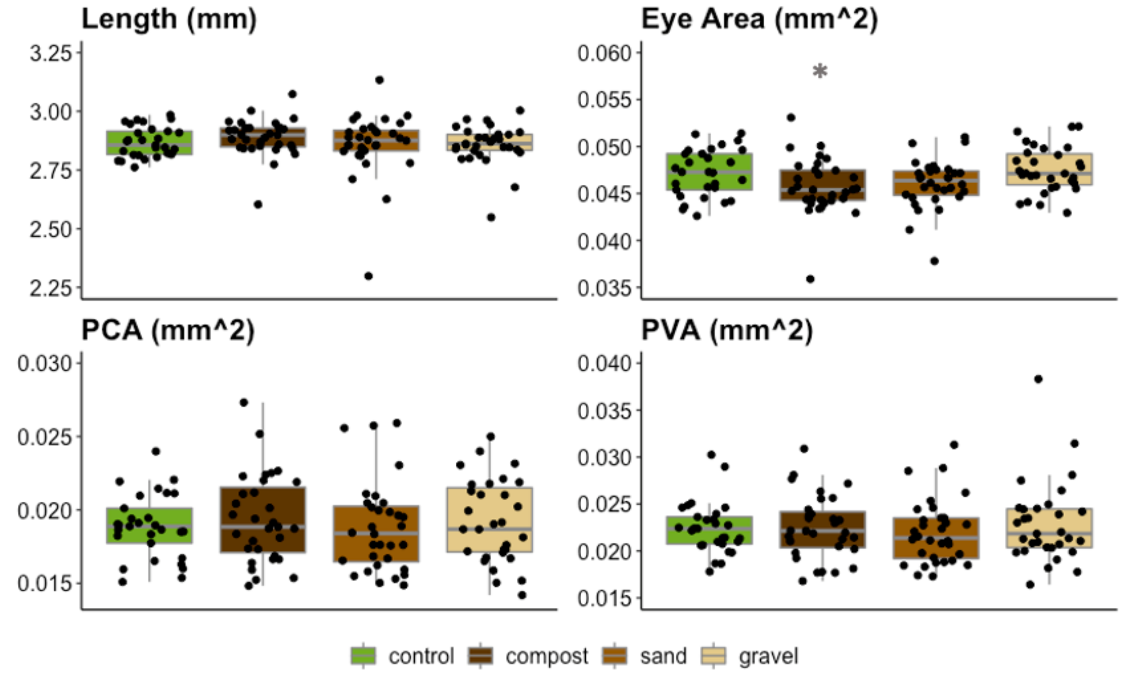


Figure 10. Zebrafish morphometrics for 48 h exposure to fish rearing water (control) and leachates of BSM components. The mulch leachate caused 100% mortality. PVA = periventral area and PCA = pericardial area. * denotes significant difference from control.

The lack of acute lethal or sublethal effects for leachates from compost, sand, and gravel was expected. Clean water effluent from previous conditioning of experimental bioretention columns containing these three components was similarly not acutely toxic to zebrafish embryos (McIntyre et al., 2020; McIntyre et al., 2016). The acute mortality caused by the mulch leachate was not completely unexpected. The mulch used for this study was arborist mulch, comprised of chipped branches and trees from arborist activities. Wood chips can leach highly toxic compounds into water, resulting in acute mortality in fish and other aquatic organisms

(Rex et al., 2016; Taylor et al., 1996; Taylor & Carmichael, 2003). Resin acids are considered some of the most abundant sources of toxic compounds naturally present in wood (Ali & Sreekrishnan, 2001). Additionally, several studies have indicated the toxic potential of phenanthrene and its alkyl homologs (especially retene) to fish (Brinkworth et al., 2003; Hawkins et al., 2002; Mu et al., 2014). Brinkworth et al. (2003), for example, observed hemorrhaging, yolk-sac edema, and mortality in early life-stage rainbow trout exposed to retene. Phenanthrene was by far the most abundant PAH measured in mulch leachate and was several orders of magnitude higher in the mulch leachate than in the other bioretention components. Among the innovations explored by the pulping and forestry industries to address toxicity of runoff from wood include soil infiltration (Hedmark & Scholz, 2008) and pre-treatment of wood chips with fungi (Dorado et al., 2000; Hedmark & Scholz, 2008; Wang et al., 1995), both of which tend to be involved in bioretention applications.

3.2. Stormwater treatment

3.2.1. Water quality of treated stormwater

Contaminants in stormwater were measured in influent and effluent waters including DOC, TSS, total and dissolved metals (arsenic, cadmium, copper, nickel, lead, zinc), bacteria (fecal coliform), nutrients (nitrates, orthophosphate) and a suite of PAHs (Figure 11). The conventional metals calcium, magnesium, and sodium were also measured and are summarized in Appendix 2 but are not included in the analysis. 6PPD-quinone was measured for two events, summarized in Section 3.2.1.1.

Urban stormwater runoff collected for this study contained a suite of contaminants typical of roadway runoff (Kayhanian et al., 2012; McIntyre et al., 2014; Shinya et al., 2000). Bioretention treatment of stormwater significantly improved water quality by removing metals, suspended solids, and aromatic hydrocarbons from influent stormwater. For the standard depth bioretention treatment (R18), median concentration reduction was >90% for TSS, TPAH, total Pb and total Zn (Table 13). These results are comparable with previous bioretention studies that reported removals of lead and zinc reaching close to 100% (Davis et al., 2003), TSS at 29 to >96% (Hsieh & Davis, 2005), and hydrocarbon contaminants at 80 to 95% (Hong et al., 2006).

Median concentration reductions of more than 50% were achieved for dissolved Zn (80%), total copper (73%), total nickel (59%). Poorer removal efficiencies (<50%) were achieved for dissolved copper (37%), dissolved arsenic (28%), and dissolved Ni (23%). Finally, DOC and nutrients were primarily exported from the bioretention media (Figure 11), with negative median removal efficiencies and median net exported concentrations of 0.26 mg/L for DOC, 0.46 mg/L for nitrates, and 0.05 mg/L for orthophosphate (Table 13).

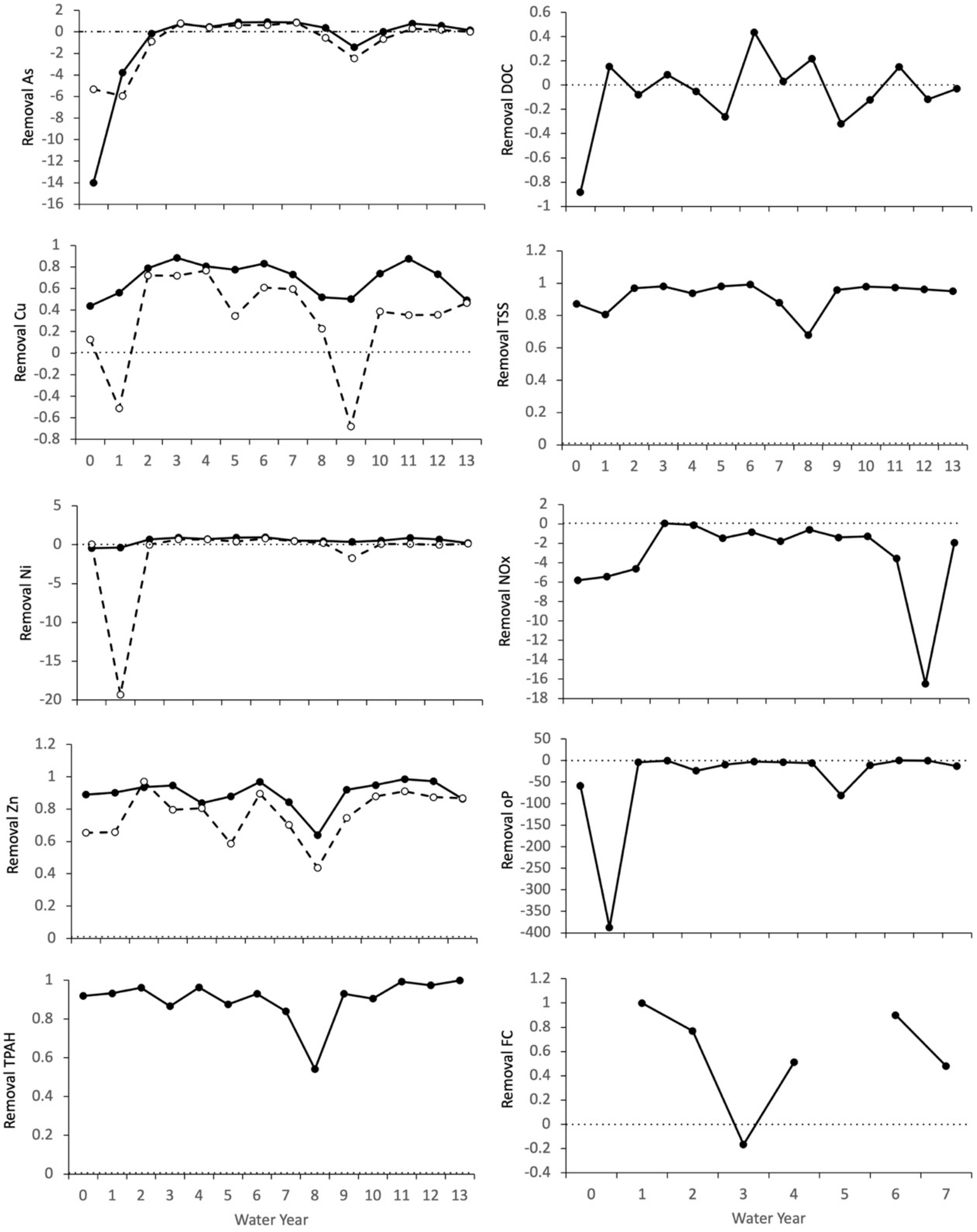


Figure 11. Proportional removal of contaminants from influent stormwater runoff with the standard depth (18”) of bioretention media over 13 water years. For metals, closed symbols are total and open symbols are dissolved. Dotted horizontal lines highlight the division between net removal and net export.

Table 13. Percent removal (%) of compounds from stormwater treated in the standard depth (18”) of bioretention media (R18) averaged across water years for analytes with >40% detection in influent stormwater. DOC and nutrients had high rates of export in effluent and are therefore additionally shown as average net concentration exported with effluent waters.

Analyte	Median (Min – Max)	Influent Concentration (Min-Max)
Percent Removal per Event		
TSS	96 (68 – 99)	47 (10–126) mg/L
TPAH	93 (54 – 100)	0.49 (0.05–1.36) µg/L
Pb	91 (64 – 99)	3.4 (0.3–7.5) µg/L
Zn	91 (64 – 98)	109 (50–198) µg/L
dZn	80 (44 – 97)	28.5 (8.8–73.4) µg/L
Cu	73 (49 – 88)	35 (16–86) µg/L
Ni	59 (22 – 94)	3.2 (1.2–8.4) µg/L
FC	48 (-4633 – 90)	9000 (0–69000) CFU/100 mL
As	47 (-143 – 90)	1.2 (0.1–3.0) µg/L
dCu	37 (-68 – 77)	15 (4–70) µg/L
dAs	28 (-247 – 85)	0.6 (0.1–2.7) µg/L
dNi	23 (0 – 79)	1.5 (0.1–6.9) µg/L
DOC	-4 (-32 – 43)	18.6 (3.2–90.0) mg/L
Nox	-162 (-1648 – 6)	1.09 (0.06–7.22) mg/L
oP	-429 (-2367 – 42)	0.021 (0.004–0.073) mg/L
Net Leaching per Event (mg/L)		
DOC	0.26 (-11.90 – 4.63)	18.6 (3.2–90.0) mg/L
NOx	0.46 (-0.20 – 3.56)	1.09 (0.06–7.22) mg/L
oP	0.05 (-0.03 – 0.41)	0.021 (0.004–0.073) mg/L

Through effluent from the clean water control bioretention media (C18) we were able to show that the media was a sustained source of various compounds to effluent waters throughout the study. Following higher initial flushing during WY0 and WY1 (Figure 12), sustained leaching resulted in average effluent concentrations of 0.02 µg/L TPAH to 3.75 mg/L DOC across the remaining water years (Table 14). When contaminant concentrations in influent runoff are low, leaching from the bioretention media becomes an important source to effluent waters. By comparing effluent concentrations from C18 to those treating runoff in the same depth of bioretention media (R18), we estimated the contribution of media leaching to effluent concentrations from R18. Across WY2-13, the media contributed on average 67-79% of Ni and As in effluents from R18, 55% of TPAHs, and 26-45% of Cu and Zn (Table 14).

Nutrients and some metals exported into the effluent from the media is consistent with other studies of bioretention systems with a high compost volume fraction (Hatt et al., 2008; Mullane et al., 2015; Paus et al., 2014; Taylor et al., 2018). A considerable decrease in nutrient concentrations within the first several storms followed by a low level of sustained leaching was expected (Mullane et al., 2015; Taylor et al., 2018). Orthophosphate in effluent was primarily derived from the bioretention media as it was below detection limits for 57% of influent stormwater samples but measurable in all stormwater effluent samples. Ortho-phosphate was the only compound to be measured at higher concentrations in effluent from C18 than from R18 (2-fold) and also the only to show a steady decline in effluents from C18 across the study (Figure 12).

Table 14. Initial concentrations (WY0) in effluent from the control media (C18) receiving only clean water show many analytes initially flushed from the media compared with the sustained concentration (averaged for WY2-13). The ratio of concentrations in C18 and R18 estimates the percent of effluent from media treating runoff derived from the media itself. Errors are one standard deviation. FC not included because detected in <10% of C18 effluent.

Analyte	WY0 *µg/L*	WY2-13 Average (SD) *µg/L*	C18/R18 (SD) %
dAs	0.9	0.4 (0.3)	79 (46)
As	1.7	0.4 (0.3)	75 (31)
Ni	3.4	0.5 (0.2)	68 (33)
dNi	3.0	0.4 (0.2)	67 (32)
TPAH	0.025	0.020 (0.013)	55 (31)
dCu	10.0	2.4 (1.2)	45 (31)
Cu	12.2	3.5 (1.3)	45 (21)
dZn	1.8	1.4 (1.1)	35 (34)
Zn	2.3	1.6 (1.4)	26 (28)
	mg/L	*mg/L*	
oP	0.4	0.15 (0.05)	210 (179)
TSS	3.3	0.86 (0.58)	61 (44)
NOx	3.6	0.62 (0.61)	52 (31)
DOC	11.1	3.75 (6.68)	30 (18)

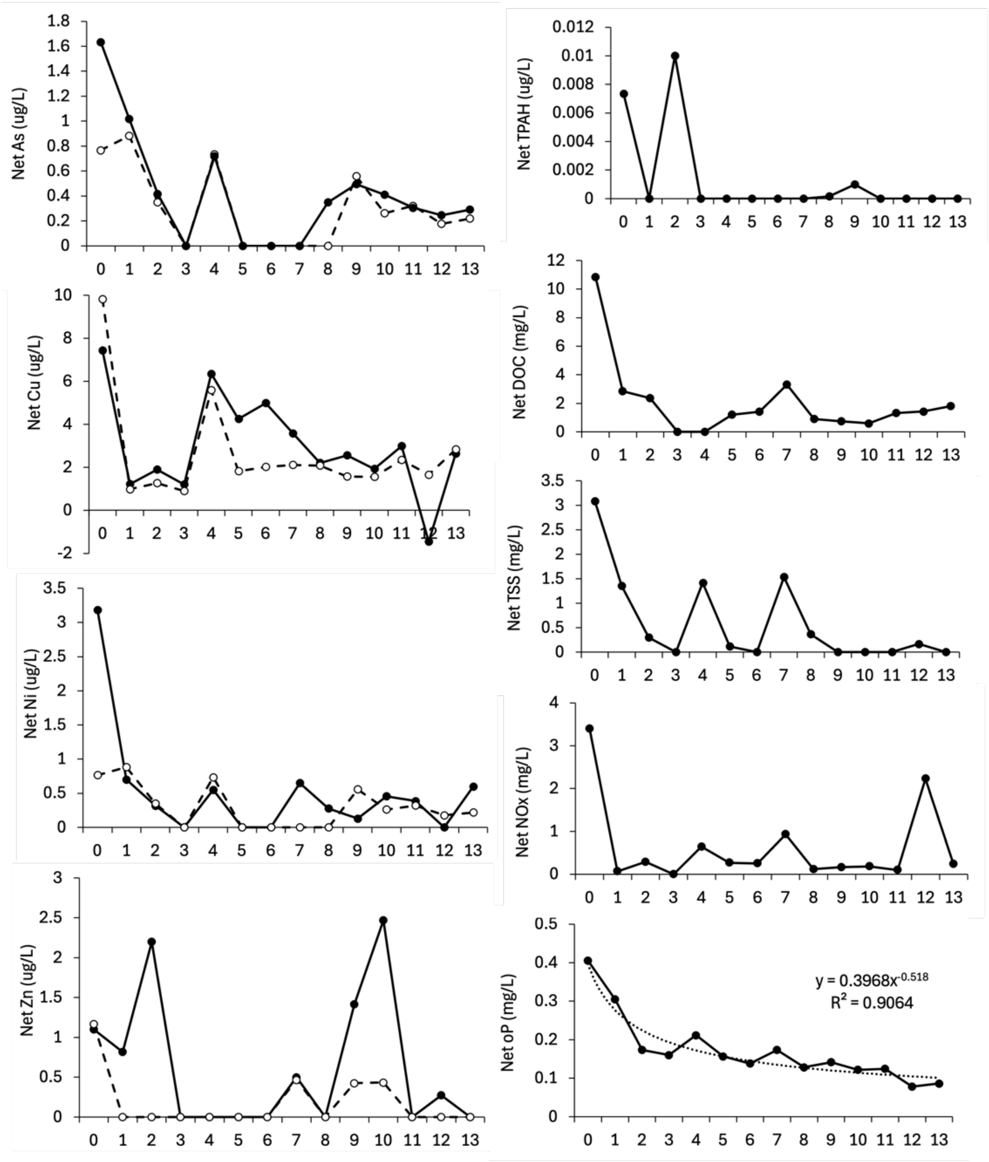


Figure 12. Average net leaching from bioretention media receiving clean control water (C18) across the 13 water years (concentration in C18 effluent minus C influent). For metals, closed symbols are total and open symbols are dissolved. Fecal coliform was detected in <10% of C18 effluent and therefore is not included.

Changes in effluent concentrations among the treatment depths and over time (Figure 13) were explored with a linear mixed effect model (Table 15). Two analytes showed a trend with time; TPAH and orthophosphate concentrations showed a slight but significant decrease over time. A lack of significance for all other parameters indicates that treatment effectiveness did not decline for any pollutants over the 13 simulated water years. Bioretention media depth affected release of several stormwater pollutants into effluent waters. Significantly more copper (total and dissolved) was released from the 6” depth than from the 12” or 18” depths. In contrast, the 6” depth released less arsenic (total and dissolved) and total nickel. Concentrations of nutrients, TSS and DOC were significantly higher from deeper media (R18>R12>R6). The rate of decline of orthophosphate in effluents from the 6” media were slightly reduced compared with the deeper media. Zinc (total and dissolved), total Pb, and dissolved nickel concentrations in effluent were independent of media depth. Average performance for each depth is also summarized in Table 16.

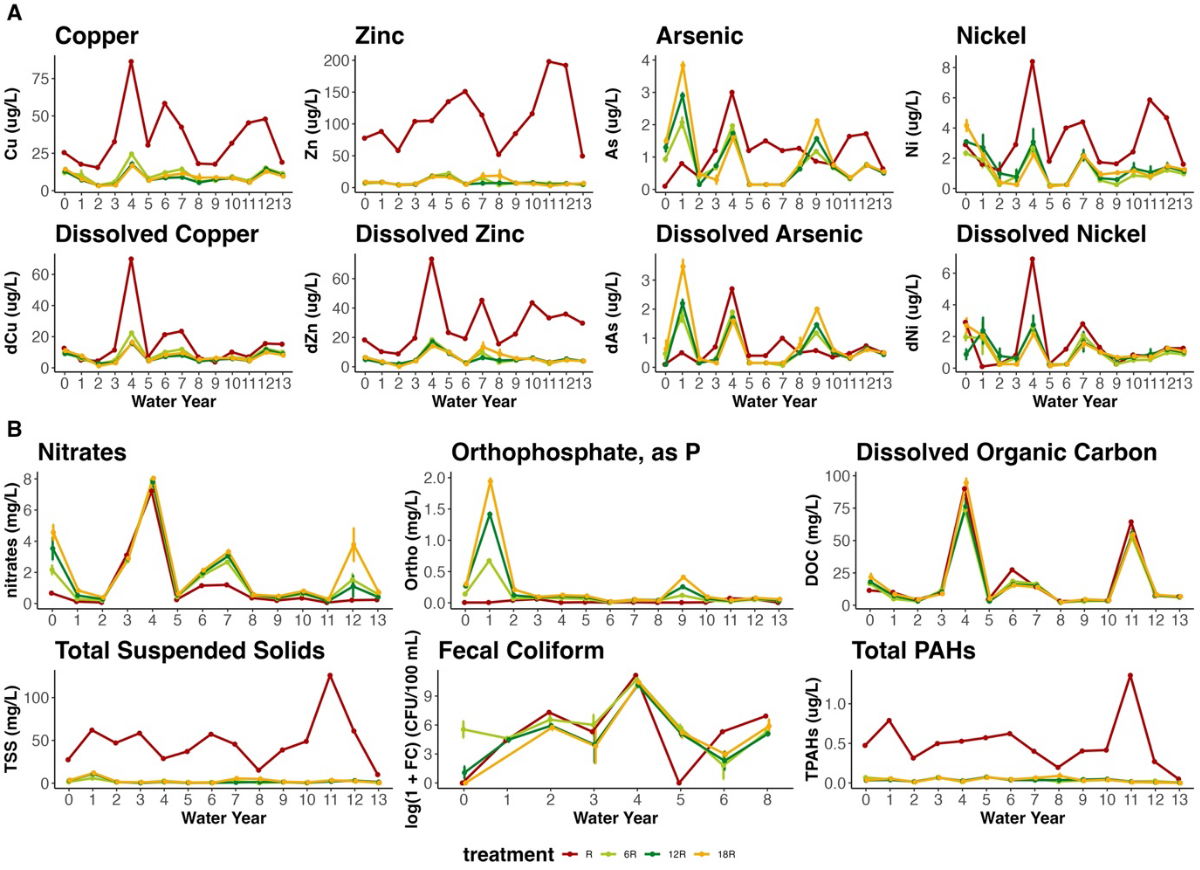


Figure 13. Mean effluent concentrations (SE) for A) metals and B) non-metals for each sampling event and treatment across 13 accelerated water years. R = stormwater influent, R6, R12, and R18 = treated-runoff effluent from the 6”, 12”, and 18” treatment depths

Table 15 Results of the linear mixed effects model for effluent concentrations for the three retained bioretention treatments across all 13 sampling events relative to the end of the study. Bold p values were statistically significant predictor variables for depth difference compared with 18” (R18). Significant p-values for R18:WY indicate a temporal trend in concentration for R18. Significant p-values for R12 or R6 indicate where concentrations, or slope with time (:WY) are different from that of R18. E.g.: Effluent concentrations of ortho-P were significantly lower for R12 and R6 than for R18 and the significant decrease in [ortho-P] in R18 over time was reduced in R6. Italicized p-values (between 0.1 and 0.05) indicate a possible trend.

Variable	Factor	Estimate	Std. Error	df	t value	p
Cu	R12	-0.0445	0.0374	108	-1.19	0.236
Cu	R6	0.11	0.0374	108	2.93	**0.004**
Cu	R18:Age (ref)	0.00014	0.0002	12.9	0.578	0.573
Cu	R12:Age	4.24e-05	7.48e-05	108	0.567	0.572
Cu	R6:Age	-2.57e-05	7.48e-05	108	-0.344	0.731
Zn	R12	-0.0466	0.0832	108	-0.56	0.577
Zn	R6	0.0216	0.0832	108	0.26	0.796
Zn	R18:Age (ref)	-0.00028	0.0002	16.7	-1.16	0.263
Zn	R12:Age	4.91e-05	0.00017	108	0.295	0.768
Zn	R6:Age	2.55e-05	0.00017	108	0.153	0.879
As	R12	-0.0395	0.0225	108	-1.76	*0.082*
As	R6	-0.0576	0.0225	108	-2.56	**0.012**
As	R18:Age (ref)	-0.0002	0.0002	12.4	-0.941	0.365
As	R12:Age	1.23e-05	4.5e-05	108	0.274	0.784
As	R6:Age	5.12e-05	4.5e-05	108	1.14	0.257
Ni	R12	0.0054	0.0448	108	0.12	0.905
Ni	R6	-0.101	0.0448	108	-2.25	**0.027**
Ni	R18:Age (ref)	-4.85e-05	0.0002	13.3	-0.21	0.837
Ni	R12:Age	-9.11e-05	8.97e-05	108	-1.02	0.312
Ni	R6:Age	-6.09e-05	8.97e-05	108	-0.679	0.499
dCu	R12	-0.00952	0.0354	108	-0.269	0.788
dCu	R6	0.12	0.0354	108	3.4	**<0.001**
dCu	R18:Age (ref)	0.0002	0.0003	12.6	0.613	0.551
dCu	R12:Age	-7.36e-06	7.08e-05	108	-0.104	0.917
dCu	R6:Age	1.47e-05	7.08e-05	108	0.208	0.836
dZn	R12	-0.0224	0.0661	108	-0.34	0.735
dZn	R6	-0.0303	0.0661	108	-0.459	0.647
dZn	R18:Age (ref)	5e-05	0.0003	13.7	0.165	0.872
dZn	R12:Age	-6.29e-05	0.0001	108	-0.476	0.635
dZn	R6:Age	-0.0002	0.0001	108	-1.48	0.141
dAs	R12	-0.0694	0.025	108	-2.77	**0.007**
dAs	R6	-0.0732	0.025	108	-2.92	**0.004**
dAs	R18:Age (ref)	-0.0001	0.0002	12.5	-0.505	0.622
dAs	R12:Age	9.21e-05	5.01e-05	108	1.84	*0.069*
dAs	R6:Age	5.43e-05	5.01e-05	108	1.08	0.281
dNi	R12	-0.0143	0.0476	108	-0.3	0.765
dNi	R6	-0.0483	0.0476	108	-1.01	0.313

Variable	Factor	Estimate	Std. Error	df	t value	p
dNi	R18:WY (ref)	-4.82e-05	0.0002	14.1	-0.242	0.812
dNi	R12:WY	1.39e-05	9.54e-05	108	0.146	0.884
dNi	R6:WY	-9.47e-05	9.54e-05	108	-0.993	0.323
Pb	R12	-0.0166	0.0241	108	-0.688	0.493
Pb	R6	0.00528	0.0241	108	0.219	0.827
Pb	R18:Age (ref)	-5.39e-05	4.41e-05	31.9	-1.22	0.230
Pb	R12:Age	4.61e-05	4.83e-05	108	0.954	0.342
Pb	R6:Age	-2.46e-05	4.83e-05	108	-0.509	0.612
NOx	R12	-0.161	0.0354	108	-4.54	**<0.001**
NOx	R6	-0.173	0.0354	108	-4.89	**<0.001**
NOx	R18:Age (ref)	-0.0003	0.0003	12.4	-1.08	0.299
NOx	R12:Age	-0.000176	7.09e-05	108	-2.48	**0.015**
NOx	R6:Age	3.31e-05	7.09e-05	108	0.467	0.641
Ortho-P	R12	-0.0335	0.0135	108	-2.48	**0.015**
Ortho-P	R6	-0.0815	0.0135	108	-6.02	**<0.001**
Ortho-P	R18:Age (ref)	-0.0002	0.0001	12.5	-2.08	**0.059**
Ortho-P	R12:Age	3.41e-05	2.71e-05	108	1.26	0.212
Ortho-P	R6:Age	0.000115	2.71e-05	108	4.25	**<0.001**
TSS	R12	-0.202	0.0825	108	-2.45	**0.016**
TSS	R6	-0.218	0.0825	108	-2.64	**0.010**
TSS	R18:Age (ref)	-0.0003	0.0003	14.9	-1.1	0.290
TSS	R12:Age	0.0001	0.0002	108	0.838	0.404
TSS	R6:Age	0.0001	0.0002	108	0.862	0.390
DOC	R12	-0.0712	0.0304	108	-2.35	**0.021**
DOC	R6	-0.107	0.0304	108	-3.51	**<0.001**
DOC	R18:Age (ref)	-0.0002	0.0005	12.1	-0.425	0.679
DOC	R12:Age	8.68e-05	6.08e-05	108	1.43	0.156
DOC	R6:Age	0.0001	6.08e-05	108	1.73	*0.086*
TPAHs	R12	-0.0036	0.00368	108	-0.969	0.335
TPAHs	R6	-0.0018	0.00368	108	-0.495	0.621
TPAHs	R18:Age (ref)	-2.35e-05	1.06e-05	17	-2.22	**0.040**
TPAHs	R12:Age	4.48e-06	7.38e-06	108	0.607	0.545
TPAHs	R6:Age	3.09e-07	7.38e-06	108	0.0419	0.967

Table 16. Summary of overall performance by compound and treatment depth. Influent concentrations (R) and percent removal are presented as mean (SE) across 13 sampled events. R6, R12, and R18 = treated-runoff effluent from the 6”, 12”, and 18” treatment depths. Different letters in a row indicate statistically distinct removal rate.

	R	R6				R12				R18			
		Mean	Min	Max	Removal (%)	Mean	Min	Max	Removal (%)	Mean	Min	Max	Removal (%)
Cu	38 (5)	10.5	3.5	25.5	66.6^{a} (3.8)	8.7	2.9	21.2	71.4^{b} (3.7)	9.2	3.1	17	69.1^{ab} (4.1)
Zn	103 (7)	8.9	2.9	29.3	91.3 (1.2)	8.0	2.9	19.7	91.8 (1.2)	9.3	0.095	34	89.4 (2.3)
As	1.1 (0.2)	0.8	0.15	2.4	-33.6 (63.9)	0.8	0.15	3	-65.5 (90.7)	1.0	0.15	4	-97.6 (106)
Ni	3.3 (0.5)	1.1	0.25	2.9	62.3^{a} (8.1)	1.4	0.15	4.9	47.5^{b} (11.7)	1.3	0.15	5	49.8^{ab} (11.8)
dCu	19 (5)	8.4	2.4	23.6	29 (7.5)	7	2.5	19.7	34.6 (9.5)	7.3	0.4	16.6	31.9 (11.6)
dZn	25 (5)	5.8	1.46	18.9	78.7 (2.6)	5.5	1.7	17.9	79.4 (2.1)	6.1	0.15	18.9	77 (4)
dAs	0.7 (0.2)	0.6	0.025	2	-28.8 (35)	0.6	0.1	2.5	-12 (29.7)	0.8	0.1	3.9	-86.4 (59)
dNi	1.7 (0.5)	1	0.25	4	-102 (131)	1.1	0.15	4.1	-159 (163)	1.1	0.15	3.7	-124 (140)
NOx	1.8 (0.5)	1.6	0.08	8.13	-142^{a} (41.4)	1.7	0.05	7.86	-160^{ab} (37.9)	2.1	0.16	8.08	-324^{b} (114)
oP	0.024 (0.005)	0.1	0.005	0.7	-1595 (935)	0.2	0.01	1.48	-3178 (1983)	0.26	0.02	2.02	-4315 (2727)
TSS	45 (3)	1.7	0.25	7	96 (0.7)	2.1	0.25	11	94.6 (1.5)	2.9	0.25	14	92.3 (2.4)
DOC	21 (7)	16	2.2	74.4	9.7^{a} (6.6)	16.4	2	90.3	5.5^{ab} (6.7)	18.1	2.1	102	-5.7^{b} (8.2)
TPAH	0.54 (0.03)	0.04	0.003	0.104	91.5 (1.1)	0.03	0	0.078	91.7 (1.3)	0.04	0	0.15	89.9 (3)
FC	9001 (8573)	7191	0.5	69000	-1982 (1290)	3694	0.5	36000	-404 (443)	5006	0.5	39000	-626 (625)

3.2.1.1 6PPD-quinone removal

Analysis of 6PPD-quinone was possible on samples from WY8 and WY13. Influent stormwater runoff contained 6PPD-quinone at 188 ng/L for WY8 and 270 ng/L for WY13. During WY8 runoff was observed exceeding the ponding space of various columns containing the bioretention media. Stormwater was allowed to overflow and was collected along with the treated effluent water to simulate untreated runoff flowing out of a clogged treatment system. In replicates where effluent and overflow was combined, 6PPD-quinone was present at higher concentrations (Table 17), reflecting the loss of treatment for the overflow water. In effluent from treatments without any overflow removal was 92-97%. In treatments with overflow, concentration of 6PPD-quinone was reduced to varying amounts, resulting in average removal of 89% for R12 in which one column overflowed and 57% in R18 for which two of the three replicate columns overflowed.

Table 17. Quantification of 6PPD-quinone (ng/L) in influent waters and effluent from bioretention media dosed with clean water (C) or stormwater runoff (R) at either standard depth (18”) or two experimental shallower depths (6”, 12”).

WY	Treatment	Replicate			Mean	Removal
		1	2	3	(SD)	(%)
8	C	<3	n.a.	n.a.	<3	
	C18	<3	<3	<3	<3	
	R	188			188	
	R6	10.6	4.81	5.9	7.1 (3.1)	96%
	R12	3.6	53.7*	3.5	20.3 (29.0)	89%
	R18	160^{o}	78.5*	<3	79.9 (79.4)	57%
13	C	<3	n.a.	n.a.	<3	
	C18	<3	<3	<3	<3	
	R	270			270	
	R6	5.8	10.0	27.0	14.3 (11.2)	92%
	R12	17	5.3	18	13.4 (7.1)	93%
	R18	6.6	6.1	6.1	6.3 (0.3)	97%

*overflow combined with effluent; reduced treatment

3.2.2. Pollutant accumulation in top 1” of bioretention media

The top 1” of the bioretention media was removed during WY9 to help promote infiltration. Homogenized sub-samples were assessed for porosity and concentrations of metals and TPAH. Porosity was significantly higher in the removed BSM from R18 than from R12 or R6. Average porosity was 13.76 mL (SD = 0.38) for R18, 12.39 mL (0.47) for R12 and 12.47 mL (0.56) for R6. Porosity for the control media (C18) was much more variable (13.19 mL (1.45)) and therefore not significantly different than any of the media receiving runoff. Metal and TPAH concentrations in the removed BSM were compared with values derived for the media prior to stormwater treatment (Figure 14). Bioretention media showed a net gain of most pollutants from the eight accelerated years of stormwater infiltration. Concentrations appeared higher in the shallower media (R6) than the deeper media (R12, R18), statistically so for zinc and TPAHs. Total PAHs was the only pollutant that was reduced in the media following runoff infiltration — approximately 40-fold lower than before the media began treating runoff. Surface media infiltrating ‘clean’ municipal water (C18) generally contained lower pollutant concentrations than at the start of the study – most notably for TPAHs which were 170-fold lower at WY8. Zinc concentrations in the media were not reduced by clean water infiltration and nickel concentrations were significantly elevated in C18.

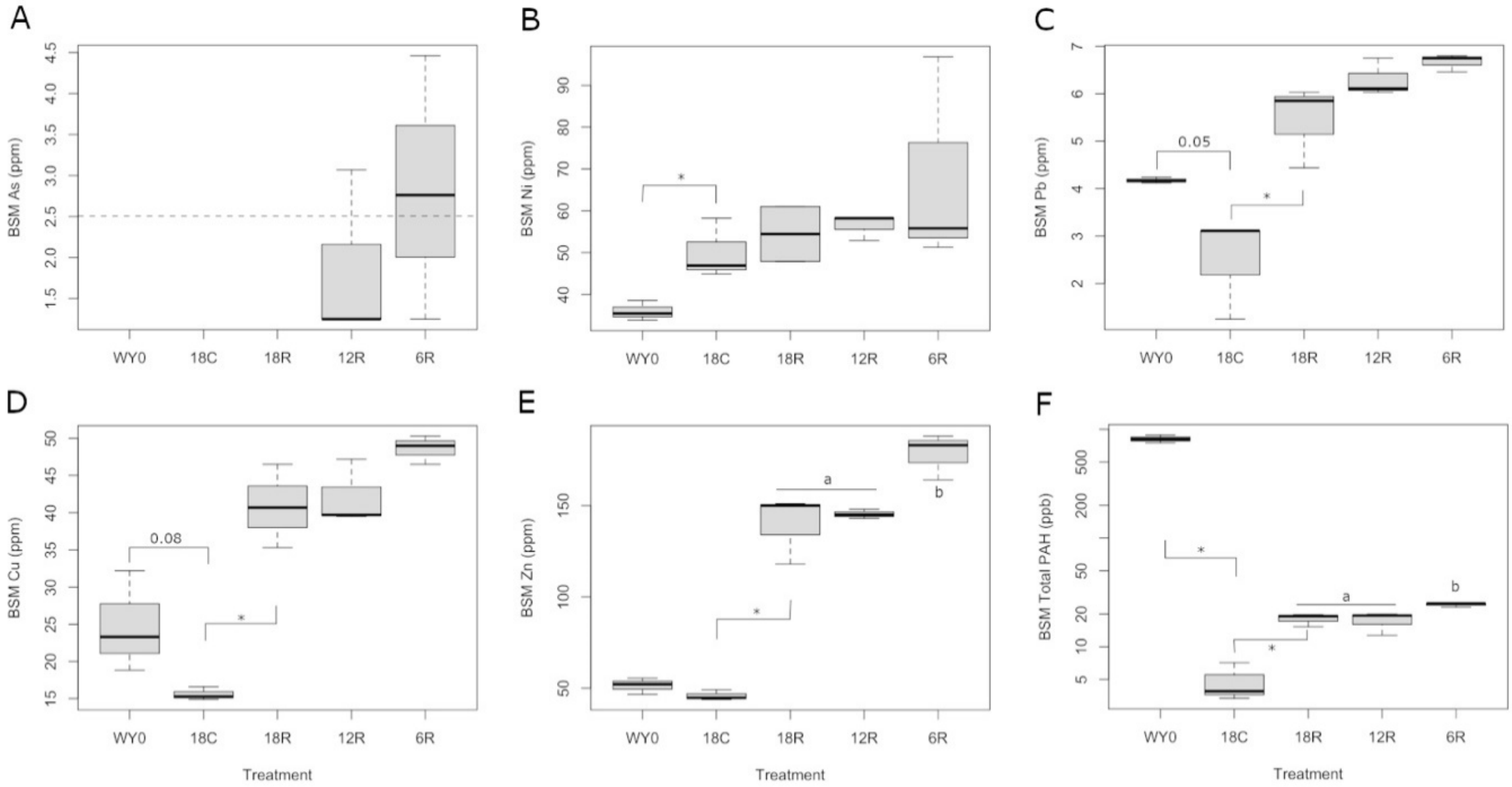


Figure 14. Concentrations of metals and TPAHs in the different depth treatments of bioretention media at WY8 for media treating stormwater runoff (R) or clean water (C) compared the beginning of the study (WY0). Arsenic was below detection limits (dashed line) at WY0 and in C18 and R18 surface soil at WY8.

Whereas metals and PAH concentrations were below screening limits for ecological health at the beginning of the study, accumulations in the top 1” of media by WY8 led to exceedance of various Eco-SSL criteria (Table 18). Copper at 41-49 μg/kg exceeded the avian Eco-SSL and met the mammalian value for the 6” depth treatments, nickel at 50-68 μg/kg exceeded the criteria for plants, and zinc at 140-178 μg/kg exceeded the invertebrate, avian, and mammalian criteria for all depth treatments, and the 6” depth exceeded the criterion for plants.

Table 18. Metal and PAH concentrations (dry weight) in the bioretention soil media at the beginning of the study (WY 0) or after WY8 for three depths of media (6”, 12”, 18”) treating runoff (R) with EPA ecological soil screening levels.

		EPA Ecological Soil Screening Levels				WY0	R6	R12	R18
		Plants	Soil Invertebrates	Avian	Mammalian				
Metals									
As	μg/kg	18	n.a.	43	46	n.m.	2.8	5.0	n.m.
Cu	μg/kg	70	80	28	49	24.8	48.6	42.1	40.8
Pb	μg/kg	120	1700	11	56	4.2	6.7	6.3	5.4
Ni	μg/kg	38	280	210	130	36	68.0	56.5	54.5
Zn	μg/kg	160	120	46	79	51.1	178.3	145.3	139.7
PAHs									
LMW	mg/kg	n.a.	29	n.a.	100	0.54	0.21	0.15	0.17
HMW	mg/kg	n.a.	18	n.a.	1.1	0.28	0.28	0.19	0.19

n.a. = not available; n.m. = not measured

3.2.3. Assessment of toxicity during stormwater treatment

3.2.3.1 Potential for neurotoxicity

Dissolved copper is an important pollutant in urban stormwater runoff, notably originating from vehicle brake pad wear. In the current study, dissolved copper, second to zinc, was detected at the highest concentrations in influent roadway runoff. Detected concentrations of dissolved copper ranged from 4.4–69.9 µg/L (median = 5.2 µg/L) across 14 sampling events. Previous studies have demonstrated the inhibitory effects of dissolved copper to the salmon olfactory nervous systems at low ppb concentrations (Baldwin et al., 2003). Olfaction is important to the survival and migratory success of salmon, allowing the species to receive information about habitat quality, predators, mates, and the animal’s natal stream (Brown & Smith, 1997; Dittman & Quinn, 1996; Hansen et al., 1999; Quinn & Busack, 1985). Influxes of copper to surface waters are, therefore, highly concerning. However, olfactory capacity may be protected in surface water with high DOC content (McIntyre et al., 2008). At ratios of DOC:dCu (ppm:ppb) greater than approximately 1:3 (0.33), copper is not bioavailable to induce neurotoxicity. In the current study, the DOC:dCu ratio ranged from 0.35–9.2 (median = 0.88, SD = 2.3) in influent stormwater runoff. None of the influent stormwater samples across the 14 sampling events had a DOC:dCu ratio below the threshold expected to protect against neurotoxicity (Figure 15). The DOC:dCu ratios in effluent waters (0.41–16, median = 1.2, SD = 3.1) were higher than in influent waters because bioretention treatment generally reduced dCu concentrations. Bioretention treatment, therefore, appeared completely protective of salmon olfactory capacity.

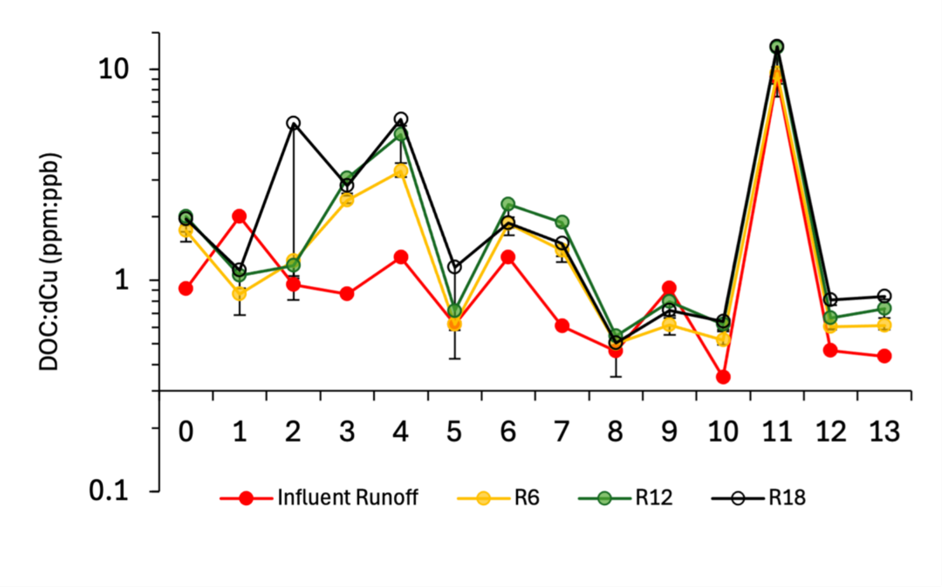


Figure 15. The ratio of average dissolved organic carbon to dissolved copper in influent stormwater and effluent from the three bioretention depths (6", 12", 18"). The x-axis at 0.3 delineates the threshold of expected neurotoxicity (<0.3). Influent values were singular. Effluent values are shown with -1 standard deviation.

3.2.2.2 Zebrafish embryos

Morphometrics were assessed for zebrafish embryos after 48 hours of development in waters from the event ending WY2 and WY3. Survival was 91% in influent stormwater for WY2 and 100% for WY3, was 94–100% for effluent from the across the BSM depths treating runoff and was 100% for the clean water control influent and effluent, which were not significantly different from each other. Some differences in morphometrics among treatments were detected for the individual storm events (Table 19), although differences from controls were not statistically significant. Comparing just improvements from runoff treatment, and across both events, treatment had a measurable impact on eye size (LMM). Eye development was significantly improved from untreated runoff exposure for all of the BSM runoff treatments (p <0.001-0.022), with a moderately higher improvement for R18 (4%) than for R12 or R6 (3%) (Figure 16). This small improvement reflects the lack of strong impact for the runoff itself and may be related to the low concentration of PAHs for these samples (0.3 μg/L for WY2 and 0.5 μg/L for WY3) compared with prior studies of zebrafish morphometrics from runoff exposure. When runoff contained 9 μg/L total PAHs, eye area was reduced by 50% from controls and showed a 43% improvement with bioretention treatment (McIntyre et al., 2014). Potential effects on PCA and PVA were not robust enough to detect with the mixed effects model, indicating that there was no benefit provided by treatment that was common to both events for the cardiac-related endpoints. The low response of morphometrics to influent runoff in this study was the reason that we switched to using qPCR to explore treatment effects for later events.

Table 19. Morphometrics of zebrafish embryos (48 hours post-fertilization) developing in influent or bioretention-treated effluent for the event ending WY2 and WY3. Values presented are mean (SD). R = stormwater influent, and R6, R12, and R18 = treated runoff effluent from the 6”, 12”, and 18” treatment depths.

Treatment	PCA (mm^2)	PVA (mm^2)	Eye Area (mm^2)	Length (mm)
WY2				
R	0.022 (0.004)	0.025 (0.004)	0.052 (0.004)	3.13 (0.12)
R6	0.022 (0.003)	0.026 (0.006)	0.054 (0.003)	3.16 (0.07)
R12	0.023 (0.002)	0.027 (0.004)	0.052 (0.004)	3.13 (0.11)
R18	0.024 (0.004)*	0.030 (0.007)*	0.053 (0.003)	3.18 (0.06)
WY3				
R	0.021 (0.003)	0.023 (0.003)	0.044 (0.002)	3.10 (0.07)
R6	0.019 (0.003)	0.021 (0.003)	0.044 (0.004)	3.10 (0.06)
R12	0.019 (0.002)	0.021 (0.002)*	0.046 (0.002)*	3.11 (0.06)
R18	0.020 (0.003)	0.022 (0.003)	0.046 (0.004)*	3.10 (0.05)

* $p < 0.05$ with R as the reference treatment

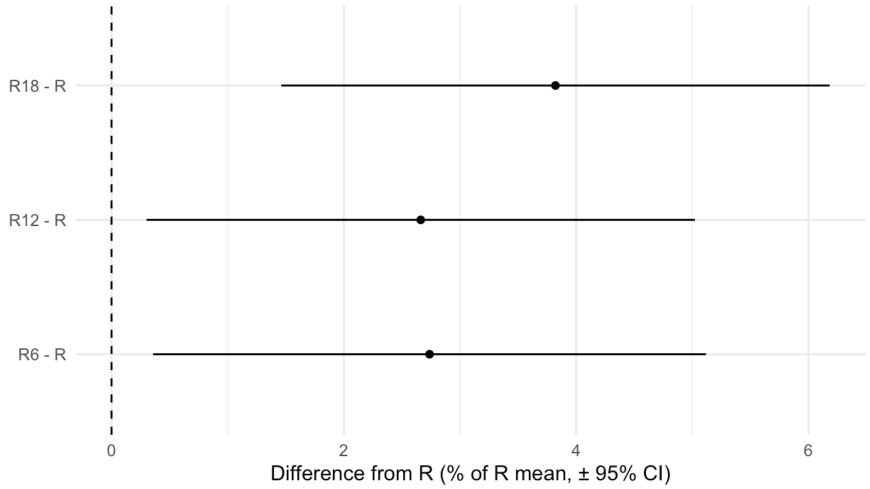


Figure 16. Difference between eye area for zebrafish exposed to effluent from R6, R12, or R18 and untreated effluent R from Dunnett test of the linear mixed effects model including results of morphometrics from WY2 and WY3. Error bars not overlapping zero demonstrate significant improvement in eye area from treatment.

3.2.2.3 Transcriptional response of zebrafish

Zebrafish embryos exposed to influent stormwater runoff showed an increased abundance of mRNA transcript of *cyp1a* (average: 5-fold to 67-fold from controls; Figure 17). This response was expected based on prior studies exposing zebrafish to collected roadway runoff from other sources (McIntyre et al. 2014, Bellevue report). Bioretention treatment of runoff decreased the transcriptional response (LMM: estimated marginal means p <0.001), indicating less impact to the developing fish compared with untreated runoff. Across the water years evaluated, the reduction was 2-fold to 19-fold, with no difference among the bioretention depths (p = 0.22-0.83). Exposure to effluent from C18 did not increase induction of *cyp1a* (p <0.001), indicating that the induction from effluents of bioretention treating runoff was caused by residual contaminants from runoff not from the BSM itself. For three of the five evaluated storm events, there was a tendency for the shallower BSM to reduce induction less than for the deeper BSM (WY5, WY6, WY12).

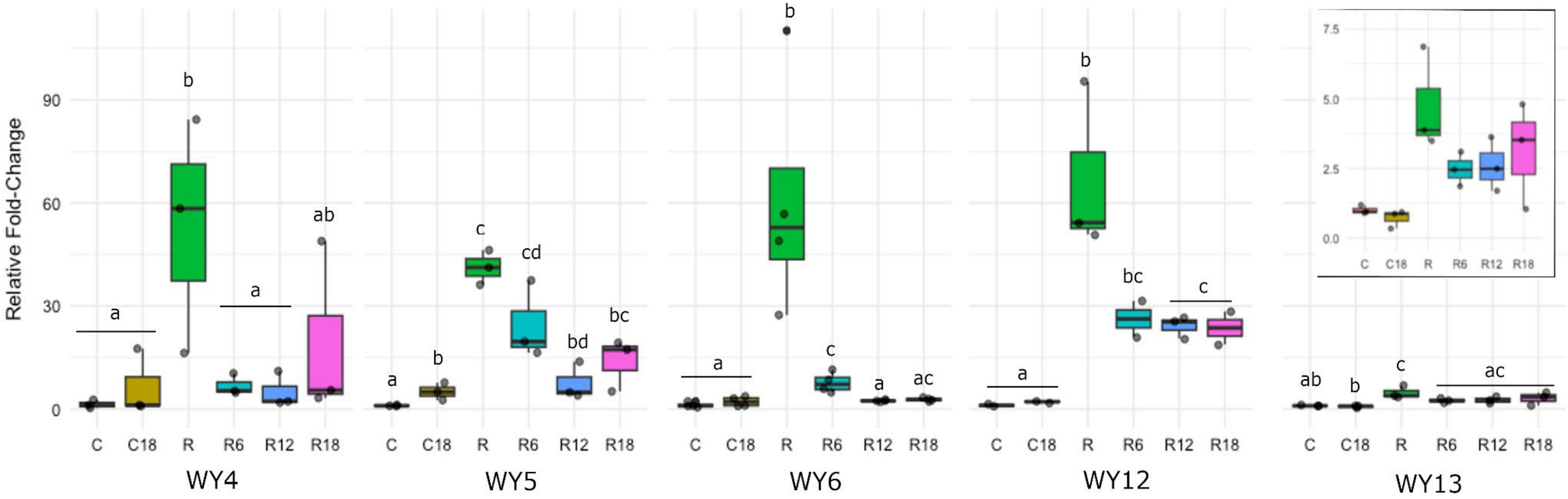


Figure 17. Transcriptional response of cyp1a in zebrafish embryos exposed 48 h to test waters from five sampling events relative to laboratory control water (C). Data points are shown with the boxplots. Treatments sharing a letter were not statistically different ($p > 0.05$). Data for WY13 are additionally shown as an inset with an expanded axis due to the low overall response for this event.

3.2.2.4 Juvenile coho salmon

Influent stormwater was acutely lethal to juvenile coho salmon for each of the six sampled events with a median (range) mortality of 88% (56-100%; Figure 18). For most events, there was no mortality in effluent waters. Effluent was pooled across the three replicate BSM treatments and then re-aliquoted into three biological replicates for the salmon tests. Therefore, fish were exposed to a mixture of runoff effluent containing the average concentration of 6PPD-quinone across the three replicates. For R18 (two of three replicates overflowed), mean 6PPD-quinone concentration was 80 ng/L. This corresponded with an average mortality of 20%. For R12 (one of three replicates overflowed), mean 6PPD-quinone concentration was 20 ng/L and no mortality was observed. The R6 effluent (7 ng/L) also produced no mortality. Prior to WY8 all applied stormwater infiltrated through the bioretention media and produced no coho mortality. After WY8, any overflow was kept separate from the treated effluent, and the treated effluent again completely prevented juvenile coho mortality for all bioretention treatment depths. For WY13, average 6PPD-quinone concentrations in effluents averaged 6 ng/L from R18, 13 ng/L from R12, and 14 ng/L from R6 — near or below proposed Washington state criterion to protect coho salmon in receiving waters (12 ng/L). Therefore, stormwater infiltrated through bioretention media was able to prevent acute lethal toxicity through 13 accelerated years of treatment (Figure 18). No mortality was observed in the controls or in any bioretention treatment effluent except for R18 during WY8.

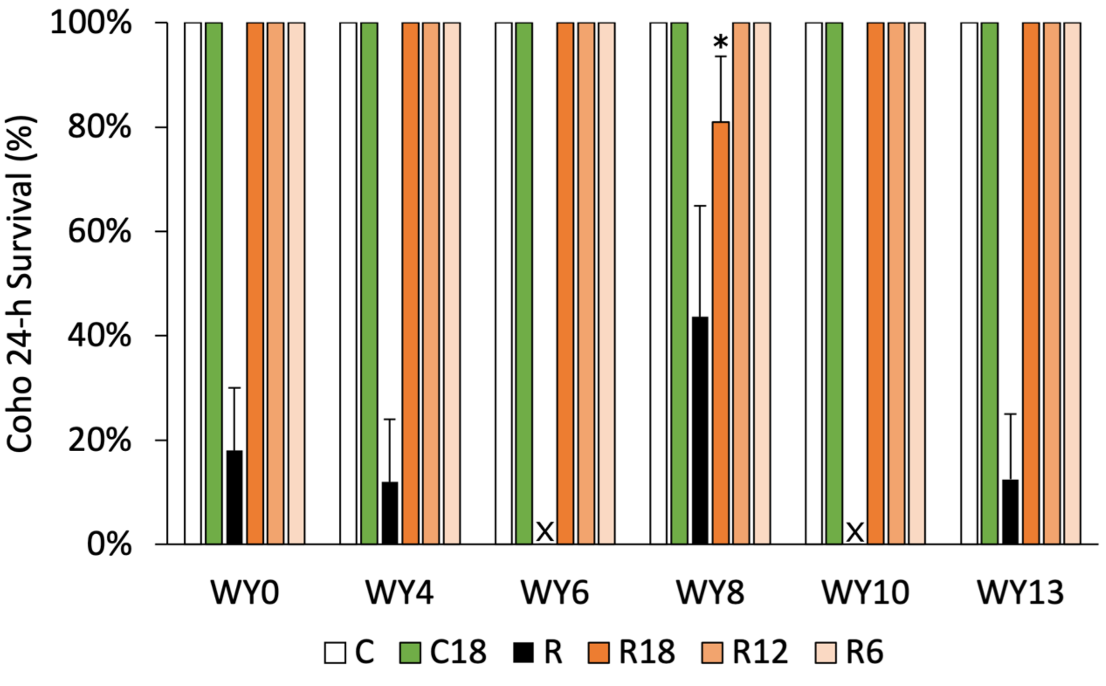


Figure 18. Survival of juvenile coho salmon following 24-h exposures to influent and effluent waters for Events beginning (WY0) or ending (all others) specific water years (WY). Coho were exposed to triplicates of influent control water (C) or runoff (R), or effluents from bioretention columns of specific depths (18”, 12”, 6”).* Overflow from

clogged column(s) in R18 were included in the treated effluent, contributing toxicity. C = 100% mortality.

3.3 Hydraulic conductivity

After the bioretention media was conditioned with clean water, the average saturated hydraulic conductivity (K_{sat}) across all treatments was 459 cm/hr (SD = 60). After the media began treating stormwater, saturated hydraulic conductivity was measured at the conclusion of each WY. Average K_{sat} values generally decreased for all treatments over the first four water years after which they tended to lower values for BSM receiving runoff whereas BSM receiving clean water remained higher (Figure 19). The single large increase in K_{sat} for all treatments at WY5 may have been a result of a longer antecedent dry period (11 days) than for any prior K_{sat} determination (1-5 days) — drying may have allowed more pore spaces to develop. Clogging began to occur during WY7, resulting in overflow from the ponding area of some of the experimental columns. Removing the top layer of BSM from all replicates prior to the WY9 sampling temporarily increased hydraulic conductivity (Figure 19; SR); however, clogging and resulting overflow began occurring again soon after. Ecology requested backflushing the columns, which was conducted after the WY9 sampling event but prior to conducting measuring K_{sat} for WY9. Another backflush (BF2) preceded the K_{sat} conducted following the WY11 sampling but produced almost no improvement in K_{sat} and no reduction in clogging for subsequent events. By the end of WY13 the K_{sat} for the BSM receiving clean water (C18) was reduced to 199 cm/hr (SD = 22), more than half its starting value following conditioning, but repeated measures model analysis of hydraulic conductivity for C18 determined only a weak relationship with time (p = 0.072; Table 20). The media receiving runoff had significantly lower K_{sat} by the end of WY13 (Table 20). The rate of decline overall was fastest for R6>R12>R18.

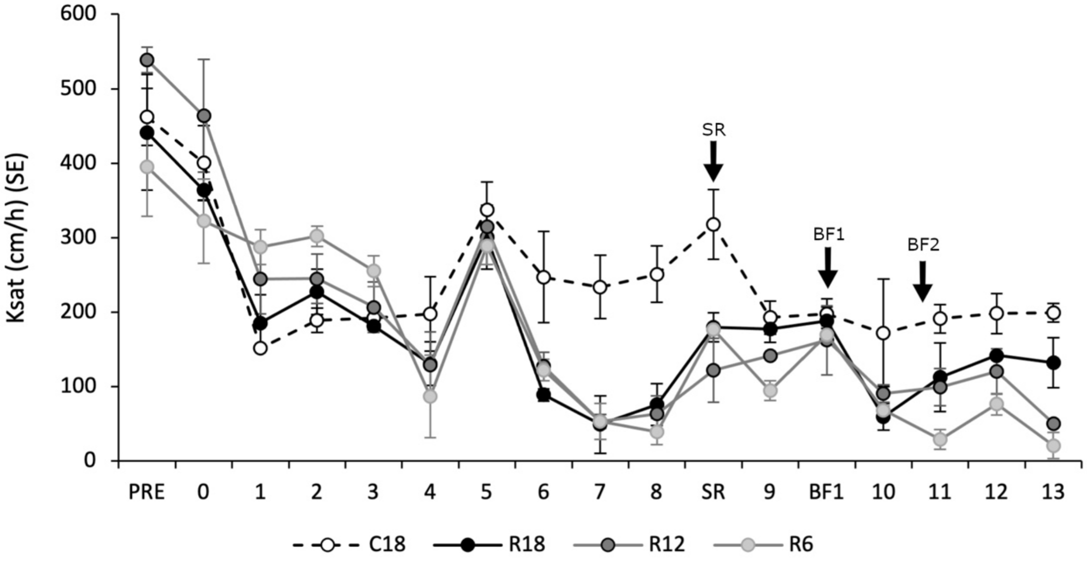


Figure 19. Saturated hydraulic conductivity (mean ± SE) for three replicates of bioretention media at three depths (6”, 12”, 18”) after conditioning with clean water (PRE)

or after receiving runoff (R) or clean water (C; 18" only) after each of 13 water years (WY). Arrows indicate maintenance actions to address clogging including top 1" soil reduction (SR) and backflushing (BF).

Table 20. Main effects of the linear mixed effects model of saturated hydraulic conductivity (Ksat) over time for the three depths of bioretention soil media (BSM) receiving runoff (R6, R12, R18) compared with the control BSM (C18) as a function of water year (WY), evaluated relative to WY13.

Factor	Estimate	SE	df	t value	p	Interpretation
C18 (reference)	189	24	69	7.8	<0.001	Expected K_{sat} for reference group at WY13
R18	-114	33	158	-3.4	<0.001	Difference in Ksat for R18 relative to C18
R12	-159	33	158	-4.8	0.009	… for R12 relative to C18
R6	-197	33	158	-5.9	<0.001	…for R6 relative to C18
WY (C18)	-6	3	158	-1.8	0.072	Slope of K_{sat} change over time for C18
R18:WY	-7	4	158	-1.7	0.094	Difference in slope for R18 relative to C18
R12:WY	-16	4	158	-3.6	<0.001	…for R12 relative to C18
R6:WY	-18	4	158	-4.2	<0.001	…for R6 relative to C18

4. Discussion

4.1. Longevity of bioretention performance

Lifespan in bioretention studies can generally be defined by any one of three conditions: 1) pollutant breakthrough, 2) pollutant accumulation to hazardous levels, and 3) clogging and hydraulic failure. Breakthrough is typically defined as the saturation of binding sites within media to the point that heavy metals are exported to receiving waters. By this definition, some studies suggest that breakthrough can be resisted for decades.

However, reductions in bioretention performance can be observed before breakthrough and still pose a large threat to receiving water bodies. In a laboratory-scale accelerated dosing experiment, Paus et al. (2014) defined performance declines as a percentage of breakthrough. Even by this more conservative metric, Paus et al. estimated that 10% breakthrough (i.e., when the effluent concentration is 10% of influent concentrations) can be resisted for decades, assuming the media has sufficient organic matter content. However, limitations of these breakthrough experiments make it difficult to extrapolate to the performance of field-scale installations. Common limitations include a small-scale, continuous-flow laboratory approach and the use of synthetic or finely filtered stormwater (often with contamination concentrations higher than that of typical stormwater concentrations), with the intention of achieving breakthrough in a convenient timeframe. The current study addresses some of these limitations through increased experimental column size, the application of discrete dosing events, the inclusion of a climatic regime, and the use of field-collected stormwater runoff; all factors that more closely mimic conditions expected of in-ground bioretention installations.

We hypothesized that under the more realistic conditions of the current study, water quality from bioretention treatment would decline within five to eight water years of operation. However, breakthrough or even more general reductions in chemical removals were not observed over the 13 water years. Performance declines may have been delayed in the current study because of the high organic matter content and cation exchange capacity (CEC) of the mulch and compost components, which are likely to increase the adsorptive capacity of bioretention and delay contaminant breakthrough (Hatt et al., 2011; Paus et al., 2014).

Initially, the bioretention media was a source of nutrients, and transiently metals, to effluent waters. This initial flush of nutrients and metals is not common across studies and regions with varying bioretention soil composition and design specifications. Reports of nutrients removal, especially, are variable among studies using different media composition (Davis et al., 2009). In a laboratory study of bioretention systems, Davis et al. (2001) reported that ammonium, total Kjeldahl nitrogen, and nitrates patterns were similar throughout the entire testing period, suggesting that an initial flush did not occur. The initial flush observed in the current study and other studies in Washington State is likely a result of the high compost volume fraction (CVF) of the bioretention media specified by the Washington State Department of Ecology (Ecology, 2012). Although a valuable amendment in terms of supporting vegetative growth and removing toxic metals, compost is also subject to sustained leaching of nutrients and dissolved organic matter, which readily associates with metals. The compost used in the current study had an organic matter content of 57% and the sand 1%. In the 60:40 (v:v) bioretention blend, the resulting media had an estimated organic matter content of 34.6%. For comparison, the sandy loam soil used in bioretention systems in the study by Davis et al. (2001) had an organic matter content of just 0.6%.

An investigation of the effects of CVF on toxic metal sorption revealed that increasing CVF in the media delayed breakthrough for cadmium and zinc (Paus et al., 2014). The study estimated that a 30% CVF media with a depth of 22 cm would prevent 10% breakthrough of zinc for approximately 19 years (compared to five years for a 10% CVF media). However, the 30% CVF columns were also a significant source of phosphorous (Paus et al., 2014). Research on the CVF needed to sustain plant growth in GSI is limited and likely varies by region and plant species. A recent study comparing the Washington State media containing 40% CVF with an alternative media containing no compost (70% sand/20% coconut coir/10% biochar) showed significantly less leaching of nutrients from the compost-free media but also clearly showed the trade-off with plant growth (Ullom et al., 2024). BSM longevity in terms of dissolved metals removal therefore requires striking a balance between the benefits of compost as the organic matter (increased sorption capacity and plant growth) and limiting export of nutrients from the organic matter. In the current study, we saw no temporal trend in effluent metal concentrations over 13 water years, indicating no loss of metal removal performance across the study for the Washington State media containing 40% CVF. Meanwhile, the high CVF also resulted in continued export of nutrients and contribution of metals to effluents.

Long-term sequestrations of pollutants within bioretention soil media can elevate concentrations to levels associated with risk to environmental and human health. However, estimates and guidelines concerning the effective lifespan of bioretention soils in regard to pollutant accumulation are inconsistent. This is compounded by the fact that the BMP media are not included in hazardous waste disposal legislation in the US. Several researchers have compared metal loadings in stormwater BMP media to those permitted on US lands through the application of wastewater biosolids (Davis et al., 2003; EPA, 1993). Although street waste is classified as non-hazardous, BMP media can also be evaluated with respect to hazardous waste classification criteria based on allowable toxicity characteristics leaching procedures (TCLP) concentrations. TCLP is used to determine whether a waste can be accepted into a typical municipal landfill based on the potential of the waste to leach toxic chemicals (as defined by RCRA Subtitle D). However, TCLP limits only exist for cadmium and lead. Based on both wastewater biosolids and TCLP criteria, Davis et al. (2003) estimated that pollutant accumulation in bioretention systems is not expected to be a concern for at least 15 years. Similarly, measured metals (copper and zinc) concentrations in media from an 11-year-old bioretention cell in North Carolina remained far below remediation thresholds, when compared to EPA’s Ecological Soil Screening Levels (Johnson & Hunt, 2016). Another study estimated that bioretention soils can remain operational for decades if soils are classified according to German thresholds for construction use (i.e., limited human contact) (Ingvertsen et al., 2012). However, an Australian study reported that after the equivalent of 12-15 years of stormwater application, cadmium, copper, and zinc levels in the BSM of laboratory-scale bioretention columns exceeded ecological guidelines and cadmium concentrations also exceeded human health guidelines, according to thresholds established by Australia’s National Environment Protection Council (Hatt et al., 2011). In the current study, we saw copper, nickel, and zinc exceeded EPA soil screening levels in the surface layer of BSM after the equivalent of 8 years of stormwater application. Levels were not exceeded for arsenic, lead, or PAHs. Maintenance actions that were performed to alleviate clogging (top 1” soil removal and backflushing) prevented re-assessing contaminant accumulation by the end of the study.

Clogging is often cited as one of the biggest disadvantages of infiltration systems, and various studies predict that bioretention filter media are limited by clogging, instead of by contaminant breakthrough (Li & Davis, 2008a; Paus et al., 2013; Siriwardene et al., 2007). Infiltration capacity in the current study remained effective through seven accelerated years of treatment. This finding is supported by laboratory studies estimating that bioretention cells can maintain effective infiltration capacity for more than six years (Paus et al., 2013). Clogging however became evident during the 8th water year as applied water began exceeding the ponding capacity of the experimental columns and overflowing. As the focus of the study was on the performance of the media itself, plants were not included in the experimental columns of BSM. Plants with thicker roots can help maintain hydraulic conductivity in bioretention systems (Techer & Berthier, 2023). The limitation of clogging observed in the current study may therefore have been prevented had appropriate plants been included in the experimental columns. Evaluation of the long-term performance of bioretention media’s filtration capacity for chemical pollutants should include appropriate plants.

Endpoints that are evaluated when considering bioretention cell lifespan do not typically consider 'biological performance' — meaning the ability to prevent toxicity. Of the few studies that have addressed the role of bioretention in reducing the toxicity of stormwater runoff to aquatic organisms, performance was evaluated after only a single storm (McIntyre, Edmunds, Anulacion, et al., 2016; McIntyre, Edmunds, Redig, et al., 2016; McIntyre et al., 2014, 2015) or less than two years of service (Spromberg et al., 2016; Young et al., 2018). It is therefore unclear whether biological performance follows the same timeline as chemical performance.

Toxicity testing in the current study using juvenile coho salmon, a species especially sensitive to runoff (McIntyre et al., 2018), points to the continued ability of all bioretention depths to prevent acute lethal toxicity after 13 simulated years of treatment. Influent stormwater that caused 56-100% mortality was no longer toxic after infiltrated through bioretention even at the end of the study. This phenomenon is consistent with past research showing that bioretention treatment of roadway runoff is protective of juvenile (McIntyre et al., 2015) and adult coho salmon (Spromberg et al., 2016), extending our confidence that this benefit continues for at least a decade of treatment. The toxicant in runoff primarily responsible for coho salmon mortality (6PPD-quinone; LC50 = 95 ng/L; Tian et al., 2002) was measured for two of the six events to which coho were exposed. Mortality was only present when overflow was pooled with effluent, resulting in concentrations >79 ng/L. When all runoff was infiltrated, 6PPD-quinone concentrations were significantly reduced (92-97%), resulting in concentrations <15 ng/L which caused no mortality.

Although the sensitivity of coho salmon to roadway runoff makes them a good test organism for evaluating the toxicological effects of stormwater treatment, acute survival tests do not capture ecologically relevant toxicity effects that may reduce fitness or precede mortality. Zebrafish embryos were used to examine sublethal effects of runoff and treatment effectiveness. Early water year testing in the current study using zebrafish embryos indicated a lack of sensitivity to influent stormwater for morphometric endpoints. This is in contrast to previous studies in which embryos exposed to stormwater runoff showed morphometric impairments (Harding et al., 2020; McIntyre et al., 2014; Skinner et al., 1999) consistent with exposure to aromatic hydrocarbons such as PAHs (Harding et al., 2020; Incardona et al., 2004, 2006, 2011). The lack of sensitivity to influent stormwater in the current study was likely due to lower concentrations of contaminants in the influent stormwater samples compared with stormwater samples that produced acutely lethal and sublethal toxicity in previous studies. Total PAHs in runoff used in toxicity testing in previous studies inducing morphometric changes ranged from 2 to 23 μg/L (Harding et al., 2020; McIntyre et al., 2014). In the current study the maximum concentration measured was 1.4 μg/L (Appendix 2), helping to explain why morphometric responses in the current study were weak.

Previous studies have shown that transcription of *cyp1a* is highly responsive to stormwater, with gene expression that can be 100-fold or higher than controls (McIntyre et al. 2016). Aromatic hydrocarbons, including many PAHs, are strong agonists of this gene which can respond even when PAHs are below detection limits. In a study by McIntyre et al. (2015), PAHs were reduced by bioretention treatment of runoff to below detection limits, but juvenile coho had elevated levels of PAH metabolites in their bile relative to controls. Similarly, in zebrafish embryos reared with bioretention-treated runoff, morphometric impairments were barely or completely indistinguishable from controls (McIntyre et al., 2014) and yet the transcriptional response of *cyp1a* was only partially prevented (McIntyre et al., 2016). In the current study, zebrafish gene expression of *cyp1a* was used as a biomarker of sublethal response to runoff and treatment. As expected, influent runoff across the five measured events induced a robust response of *cyp1a*. Low concentrations of PAHs (0.05-0.62 μg/L) resulted in mRNA levels that were up to 67-fold higher than controls. It is likely that other aromatic contaminants besides PAHs contribute to the response of *cyp1a* and this should be examined in greater detail to better understand what determines the *cyp1a* response. Treatment (R18) reduced the response of *cyp1a* to 3-24 -fold above controls, representing a reduction of 34-95% in response concurrent with >99% reduction of PAHs. In the first three tested events (WY 4-6) the response to R18 effluent was reduced to levels not different from the control treatment (C18). In the final two tested events (WY 12-13) R18 effluent caused a response significantly greater than the C18 treatment. This result may imply that treatment effectiveness was reduced towards the end of the study compared with earlier performance.

4.2. Water quality as a function of bioretention depth

Bioretention systems with a high CVF, like those used in western Washington, are expected to export nutrients and metals (Mullane et al., 2015; Paus et al., 2014; Taylor et al., 2018). Therefore, we hypothesized that deeper media depths would initially leach greater concentrations of contaminants in BSM and for a longer period of time. As expected, deeper media depths did initially result in higher effluent contaminant concentrations. For the first event, average effluent concentrations of nutrients and TSS generally increased with increasing BSM depth. Additionally, concentrations of total copper, zinc, and nickel were significantly higher in effluent from the 18” depth compared with the 6” depth. However, initial differences in leaching between depths decreased with time, often to the extent that differences between depths were not discernable by WY3.

Although the increased mass of compost in greater media depths resulted in greater initial export of contaminants in BSM, the organic fraction of BSM is valuable for its ability to complex and retain toxic metals. The tradeoff between initial export for some contaminants and increased adsorptive capacity for deeper BSM should be weighed. We hypothesized that deeper media depths would delay chemical and toxicological declines in bioretention performance. Evaluated across all 13 WYs, even with high overall removal rates for many contaminants, deeper media tended to release more nutrients, DOC, TSS, total and dissolved As, and total Ni across the study as a whole. In contrast, dissolved and total copper concentrations were significantly greater in effluent from the 6” compared to the 12” and 18” depths. BSM depth did not influence effluent concentrations of total or dissolved lead or zinc, dissolved nickel, or total PAHs. Overall, BSM depth did not influence performance decline (reduced % removals) across the 13 accelerated years of treatment; only TPAH and ortho-P showed a trend in effluent concentrations over the study (decreasing) and this pattern was not different among the BSM depths. For toxicology endpoints, all depths prevented coho mortality (for 100% infiltrated runoff) for all study years, and while there was a slight tendency for the deeper media to produce fewer toxic outcomes with zebrafish endpoints, more samples were needed to determine if that was an important difference. Therefore, shallower depths of BSM may be sufficient to reach bioretention system treatment goals.

We also hypothesized that a greater mass of compost could also affect hydraulic conductivity. Li and Davis (2008a) suggest that while a design media depth that is too shallow has the potential to compromise effluent quality, an overdesigned media depth could increase flow resistance. In support of this, clogging was initially observed in the deepest media treating runoff (R18), however we ultimately found that the shallowest media had the most impacted hydraulic conductivity across the study as a whole. Maintenance actions temporarily produced better outcomes for the 6” media than for the deeper media but the 6” media was also quick to return to even poorer saturated hydraulic conductivity. Additionally, when the top 1” of soil was removed from all treatments as the first maintenance action, the surface layer of the 6” depth treatment had the lowest porosity and had accumulated higher concentrations of most contaminants measured. The inclusion of appropriate plants in a long-term study to help mitigate clogging is needed to help determine if shallower depths of bioretention provide the desired balance between leaching nutrients and other contaminants and maintaining hydraulic conductivity. The performance of shallower media depths is especially promising for areas that may be constrained by space or resources or where nutrient export is a concern.

4.3. Potential study limitations

4.3.1 Extrapolation to real-world systems

Our study strove to simulate 13 water years across an approximately 4-year study period. This accelerated timescale was accomplished in two ways: 1) application of runoff volume at an accelerated rate (the equivalent of one year of runoff was applied across an approximately two-month period) and 2) establishment of a climatic regime (temperature and humidity regulation) to simulate conditions that would be expected of in-ground bioretention installations during a typical calendar year. However, certain systems, for example microbiological, do not abide by the experimental, accelerated timescale. The current study may present a more conservative estimate for how bioretention systems perform over time in terms of pollutant accumulation because microorganisms that are capable of sustaining or even regenerating the sorption capacity of BSM may not have had the time to fully establish (LeFevre et al., 2015).

Additionally, the accelerated loading approach may overestimate the time to clogging because columns are not exposed to the same frequency of wetting and drying cycles as during a typical calendar schedule. Allowing soils to dry out periodically helps to maintain infiltration rates, according to the Department of Ecology’s SWMMWW (Ecology, 2012). The benefits of intermittent wetting and drying to the functionality and lifespan of bioretention are supported by additional studies (Li & Davis, 2008a; Subramaniam et al., 2018). Under varied hydraulic conditions, particles are better able to “adjust” and redistribute throughout the media, creating a more accessible flow path for incoming solids (Li & Davis, 2008a; Subramaniam et al., 2018). Additional processes that can contribute to reduced hydraulic conductivity, but that may occur on a greater timescale than was simulated in the current study, include accumulation of microorganisms on the surface and in pore spaces (Rodgers et al., 2004) and gradual filling of pore spaces as organic matter is degraded (Hatt et al., 2008). The accelerated loading approach may be obscuring the effects of time on the hydraulic conductivity of various treatment depths. Although the fraction of compost is the same across columns, microbial degradation of organic matter could mobilize a greater mass of finer particles in deeper media depths over time. An interesting future study would be a side-by-side comparison of bioretention performance across accelerated and normal timescales. The other realistic factor to include in future testing, as mentioned previously, is the inclusion of plants with thicker/deeper roots that assist in maintaining hydraulic conductivity.

4.3.2. Reliability of fecal coliform bacteria results

Conventional culture methods are commonly used to detect fecal indicator bacteria, including fecal coliforms, in environmental water samples as a proxy for the presence of fecal contamination. However, fecal coliform results in the current study were inconsistent, raising questions about their reliability. For example, for WY0, the concentration of fecal coliform bacteria in influent clean water was reported at 5000 CFU/100mL. This result seems unlikely because the clean water was treated by reverse osmosis, which has a very high effectiveness in removing bacteria (CDC, 2008). For the six subsequent sampling events, fecal coliform counts in clean water were below detectable limits. In influent roadway runoff, fecal coliform counts for several events were also below detectable limits, despite counts in treated runoff above detectable limits. For some samples, background bacterial growth was high enough to completely preclude enumeration of fecal coliform bacteria. Several studies have proposed alternative methods to the standard culture method, including PCR-based methods, which could increase detection accuracy and speed (Ahmed et al., 2008; Delgado-Viscogliosi et al., 2005). Such alternatives should be considered in future stormwater studies.

4.3.3. Seasonality of contaminant leaching

The current study was designed in such a way that water-quality sampling events fell within the simulated month of January for each accelerated water year. For simulated January, the temperature in the environmental chamber where experimental columns were housed was approximately 4-5 °C (39-41 °F). However, increased temperatures were observed for one sampling event (WY4) due a temporary malfunction in thermal regulation in the environmental chamber. During this malfunction, temperatures were above 10 °C (50 °F) for approximately nine hours and peaked at 18 °C (64 °F). Coincident with this increase in temperature was a spike in concentration for several contaminants in the effluent of all treatments, including the clean water control BSM. This suggests a possible seasonal effect of contaminant leaching that was otherwise not captured in the current study. At warmer temperatures, biological activities of the BSM are increased, often resulting in greater decomposition of organic matter and a subsequent flushing of DOM in the effluent (Blecken et al., 2011). Metals and other contaminants that complex with DOM may be released in effluent as well. In a study by Martínez et al. (2003) effluent concentrations of DOC and soluble metals from a metal contaminated soil correlated with temperature. Additionally, several studies of BSM have reported a significant seasonal effect on bioretention pollutant reduction (Manka et al., 2016; Shrestha et al., 2018). These findings support that the spike in effluent concentration for WY4 of the current study was likely related to elevated BSM temperatures during the treatment event. The inadvertent temperature spike in the environmental chamber highlights the need to consider seasonality when evaluating the long-term ability of bioretention systems to meet effluent water quality criteria.

The WY4 sampling event also coincided with major wildfires across the West Coast in autumn 2020. During wildfire events, indoor air quality is often comparable to outdoor air quality (Mendoza et al., 2021). Another possible explanation for the contaminant concentration spike observed for WY4 is infiltration of particulate matter and contaminants associated with wildfire smoke into the environmental chamber from outside, resulting in possible sample contamination. This explanation accounts for the otherwise inexplicable elevation in the concentration of some contaminants for influent clean water samples.

5. Conclusion

Our results suggest that bioretention comprised of just 6” of a mixture of 60% sand: 40% compost by volume and sized at 5% of the impervious catchment area has the capacity to prevent acute mortality of coho salmon for more than 13 years, if hydraulic conductivity is maintained, while also contributing less metals and other contaminants to effluent waters than deeper BSM. This finding has broad implications for bioretention design and implementation. Bioretention that is effective at 1/3 of the depth currently required in WA State reduces construction and materials costs, limits the potentially deleterious export of BSM source contaminants to receiving waters, and overall expands options for use. However, the current study represents just one of a handful that have evaluated bioretention beyond a few years of service. Predicting bioretention performance in other environments can be problematic because of the many factors that can vary with time and space, including influent water quality and quantity. Additionally, some potential limitations of the accelerated loading approach limit extrapolation to real-world installations. Although the results of the current study are promising, additional studies are needed to further investigate design requirements that optimize bioretention performance and longevity.

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Appendix 1

Diagram of rainheads for applying influent water

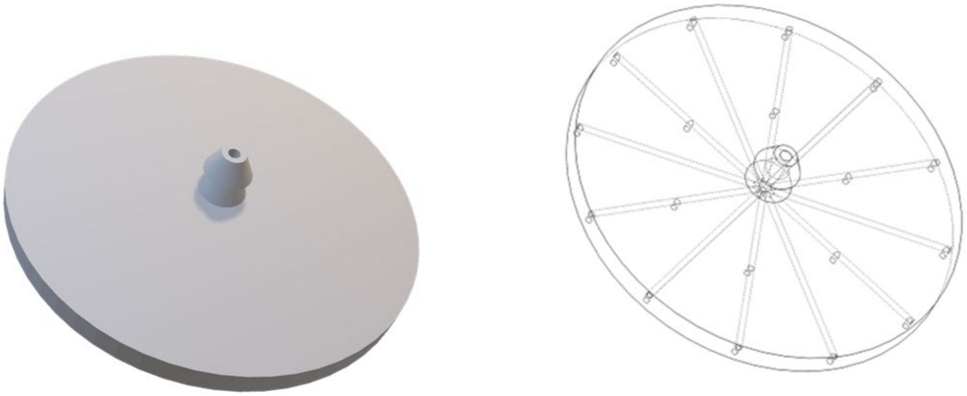

Figure A 1. Rainheads were 3-D printed using a Fused Deposition Modeling (FDM) 3D printer (Prusa i3 MK3S) and HatchBox polyethylene terephthalate glycol (PETG) filament (1.75 mm) at 0.1 mm layer height.

Appendix 2

Summary tables of each water year sampling event

Table A1.1 Average water chemistry values (standard error) for influent waters (clean water and influent stormwater runoff) and triplicate effluent waters from each of the five treatment depths plus the clean water control for Event 1. One-half of the value of the detection limit was substituted for the value of non-detects in calculating means; used when the compound was detected in at least one replicate for the treatment. Values following '<' are equal to the detection limit. n.m. = not measured for this event. CFU = colony forming units.

Compound	Units	Clean Water	Influent	6"	9"	12"	15"	18"	18" CWC
Conventional									
Dissolved Organic Carbon	mg/L	<0.08	11.5	16.8 (0.3)	19 (1)	18.1 (0.9)	19.3 (0.8)	22 (3)	11.1 (0.5)
Total Suspended Solids	mg/L	<0.5	27.4	1.5 (0.2)	1.7 (0.4)	3.1 (0.7)	2.8 (0.5)	3.5 (0.5)	3.3 (0.2)
Turbidity	NTU	0.25	17.933	3.6 (0.3)	8 (2)	9.1 (0.7)	8 (1)	12 (1)	14 (1)
Conductivity	µS/cm	1561	1478	1503 (7)	1507 (6)	1529 (8)	1539 (10)	1541 (4)	1645 (5)
pH	n.a.	7.68	7.755	7.51 (0.01)	7.489 (0.009)	7.45 (0.02)	7.43 (0.02)	7.462 (0.009)	7.47 (0.01)
Alkalinity	mg/L as $CaCO_3$	32.9	62.2	53.5 (0.8)	56 (1)	54.3 (0.4)	61 (2)	60 (2)	39.6 (0.8)
Temperature	°F	n.m.	n.m.	n.m.	n.m.	n.m.	n.m.	n.m.	n.m.
Dissolved Calcium	mg/L	7.89	86.3	60 (2)	54 (2)	52 (2)	54 (4)	49 (1)	32.6 (0.3)
Dissolved Magnesium	mg/L	20.9	3.1	12.3 (0.9)	13.8 (0.9)	14 (0.9)	13.9 (0.5)	14.5 (0.6)	10.9 (0.1)
Dissolved Sodium	mg/L	239	195	198 (1)	202 (2)	196 (2)	199 (2)	195 (2)	230 (0.6)
Nutrients									
Nitrate/Nitrite	mg/L-N	0.17	0.67	2.2 (0.3)	2.6 (0.4)	3.5 (0.7)	5 (2)	4.6 (0.5)	3.6 (0.4)
Orthophosphate, as P	mg/L	0.04	0.04	0.14 (0.02)	0.22 (0.04)	0.27 (0.02)	0.29 (0.03)	0.3 (0.02)	0.41 (0.05)
Microbiology									
Fecal Coliform by MF	CFU/ 100 mL	5000	10	453 (300)	177 (100)	10 (0)	27 (10)	10 (0)	700 (700)

Table A1.2. Average concentrations of dissolved and total metals in μg/L (standard error) for influent waters (clean water and influent stormwater runoff) and triplicate effluent waters from each of the five treatment depths plus the clean water control for Event 1. One-half of the value of the detection limit was substituted for the value of non-detects in calculating means; used when the compound was detected in at least one replicate for the treatment. Values following '<' are equal to the detection limit.

Compound	Clean Water	Influent	6"	9"	12"	15"	18"	18" CWC
Dissolved As	<0.05	<0.05	0.4 (0.2)	<0.05	<0.05	1 (0.6)	0.6 (0.3)	0.8 (0.6)
Dissolved Cd	<0.05	<0.05	<0.05	<0.05	<0.05	<0.05	<0.05	<0.05
Dissolved Cu	<0.05	12.6	9.8 (0.1)	10 (1)	9.1 (0.9)	7 (3)	11 (0.6)	9 (0.1)
Dissolved Pb	0.8	<0.079	<0.079	<0.079	<0.079	3 (3)	<0.079	<0.079
Dissolved Ni	<0.2	2.9	2 (0.3)	0.75 (0.09)	0.8 (0.4)	1.7 (0.6)	2.7 (0.5)	3 (0.4)
Dissolved Zn	0.6	18.3	6 (1)	4 (1)	5 (1)	7.2 (0.3)	6.3 (0.7)	1.8 (0.3)
As	<0.05	<0.05	0.93 (0.09)	0.9 (0.2)	1.3 (0.2)	1.5 (0.4)	1.5 (0.1)	1.7 (0.5)
Cd	<0.05	<0.05	<0.05	<0.05	<0.05	<0.05	<0.05	<0.05
Cu	0.317	25.5	12 (0.5)	12.93 (0.07)	13 (0.3)	12.2 (0.5)	14.3 (0.7)	12.2 (0.4)
Pb	<0.079	1.5	0.2 (0.2)	<0.079	<0.079	<0.079	<0.079	<0.079
Ni	<0.2	2.9	2.3 (0.2)	2.75 (0.09)	3.1 (0.06)	3.1 (0.2)	4.2 (0.4)	3.4 (0.2)
Zn	1.2	77.6	6.6 (0.5)	7.2 (0.3)	8.1 (0.2)	7.9 (0.2)	8.57 (0.09)	2.3 (0.3)

Table A1. 3. Average polycyclic aromatic hydrocarbon (PAH) concentrations in μg/L (standard error) for influent waters (clean water and influent stormwater runoff) and triplicate effluent waters from each of the five treatment depths plus the clean water control for Event 1. One-half of the value of the detection limit was substituted for the value of non-detects in calculating means; used when the compound was detected in at least one replicate for the treatment. Values following '<' are equal to the detection limit.

PAHs	Clean Water	Influent	6"	9"	12"	15"	18"	18" CWC
1-Methylnaphthalene	0.002	0.011	0.007 (3e-04)	0.007 (6e-04)	0.007 (3e-04)	0.007 (3e-04)	0.007 (0)	0.003 (3e-04)
2-Chloronaphthalene	<0.001	0.002	0.001 (8e-04)	<0.001	<0.001	<0.001	<0.001	<0.001
2-Methylnaphthalene	0.005	0.015	0.005 (9e-04)	0.005 (3e-04)	0.005 (7e-04)	0.005 (7e-04)	0.005 (3e-04)	0.003 (3e-04)
Naphthalene	0.011	0.04	0.013 (7e-04)	0.011 (7e-04)	0.01 (9e-04)	0.011 (0.002)	0.011 (0.001)	0.009 (3e-04)
Acenaphthene	<0.003	0.006	0.002 (8e-04)	<0.003	<0.003	<0.003	<0.003	<0.003
Acenaphthylene	<0.002	0.003	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002
Anthracene	<0.001	0.013	0.001 (5e-04)	<0.001	<0.001	<0.001	<0.001	<0.001
Carbazole	<0.001	0.143	0.017 (0.008)	0.005 (0.001)	0.004 (3e-04)	0.006 (0.002)	0.005 (0.002)	0.007 (3e-04)
Dibenzofuran	<0.002	0.011	0.004 (7e-04)	0.003 (3e-04)	0.001 (3e-04)	0.002 (0)	0.002 (0)	<0.002
Fluorene	<0.002	0.012	0.003 (9e-04)	<0.002	<0.002	<0.002	<0.002	<0.002
Phenanthrene	<0.001	0.1	0.007 (0.003)	0.004 (3e-04)	0.004 (3e-04)	0.003 (3e-04)	0.003 (3e-04)	0.002 (3e-04)
Benz[a]anthracene	<8e-04	0.005	0.001 (2e-04)	<8e-04	<8e-04	<8e-04	<8e-04	<8e-04
Chrysene	<9e-04	0.01	0.001 (3e-04)	0.001 (0)	0.001 (0)	0.002 (3e-04)	0.002 (3e-04)	0.001 (0)
Fluoranthene	<0.002	0.047	0.002 (7e-04)	<0.002	<0.002	<0.002	0.001 (3e-04)	<0.002
Pyrene	<0.001	0.038	0.002 (3e-04)	0.002 (3e-04)	0.001 (3e-04)	0.002 (0)	0.002 (0)	0.001 (2e-04)
Benzo(a)pyrene	<0.002	0.003	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002
Benzo(b)fluoranthene	<5e-04	0.004	<5e-04	<5e-04	<5e-04	<5e-04	<5e-04	<5e-04
Benzo(j)fluoranthene	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002
Benzo(k)fluoranthene	<0.003	<0.003	<0.003	<0.003	<0.003	<0.003	<0.003	<0.003
Dibenzo(a,h)anthracene	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001
Perylene	<0.006	<0.006	<0.006	<0.006	<0.006	<0.006	<0.006	<0.006
Benzo(ghi)perylene	<0.001	0.008	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001
Indeno(1,2,3-cd)pyrene	<0.001	0.003	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001
Total PAHs	**0.035**	**0.48**	**0.077 (0.009)**	**0.051 (0.002)**	**0.047 (0.001)**	**0.051 (0.002)**	**0.052 (0.002)**	**0.04 (8e-04)**

Table A1. 4. Average water chemistry values (standard error) for influent waters (clean water and influent stormwater runoff) and triplicate effluent waters from each of the five treatment depths plus the clean water control for Water Year 1. One-half of the value of the detection limit was substituted for the value of non-detects in calculating means; used when the compound was detected in at least one replicate for the treatment. Values following '<' are equal to the detection limit. CFU = colony forming units.

Compound	Units	Clean Water	Influent	6"	9"	12"	15"	18"	18" CWC
Conventional									
Dissolved Organic Carbon	mg/L	<0.08	<0.08	5 (0.4)	5.4 (0.1)	7.2 (0.4)	7.2 (0.2)	8.5 (0.4)	3.1 (0.06)
Total Suspended Solids	mg/L	<0.5	62	5.9 (0.5)	6.9 (0.9)	11 (0)	11 (1)	12 (1)	1.6 (0.2)
Turbidity	NTU	0.09	58.9	27 (3)	28 (1)	30.4 (0.9)	32 (1)	34 (2)	5.2 (0.3)
Conductivity	µS/cm	1559	76.7	750 (10)	942 (40)	1146 (20)	1264 (5)	1490 (8)	1540 (4)
pH	n.a.	7.571	7.73	8.137 (0.009)	7.99 (0.03)	7.87 (0.01)	7.86 (0.01)	7.778 (0.008)	7.48 (0.01)
Alkalinity	mg/L as $CaCO_3$	48.3	40.3	51 (2)	54 (1)	60 (2)	60 (1)	64.7 (0.2)	51 (2)
Temperature	°F	46.7	39.3	39.9 (0.2)	40.8 (0.3)	39.5 (0.8)	39.2 (0.2)	39.6 (0.9)	39.9 (0.4)
Dissolved Calcium	µg/L	6.91	7.66	3.3 (0.1)	3.5 (0.6)	5.5 (0.5)	6.6 (0.3)	7.6 (0.2)	19 (0.5)
Dissolved Magnesium	µg/L	19.4	0.184	0.35 (0.02)	0.39 (0.05)	0.63 (0.05)	0.78 (0.04)	0.917 (0.004)	15.7 (0.2)
Dissolved Sodium	µg/L	196	6.45	120 (1)	126 (9)	166 (10)	189.7 (0.7)	213 (6)	193.3 (0.9)
Nutrients									
Nitrate/Nitrite	mg/L	0.15	0.13	0.3 (0.06)	0.25 (0.09)	0.5 (0.1)	0.5 (0.2)	0.84 (0.02)	0.22 (0.07)
Orthophosphate, as P	mg/L	<0.01	<0.01	0.67 (0.01)	0.97 (0.09)	1.42 (0.03)	1.6 (0.1)	1.94 (0.05)	0.31 (0.02)
Microbiology									
Fecal Coliform by MF	CFU/ 100 mL	1	80	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.

Table A1. 5. Average concentrations of dissolved and total metals in μg/L (standard error) for influent waters (clean water and influent stormwater runoff) and triplicate effluent waters from each of the five treatment depths plus the clean water control for Water Year 1. One-half of the value of the detection limit was substituted for the value of non-detects in calculating means; used when the compound was detected in at least one replicate for the treatment. Values following '<' are equal to the detection limit.

Compound	Clean Water	Influent	6"	9"	12"	15"	18"	18" CWC
Dissolved As	<0.05	0.5	1.8 (0.2)	2.13 (0.09)	2.2 (0.2)	2.5 (0.2)	3.5 (0.3)	1.03 (0.03)
Dissolved Cd	<0.05	<0.05	<0.05	<0.05	<0.05	<0.05	<0.05	<0.05
Dissolved Cu	<0.05	5	5.9 (0.4)	5.7 (0.1)	6.9 (0.2)	7.1 (0.3)	7.6 (0.2)	1.13 (0.09)
Dissolved Pb	<0.079	<0.079	<0.079	<0.079	<0.079	<0.079	<0.079	<0.079
Dissolved Ni	<0.2	<0.2	1.1 (0.3)	2 (1)	2.4 (0.9)	1.6 (0.1)	2 (0.1)	0.6 (0.06)
Dissolved Zn	<0.19	10.3	3.1 (0.8)	2.43 (0.09)	2.7 (0.1)	4 (1)	3.53 (0.09)	<0.19
As	<0.05	0.8	2.1 (0.2)	2.5 (0.2)	2.9 (0.1)	3 (0.2)	3.8 (0.1)	1.2 (0.1)
Cd	<0.05	<0.05	<0.05	<0.05	<0.05	<0.05	<0.05	<0.05
Cu	<0.2	17.7	11 (3)	9 (2)	7.1 (0.2)	7.23 (0.03)	7.8 (0.3)	1.37 (0.07)
Pb	<0.079	3.8	0.8 (0.5)	0.4 (0.4)	<0.079	0.2 (0.2)	0.4 (0.2)	<0.079
Ni	<0.2	1.6	1.9 (0.5)	3 (2)	2.7 (0.9)	2 (0.06)	2.2 (0.1)	0.8 (0.06)
Zn	<0.19	88.1	15 (7)	15 (4)	8.73 (0.03)	8.9 (0.8)	8.7 (0.3)	0.9 (0.8)

Table A1. 6. Average polycyclic aromatic hydrocarbon (PAH) concentrations in μg/L (standard error) for influent waters (clean water and influent stormwater runoff) and triplicate effluent waters from each of the five treatment depths plus the clean water control for Water Year 1. One-half of the value of the detection limit was substituted for the value of non-detects in calculating means; used when the compound was detected in at least one replicate for the treatment. Values following '<' are equal to the detection limit.

PAHs	Clean Water	Influent	6"	9"	12"	15"	18"	18" CWC
1-Methylnaphthalene	0.006	0.021	0.007 (0.001)	0.012 (0.001)	0.009 (0.001)	0.016 (0.007)	0.018 (0.002)	0.012 (0.002)
2-Chloronaphthalene	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001
2-Methylnaphthalene	0.004	0.036	0.004 (3e-04)	0.004 (3e-04)	0.003 (3e-04)	0.003 (3e-04)	0.005 (6e-04)	0.004 (3e-04)
Naphthalene	0.011	0.114	0.012 (7e-04)	0.01 (7e-04)	0.008 (7e-04)	0.01 (0.001)	0.011 (3e-04)	0.009 (0.001)
Acenaphthene	<0.003	0.004	<0.003	<0.003	<0.003	<0.003	<0.003	<0.003
Acenaphthylene	<0.002	0.008	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002
Anthracene	<0.001	0.01	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001
Carbazole	0.004	0.04	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001
Dibenzofuran	<0.002	0.008	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002
Fluorene	<0.002	0.008	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002
Phenanthrene	0.004	0.084	0.004 (0)	0.003 (0.001)	0.004 (7e-04)	0.004 (3e-04)	0.004 (0)	0.003 (7e-04)
Benz[a]anthracene	<8e-04	0.022	0.001 (5e-04)	<8e-04	<8e-04	0.001 (5e-04)	0.002 (0)	<8e-04
Chrysene	<0.001	0.043	0.003 (0)	0.002 (0)	0.003 (3e-04)	0.003 (0)	0.003 (0)	<0.001
Fluoranthene	<0.002	0.095	0.004 (0)	0.003 (3e-04)	0.003 (0)	0.003 (0)	0.003 (0)	<0.002
Pyrene	<0.001	0.131	0.005 (3e-04)	0.004 (3e-04)	0.004 (3e-04)	0.003 (3e-04)	0.003 (0)	<0.001
Benzo(a)pyrene	<0.003	0.019	<0.003	0.002 (5e-04)	<0.003	<0.003	<0.003	<0.003
Benzo(b)fluoranthene	<5e-04	0.031	0.002 (0)	0.002 (0)	0.001 (9e-04)	0.002 (0)	0.002 (3e-04)	<5e-04
Benzo(j)fluoranthene	<0.002	0.01	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002
Benzo(k)fluoranthene	<0.003	0.014	<0.004	<0.004	<0.004	<0.004	<0.004	<0.004
Dibenzo(a,h)anthracene	<0.001	0.003	<0.001	<0.001	<0.001	<0.001	<0.002	<0.002
Perylene	<0.006	0.01	<0.007	<0.007	<0.007	<0.007	<0.007	<0.007
Benzo(ghi)perylene	<0.002	0.057	0.003 (0.001)	0.004 (0)	0.003 (3e-04)	0.003 (3e-04)	0.002 (7e-04)	<0.002
Indeno(1,2,3-cd)pyrene	<0.001	0.021	0.001 (8e-04)	0.002 (3e-04)	0.001 (5e-04)	0.001 (5e-04)	<0.001	<0.001
Total PAHs	**0.046**	**0.789**	**0.061 (0.002)**	**0.061 (0.002)**	**0.054 (0.002)**	**0.063 (0.007)**	**0.068 (0.002)**	**0.046 (0.002)**

Table A1. 7. Average water chemistry values (standard error) for influent waters (clean water and influent stormwater runoff) and triplicate effluent waters from each of the five treatment depths plus the clean water control for Water Year 2. One-half of the value of the detection limit was substituted for the value of non-detects in calculating means; used when the compound was detected in at least one replicate for the treatment. Values following '<' are equal to the detection limit. CFU = colony forming units.

Compound	Units	Clean Water	Influent	6"	9"	12"	15"	18"	18" CWC
Conventional									
Dissolved Organic Carbon	mg/L	<0.08	4.2	3 (0.3)	2.8 (0.1)	3.3 (0.1)	4 (0.3)	4.53 (0.09)	2.9 (0.1)
Total Suspended Solids	mg/L	<0.5	47	1.2 (0.4)	1.7 (0.6)	1.47 (0.07)	1.9 (0.2)	1.5 (0.3)	0.6 (0.2)
Turbidity	NTU	0.09	53.7	7.7 (0.5)	12 (1)	14 (3)	16 (1)	17 (1)	3 (0.3)
Conductivity	µS/cm	1486	111.7	108 (2)	105 (1)	102.4 (0.8)	102.2 (0.7)	105.3 (0.4)	1493 (2)
pH	n.a.	7.548	7.673	7.48 (0.03)	7.37 (0.02)	7.26 (0.02)	7.24 (0.02)	7.26 (0.03)	7.288 (0.006)
Alkalinity	mg/L as $CaCO_3$	24.7	38	31.8 (0.6)	27 (3)	21 (1)	22.5 (0.4)	20.3 (0.8)	25.1 (0.2)
Temperature	°F	NA	33.2	36 (0.3)	34.6 (0.3)	33.5 (0.7)	32.7 (0.4)	34.3 (0.4)	35.5 (0.4)
Dissolved Calcium	µg/L	6.93	12.3	4 (0.2)	1.1 (0.7)	0.21 (0.09)	0.057 (0.006)	0.039 (0.003)	10.8 (0.3)
Dissolved Magnesium	µg/L	18.1	0.386	0.77 (0.05)	0.3 (0.2)	0.08 (0.02)	0.051 (0.004)	0.047 (0.004)	19.1 (0.1)
Dissolved Sodium	µg/L	236	6.49	14.1 (0.5)	19 (2)	20.7 (0.7)	21.8 (0.1)	21.9 (0.3)	228 (0)
Nutrients									
Nitrate/Nitrite	mg/L	0.08	0.07	0.2 (0)	0.26 (0.02)	0.28 (0.02)	0.32 (0.01)	0.39 (0.04)	0.37 (0.01)
Orthophosphate, as P	mg/L	0.04	0.04	0.067 (0.003)	0.087 (0.007)	0.12 (0.006)	0.15 (0.01)	0.21 (0.01)	0.213 (0.009)
Microbiology									
Fecal Coliform by MF	CFU/ 100 mL	20	1500	953 (500)	560 (300)	387 (60)	493 (200)	347 (100)	20 (0)

Table A1. 8. Average concentrations of dissolved and total metals in µg/L (standard error) for influent waters (clean water and influent stormwater runoff) and triplicate effluent waters from each of the five treatment depths plus the clean water control for Water Year 2. One-half of the value of the detection limit was substituted for the value of non-detects in calculating means; used when the compound was detected in at least one replicate for the treatment. Values following '<' are equal to the detection limit.

Compound	Clean Water	Influent	6"	9"	12"	15"	18"	18" CWC
Dissolved As	<0.05	<0.05	<0.05	<0.05	<0.05	0.2 (0.1)	0.2 (0.1)	0.5 (0.06)
Dissolved Cd	<0.05	<0.05	<0.05	<0.05	<0.05	<0.05	<0.05	<0.05
Dissolved Cu	0.3	4.4	2.43 (0.03)	2.8 (0.1)	2.8 (0.2)	2.6 (0.5)	1.2 (0.4)	1.6 (0.1)
Dissolved Pb	<0.079	<0.079	<0.079	<0.079	<0.079	<0.079	<0.079	<0.079
Dissolved Ni	<0.2	<0.2	<0.2	1 (1)	0.7 (0.6)	0.2 (0.1)	<0.2	<0.2
Dissolved Zn	0.5	8.8	1.83 (0.03)	2.1 (0.2)	2 (0.2)	1.6 (0.6)	0.2 (0.1)	0.2 (0.1)
As	<0.05	0.4	0.3 (0.2)	<0.05	<0.05	0.43 (0.03)	0.47 (0.09)	0.6 (0.1)
Cd	<0.05	<0.05	<0.05	<0.05	<0.05	<0.05	<0.05	<0.05
Cu	0.7	15.5	3.53 (0.03)	3.37 (0.09)	3.5 (0.5)	3.4 (0.3)	3.3 (0.1)	2.6 (0.9)
Pb	<0.079	1.9	<0.079	<0.079	<0.079	<0.079	<0.079	<0.079
Ni	<0.2	1.2	<0.2	2 (2)	0.9 (0.8)	0.3 (0.2)	0.3 (0.2)	<0.2
Zn	1.3	58.2	4.9 (0.06)	4.1 (0.3)	3.8 (0.5)	4.2 (0.6)	3.8 (0.2)	0.55 (0.03)

Table A1. 9. Average polycyclic aromatic hydrocarbon (PAH) concentrations in µg/L (standard error) for influent waters (clean water and influent stormwater runoff) and triplicate effluent waters from each of the five treatment depths plus the clean water control for Water Year 2. One-half of the value of the detection limit was substituted for the value of non-detects in calculating means; used when the compound was detected in at least one replicate for the treatment. Values following '<' are equal to the detection limit.

PAHs	Clean Water	Influent	6"	9"	12"	15"	18"	18" CWC
1-Methylnaphthalene	<0.001	0.003	0.003 (0)	0.003 (0)	0.003 (3e-04)	0.003 (3e-04)	0.003 (3e-04)	0.004 (0.004)
2-Chloronaphthalene	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001
2-Methylnaphthalene	<0.001	0.005	0.002 (5e-04)	0.002 (5e-04)	0.002 (3e-04)	0.002 (0)	0.001 (5e-04)	0.006 (0.006)
Naphthalene	0.003	0.01	0.003 (3e-04)	0.004 (3e-04)	0.004 (3e-04)	0.004 (6e-04)	0.003 (3e-04)	0.005 (0.002)
Acenaphthene	<0.003	0.003	<0.003	<0.003	<0.003	<0.003	<0.003	<0.003
Acenaphthylene	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002
Anthracene	<0.001	0.005	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001
Carbazole	<0.001	0.013	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001
Dibenzofuran	<0.002	0.004	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002
Fluorene	<0.002	0.005	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002
Phenanthrene	0.002	0.041	0.002 (0)	0.002 (0)	0.002 (0)	0.002 (0)	0.002 (0)	<0.001
Benz[a]anthracene	<8e-04	0.009	<8e-04	<8e-04	<8e-04	<8e-04	<8e-04	<8e-04
Chrysene	<0.001	0.02	0.001 (2e-04)	0.001 (2e-04)	0.001 (3e-04)	0.001 (4e-04)	0.001 (4e-04)	<0.001
Fluoranthene	<0.002	0.059	0.002 (3e-04)	<0.002	0.001 (3e-04)	0.001 (3e-04)	<0.002	<0.002
Pyrene	<0.001	0.066	0.002 (0)	0.002 (0)	0.002 (0)	0.002 (0)	0.002 (0)	<0.001
Benzo(a)pyrene	<0.003	0.009	<0.003	<0.003	<0.003	<0.003	<0.003	<0.003
Benzo(b)fluoranthene	<5e-04	0.014	<5e-04	<5e-04	<5e-04	<5e-04	<5e-04	<5e-04
Benzo(j)fluoranthene	<0.002	0.004	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002
Benzo(k)fluoranthene	<0.004	0.007	<0.004	<0.004	<0.004	<0.004	<0.004	<0.004
Dibenzo(a,h)anthracene	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001
Perylene	<0.007	<0.007	<0.007	<0.007	<0.007	<0.007	<0.007	<0.007
Benzo(ghi)perylene	<0.002	0.028	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002
Indeno(1,2,3-cd)pyrene	<0.001	0.01	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001
Total PAHs	**0.025**	**0.32**	**0.031 (7e-04)**	**0.03 (6e-04)**	**0.033 (7e-04)**	**0.032 (9e-04)**	**0.03 (8e-04)**	**0.034 (0.007)**

Table A1. 10. Average water chemistry values (standard error) for influent waters (clean water and influent stormwater runoff) and triplicate effluent waters from each of the three treatment depths plus the clean water control for Water Year 3. One-half of the value of the detection limit was substituted for the value of non-detects in calculating means; used when the compound was detected in at least one replicate for the treatment. Values following '<' are equal to the detection limit. CFU = colony forming units.

Compound	Units	Clean Water	Influent	6"	12"	18"	18" CWC
Conventional							
Dissolved Organic Carbon	mg/L	<0.08	9.8	11.3 (0.3)	10 (1)	8.97 (0.03)	<0.08
Total Suspended Solids	mg/L	0.8	58.4	0.7 (0.1)	0.4 (0.1)	1.1 (0.3)	0.7 (0.2)
Turbidity	NTU	0.03	59.4	2.8 (0.2)	2.2 (0.3)	3.4 (0.2)	1.9 (0.1)
Conductivity	µS/cm	1553	163.7	157 (2)	153.2 (0.9)	148.6 (0.9)	1566.3 (0.9)
pH	n.a.	7.546	7.711	7.53 (0.03)	7.461 (0.003)	7.32 (0.01)	7.417 (0.009)
Alkalinity	mg/L as $CaCO_3$	31	39.1	41 (3)	39 (0.5)	35.1 (0.3)	29 (2)
Temperature	°F	NA	33	34.5 (0.9)	32.43 (0.07)	33.7 (0.3)	34.4 (0.4)
Dissolved Calcium	µg/L	6.77	14.6	14.63 (0.03)	13.5 (0.1)	12.4 (0.2)	7.6 (0.1)
Dissolved Magnesium	µg/L	16.8	0.645	1.24 (0.03)	1.47 (0.04)	1.5 (0.1)	16.87 (0.09)
Dissolved Sodium	µg/L	225	8.55	7.59 (0.06)	7.5 (0.05)	7.6 (0.1)	226.7 (0.3)
Nutrients							
Nitrate/Nitrite	mg/L	0.2	3.1	2.73 (0.09)	2.87 (0.03)	2.9 (0.06)	0.2 (0)
Orthophosphate, as P	mg/L	0.04	0.06	0.08 (0.01)	0.087 (0.003)	0.097 (0.003)	0.2 (0)
Microbiology							
Fecal Coliform by MF	CFU/ 100 mL	20	200	1120 (900)	340 (200)	240 (200)	20 (0)

Table A1. 11. Average concentrations of dissolved and total metals in μg/L (standard error) for influent waters (clean water and influent stormwater runoff) and triplicate effluent waters from each of the three treatment depths plus the clean water control for Water Year 3. One-half of the value of the detection limit was substituted for the value of non-detects in calculating means; used when the compound was detected in at least one replicate for the treatment. Values following '<' are equal to the detection limit.

Compound	Clean Water	Influent	6"	12"	18"	18" CWC
Dissolved As	<0.05	0.7	0.2 (0.2)	0.2 (0.2)	<0.05	<0.05
Dissolved Cd	<0.05	<0.05	<0.05	<0.05	<0.05	<0.05
Dissolved Cu	0.5	11.4	4.73 (0.03)	3.4 (0.2)	3.2 (0.2)	1.4 (0.06)
Dissolved Pb	<0.079	<0.079	<0.079	<0.079	<0.079	<0.079
Dissolved Ni	<0.2	0.8	<0.2	0.5 (0.4)	<0.2	<0.2
Dissolved Zn	1.3	19.5	4.1 (0.2)	4.4 (0.4)	4 (0.2)	0.57 (0.07)
As	<0.05	1.2	0.67 (0.03)	0.73 (0.07)	0.2 (0.2)	<0.05
Cd	<0.05	<0.05	<0.05	<0.05	<0.05	<0.05
Cu	0.8	32.7	5.8 (0.2)	4 (0.1)	3.77 (0.09)	2 (0)
Pb	<0.079	2.7	<0.079	<0.079	<0.079	<0.079
Ni	<0.2	2.9	<0.2	0.7 (0.6)	<0.2	<0.2
Zn	1.8	104	5.9 (0.2)	4.9 (0.1)	5.7 (0.2)	0.67 (0.09)

Table A1. 12. Average polycyclic aromatic hydrocarbon (PAH) concentrations in μg/L (standard error) for influent waters (clean water and influent stormwater runoff) and triplicate effluent waters from each of the three treatment depths plus the clean water control for Water Year 3. One-half of the value of the detection limit was substituted for the value of non-detects in calculating means; used when the compound was detected in at least one replicate for the treatment. Values following '<' are equal to the detection limit.

PAHs	Clean Water	Influent	6"	12"	18"	18" CWC
1-Methylnaphthalene	0.013	0.02	0.023 (0.002)	0.022 (0.002)	0.023 (9e-04)	0.014 (3e-04)
2-Chloronaphthalene	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001
2-Methylnaphthalene	0.022	0.026	0.021 (0.002)	0.02 (9e-04)	0.021 (9e-04)	0.022 (6e-04)
Naphthalene	0.013	0.036	0.012 (9e-04)	0.011 (9e-04)	0.012 (6e-04)	0.011 (3e-04)
Acenaphthene	<0.003	0.004	<0.003	<0.003	<0.003	<0.003
Acenaphthylene	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002
Anthracene	<0.001	0.006	<0.001	<0.001	<0.001	<0.001
Carbazole	0.002	0.007	<0.001	<0.001	<0.001	<0.001
Dibenzofuran	<0.002	0.006	<0.002	0.001 (3e-04)	0.002 (3e-04)	<0.002
Fluorene	<0.002	0.015	0.011 (0.002)	0.01 (0.001)	0.009 (3e-04)	<0.002
Phenanthrene	0.002	0.041	0.002 (5e-04)	0.001 (5e-04)	<0.001	<0.001
Benz[a]anthracene	<8e-04	0.009	<8e-04	<8e-04	<8e-04	<8e-04
Chrysene	<0.001	0.028	<0.001	<0.001	<0.001	<0.001
Fluoranthene	<0.002	0.074	<0.002	<0.002	<0.002	<0.002
Pyrene	<0.001	0.098	0.002 (0)	<0.001	<0.001	<0.001
Benzo(a)pyrene	<0.003	0.013	<0.003	<0.003	<0.003	<0.003
Benzo(b)fluoranthene	<5e-04	0.019	<5e-04	<5e-04	<5e-04	<5e-04
Benzo(j)fluoranthene	<0.002	0.007	<0.002	<0.002	<0.002	<0.002
Benzo(k)fluoranthene	<0.004	0.007	<0.004	<0.004	<0.004	<0.004
Dibenzo(a,h)anthracene	<0.001	0.002	<0.001	<0.001	<0.001	<0.001
Perylene	<0.007	<0.007	<0.007	<0.007	<0.007	<0.007
Benzo(ghi)perylene	<0.002	0.066	<0.002	<0.002	<0.002	<0.002
Indeno(1,2,3-cd)pyrene	<0.001	0.014	<0.001	<0.001	<0.001	<0.001
Total PAHs	**0.07**	**0.503**	**0.087 (0.003)**	**0.083 (0.003)**	**0.084 (0.001)**	**0.065 (7e-04)**

Table A1. 13. Average water chemistry values (standard error) for influent waters (clean water and influent stormwater runoff) and triplicate effluent waters from each of the three treatment depths plus the clean water control for Water Year 4. One-half of the value of the detection limit was substituted for the value of non-detects in calculating means; used when the compound was detected in at least one replicate for the treatment. Values following '<' are equal to the detection limit. n.m. = not measured for this event.

Compound	Units	Clean Water	Influent	6"	12"	18"	18" CWC
Conventional							
Dissolved Organic Carbon	mg/L	4.3	90	73.5 (0.5)	76 (7)	95 (4)	25 (1)
Total Suspended Solids	mg/L	<0.5	28.8	2.6 (0.7)	1.4 (0.2)	1.8 (0.2)	1.67 (0.07)
Turbidity	NTU	0.06	34.4	1.9 (0.4)	1.7 (0.5)	1.8 (0.4)	4.9 (0.4)
Conductivity	µS/cm	1477	406	391 (2)	377 (0)	368 (2)	1479 (5)
pH	n.a.	7.488	7.525	7.29 (0.03)	7.19 (0.01)	7.13 (0.02)	7.3 (0.01)
Alkalinity	mg/L as $CaCO_3$	33.3	96.5	87 (1)	80.2 (0.6)	78 (1)	33.1 (0.9)
Temperature	°F	n.m.	55.7	57.6 (0.4)	55.3 (0.1)	55.07 (0.07)	54.9 (0.2)
Dissolved Calcium	µg/L	8.68	52.8	55.53 (0.03)	51 (1)	49.2 (0.6)	11.33 (0.09)
Dissolved Magnesium	µg/L	20.1	2.44	2.8 (0.1)	3.1 (0.2)	3.7 (0.2)	20.1 (0.3)
Dissolved Sodium	µg/L	247	35.7	31.6 (0.3)	30.1 (0.6)	31.8 (0.3)	270 (20)
Nutrients							
Nitrate/Nitrite	mg/L	0.18	7.22	7.9 (0.1)	7.8 (0.04)	8.04 (0.02)	0.82 (0.08)
Orthophosphate, as P	mg/L	<0.01	<0.01	0.073 (0.003)	0.107 (0.003)	0.123 (0.003)	0.22 (0.01)
Microbiology							
Fecal Coliform by MF	CFU/100 mL	10	69000	49600 (10000)	27200 (6000)	33667 (3000)	17 (7)

MF = membrane filtration

Table A1. 14. Average concentrations of dissolved and total metals in μg/L (standard error) for influent waters (clean water and influent stormwater runoff) and triplicate effluent waters from each of the three treatment depths plus the clean water control for Water Year 4. One-half of the value of the detection limit was substituted for the value of non-detects in calculating means; used when the compound was detected in at least one replicate for the treatment. Values following '<' are equal to the detection limit.

Compound	Clean Water	Influent	6"	12"	18"	18" CWC
Dissolved As	<0.05	2.7	1.9 (0.06)	1.7 (0.06)	1.6 (0.06)	0.83 (0.03)
Dissolved Cd	<0.05	<0.05	<0.05	<0.05	<0.05	<0.05
Dissolved Cu	<0.05	69.9	22.4 (0.8)	16 (2)	16.3 (0.2)	5.7 (0.3)
Dissolved Pb	<0.079	<0.079	<0.079	<0.079	<0.079	<0.079
Dissolved Ni	<0.2	6.9	2.6 (0.06)	2.7 (0.6)	2.2 (0.1)	0.77 (0.09)
Dissolved Zn	2.4	73.4	18.4 (0.3)	17.1 (0.7)	14 (1)	1.3 (0.2)
As	<0.05	3	1.97 (0.03)	1.73 (0.03)	1.6 (0)	0.87 (0.03)
Cd	<0.05	<0.05	<0.05	<0.05	<0.05	<0.05
Cu	<0.2	86.4	24.5 (0.9)	18 (2)	16.8 (0.1)	6.5 (0.6)
Pb	<0.079	2.3	<0.079	<0.079	<0.079	<0.079
Ni	<0.2	8.4	2.67 (0.07)	3.1 (0.9)	2.23 (0.03)	0.8 (0.06)
Zn	2.5	105	18.47 (0.09)	17.8 (0.1)	17.13 (0.07)	1.8 (0.06)

Table A1. 15. Average polycyclic aromatic hydrocarbon (PAH) concentrations in µg/L (standard error) for influent waters (clean water and influent stormwater runoff) and triplicate effluent waters from each of the three treatment depths plus the clean water control for Water Year 4. One-half of the value of the detection limit was substituted for the value of non-detects in calculating means; used when the compound was detected in at least one replicate for the treatment. Values following '<' are equal to the detection limit.

PAHs	Clean Water	Influent	6"	12"	18"	18" CWC
1-Methylnaphthalene	0.002	0.114	<0.001	<0.001	<0.001	0.002 (0)
2-Chloronaphthalene	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001
2-Methylnaphthalene	0.003	0.038	0.001 (8e-04)	<0.001	<0.001	0.002 (3e-04)
Naphthalene	0.009	0.109	0.002 (3e-04)	0.002 (0)	0.002 (0)	0.007 (0.002)
Acenaphthene	<0.003	0.007	0.004 (0.001)	0.005 (0.002)	0.005 (3e-04)	<0.003
Acenaphthylene	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002
Anthracene	<0.001	0.006	<0.001	<0.001	<0.001	<0.001
Carbazole	0.003	<0.001	<0.001	<0.001	<0.001	<0.001
Dibenzofuran	0.002	<0.002	<0.002	<0.002	<0.002	<0.002
Fluorene	<0.002	0.03	0.008 (0.007)	0.02 (3e-04)	0.01 (0.005)	<0.002
Phenanthrene	0.003	0.034	0.001 (5e-04)	0.002 (5e-04)	0.003 (0)	0.002 (5e-04)
Benz[a]anthracene	<8e-04	0.007	<8e-04	<8e-04	<8e-04	<8e-04
Chrysene	<0.001	0.018	<0.001	<0.001	<0.001	<0.001
Fluoranthene	<0.002	0.047	<0.002	<0.002	<0.002	<0.002
Pyrene	<0.001	0.053	<0.001	<0.001	<0.001	0.001 (5e-04)
Benzo(a)pyrene	<0.003	0.008	<0.003	<0.003	<0.003	<0.003
Benzo(b)fluoranthene	<5e-04	0.013	<5e-04	<5e-04	<5e-04	<5e-04
Benzo(j)fluoranthene	<0.002	0.003	<0.002	<0.002	<0.002	<0.002
Benzo(k)fluoranthene	<0.004	0.005	<0.004	<0.004	<0.004	<0.004
Dibenzo(a,h)anthracene	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001
Perylene	<0.007	<0.007	<0.007	<0.007	<0.007	<0.007
Benzo(ghi)perylene	<0.002	0.027	<0.002	<0.002	<0.002	<0.002
Indeno(1,2,3-cd)pyrene	<0.001	0.008	<0.001	<0.001	<0.001	<0.001
Total PAHs	**0.039**	**0.534**	**0.033 (0.007)**	**0.045 (0.002)**	**0.037 (0.005)**	**0.032 (0.002)**

Table A1. 16. Average water chemistry values (standard error) for influent waters (clean water and influent stormwater runoff) and triplicate effluent waters from each of the three treatment depths plus the clean water control for Water Year 5. One-half of the value of the detection limit was substituted for the value of non-detects in calculating means; used when the compound was detected in at least one replicate for the treatment. Values following '<' are equal to the detection limit. n.m. = not measured for this event.

Compound	Units	Clean Water	Influent	6"	12"	18"	18" CWC
Conventional							
Dissolved Organic Carbon	mg/L	<0.08	4.2	3.23 (0.03)	3.13 (0.09)	5 (2)	1.47 (0.03)
Total Suspended Solids	mg/L	<0.5	37	0.8 (0.1)	0.4 (0.1)	0.7 (0.1)	0.4 (0.1)
Turbidity	NTU	0.08	71	14.4 (0.3)	10 (1)	9.2 (0.7)	1.6 (0.2)
Conductivity	μS/cm	1492	146.6	144.2 (0.3)	142 (0.2)	139.1 (0.7)	1484 (3)
pH	n.a.	7.584	7.545	7.41 (0.02)	7.316 (0.006)	7.27 (0.01)	7.248 (0.005)
Alkalinity	mg/L as $CaCO_3$	42	48.7	43.6 (0.3)	43.7 (0.8)	42 (1)	37 (1)
Temperature	°F	n.m.	33.8	38 (0.7)	34.2 (0.4)	34.9 (0.2)	34.8 (0.2)
Dissolved Calcium	µg/L	7.78	14.3	15.8 (0.1)	15.7 (0.2)	15.93 (0.07)	9.58 (0.03)
Dissolved Magnesium	µg/L	23.9	0.609	0.97 (0.01)	1.08 (0.03)	1.09 (0.01)	23.53 (0.03)
Dissolved Sodium	µg/L	313	19.3	14.8 (0.1)	13.8 (0.06)	12.73 (0.09)	306 (1)
Nutrients							
Nitrate/Nitrite	mg/L	<0.003	0.25	0.437 (0.009)	0.55 (0.01)	0.62 (0.01)	0.32 (0.01)
Orthophosphate, as P	mg/L	0.01	0.01	0.063 (0.003)	0.09 (0)	0.11 (0.006)	0.167 (0.009)
Microbiology							
Fecal Coliform by MF	CFU/100 mL	10	10	337 (100)	180 (60)	237 (100)	10 (0)

MF = membrane filtration

Table A1. 17. Average concentrations of dissolved and total metals in μg/L (standard error) for influent waters (clean water and influent stormwater runoff) and triplicate effluent waters from each of the three treatment depths plus the clean water control for Water Year 5. One-half of the value of the detection limit was substituted for the value of non-detects in calculating means; used when the compound was detected in at least one replicate for the treatment. Values following '<' are equal to the detection limit.

Compound	Clean Water	Influent	6"	12"	18"	18" CWC
Dissolved As	<0.05	0.4	<0.05	<0.05	<0.05	<0.05
Dissolved Cd	<0.05	<0.05	<0.05	<0.05	<0.05	<0.05
Dissolved Cu	<0.05	6.8	5.2 (0.1)	4.4 (0.3)	4.5 (0.2)	2 (0.4)
Dissolved Pb	<0.079	<0.079	<0.079	<0.079	<0.079	<0.079
Dissolved Ni	<0.2	<0.2	<0.2	<0.2	<0.2	<0.2
Dissolved Zn	2.4	23.3	10.2 (0.3)	9 (0.3)	10 (1)	0.5 (0.06)
As	<0.05	1.2	<0.05	<0.05	<0.05	<0.05
Cd	<0.05	<0.05	<0.05	<0.05	<0.05	<0.05
Cu	<0.2	30.5	8.4 (0.8)	6.9 (0.6)	6.9 (0.4)	4.4 (0.7)
Pb	<0.079	<0.079	<0.079	<0.079	<0.079	<0.079
Ni	<0.2	1.8	<0.2	<0.2	<0.2	<0.2
Zn	3	135	22.3 (0.3)	17 (1)	16.3 (0.9)	0.8 (0.2)

Table A1. 18. Average polycyclic aromatic hydrocarbon (PAH) concentrations in μg/L (standard error) for influent waters (clean water and influent stormwater runoff) and triplicate effluent waters from each of the three treatment depths plus the clean water control for Water Year 5. One-half of the value of the detection limit was substituted for the value of non-detects in calculating means; used when the compound was detected in at least one replicate for the treatment. Values following '<' are equal to the detection limit.

PAHs	Clean Water	Influent	6"	12"	18"	18" CWC
1-Methylnaphthalene	0.013	0.022	0.013 (0.004)	0.017 (3e-04)	0.017 (3e-04)	0.01 (0.004)
2-Chloronaphthalene	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001
2-Methylnaphthalene	0.02	0.031	0.015 (0.006)	0.022 (3e-04)	0.021 (6e-04)	0.015 (0.007)
Naphthalene	0.018	0.035	0.012 (0.003)	0.016 (3e-04)	0.016 (6e-04)	0.014 (0.003)
Acenaphthene	<0.003	<0.003	<0.003	<0.003	<0.003	<0.003
Acenaphthylene	<0.002	0.004	<0.002	<0.002	<0.002	<0.002
Anthracene	<0.001	0.009	0.001 (4e-04)	<0.001	<0.001	<0.001
Carbazole	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001
Dibenzofuran	<0.002	0.008	<0.002	0.002 (0)	0.002 (0)	<0.002
Fluorene	<0.002	0.01	0.002 (3e-04)	0.002 (0)	0.002 (3e-04)	<0.002
Phenanthrene	0.002	0.058	0.004 (3e-04)	0.003 (3e-04)	0.003 (0)	0.002 (5e-04)
Benz[a]anthracene	<8e-04	0.015	0.001 (0)	<8e-04	<8e-04	<8e-04
Chrysene	<9e-04	0.035	0.002 (0)	0.001 (2e-04)	0.001 (2e-04)	<9e-04
Fluoranthene	<0.002	0.085	0.007 (3e-04)	0.004 (9e-04)	0.004 (3e-04)	<0.002
Pyrene	<0.001	0.117	0.007 (3e-04)	0.005 (7e-04)	0.004 (0)	<0.001
Benzo(a)pyrene	<0.002	0.018	<0.002	<0.002	<0.002	<0.002
Benzo(b)fluoranthene	<5e-04	0.027	0.002 (0)	0.001 (2e-04)	0 (2e-04)	<5e-04
Benzo(j)fluoranthene	<0.002	0.007	<0.002	<0.002	<0.002	<0.002
Benzo(k)fluoranthene	<0.003	0.009	<0.003	<0.003	<0.003	<0.003
Dibenzo(a,h)anthracene	<0.001	0.002	<0.001	<0.001	<0.001	<0.001
Perylene	<0.006	0.008	<0.006	<0.006	<0.006	<0.006
Benzo(ghi)perylene	<0.001	0.052	0.003 (3e-04)	0.002 (0)	0.002 (0)	<0.001
Indeno(1,2,3-cd)pyrene	<0.001	0.019	0.001 (3e-04)	<0.001	<0.001	<0.001
Total PAHs	**0.07**	**0.574**	**0.082 (0.008)**	**0.087 (0.001)**	**0.083 (0.001)**	**0.057 (0.008)**

Table A1. 19. Average water chemistry values (standard error) for influent waters (clean water and influent stormwater runoff) and triplicate effluent waters from each of the three treatment depths plus the clean water control for Water Year 6. One-half of the value of the detection limit was substituted for the value of non-detects in calculating means; used when the compound was detected in at least one replicate for the treatment. Values following '<' are equal to the detection limit. n.m. = not measured for this event.

Compound	Units	Clean Water	Influent	6"	12"	18"	18" CWC
Conventional							
Dissolved Organic Carbon	mg/L	<0.08	27.4	18.6 (0.3)	16 (1)	15.5 (0.7)	1.67 (0.07)
Total Suspended Solids	mg/L	0.8	57.2	1 (0.2)	0.4 (0.2)	0.5 (0.1)	0.8 (0.1)
Turbidity	NTU	0.09	58.7	1.5 (0.2)	0.7 (0.2)	0.8 (0.1)	2.9 (0.7)
Conductivity	µS/cm	1515	3450	3500 (0)	3513 (7)	3530 (20)	1528 (0.6)
pH	n.a.	7.724	7.647	7.44 (0.03)	7.28 (0.02)	7.201 (0.003)	7.33 (0.02)
Alkalinity	mg/L as $CaCO_3$	54.5	67.2	59 (0.3)	56.6 (0.4)	54.1 (0.4)	35.2 (0.3)
Temperature	°F	n.m.	37.2	39.6 (0.7)	37.07 (0.07)	37.3 (0.1)	37.5 (0.3)
Dissolved Calcium	µg/L	8.41	56.5	40 (1)	34 (2)	32 (1)	9.6 (0.1)
Dissolved Magnesium	µg/L	22.6	1.55	1.18 (0.04)	1.11 (0.07)	1.07 (0.04)	22.1 (0.3)
Dissolved Sodium	µg/L	228	583	600 (10)	578 (4)	574 (3)	226 (2)
Nutrients							
Nitrate/Nitrite	mg/L	0.12	1.15	1.79 (0.05)	1.97 (0.05)	2.14 (0.04)	0.37 (0.02)
Orthophosphate, as P	mg/L	<0.01	<0.01	<0.01	0.017 (0.003)	0.02 (0)	0.143 (0.003)
Microbiology							
Fecal Coliform by MF	CFU/100 mL	10	200	40 (30)	20 (6)	20 (6)	20 (10)

MF = membrane filtration

Table A1. 20. Average concentrations of dissolved and total metals in μg/L (standard error) for influent waters (clean water and influent stormwater runoff) and triplicate effluent waters from each of the three treatment depths plus the clean water control for Water Year 6. One-half of the value of the detection limit was substituted for the value of non-detects in calculating means; used when the compound was detected in at least one replicate for the treatment. Values following '<' are equal to the detection limit.

Compound	Clean Water	Influent	6"	12"	18"	18" CWC
Dissolved As	<0.05	0.4	<0.05	<0.05	<0.05	<0.05
Dissolved Cd	<0.05	<0.05	<0.05	<0.05	<0.05	<0.05
Dissolved Cu	<0.05	21.3	10 (0.2)	7.2 (0.9)	8.3 (0.3)	2.17 (0.03)
Dissolved Pb	<0.079	<0.079	<0.079	<0.079	<0.079	<0.079
Dissolved Ni	<0.2	1.2	<0.2	<0.2	<0.2	<0.2
Dissolved Zn	<0.19	19.2	2.8 (0.06)	2.5 (0.2)	2 (0.1)	<0.19
As	<0.05	0.9	<0.05	<0.05	<0.05	<0.05
Cd	<0.05	0.3	<0.05	<0.05	<0.05	<0.05
Cu	<0.2	58.4	12 (0.2)	8.6 (0.9)	10 (0.5)	5.1 (0.2)
Pb	<0.079	3.5	<0.079	<0.079	<0.079	<0.079
Ni	<0.2	4	<0.2	0.4 (0.3)	<0.2	<0.2
Zn	<0.19	151	5.7 (0.2)	5.3 (0.2)	4.9 (0.2)	<0.19

Table A1. 21. Average polycyclic aromatic hydrocarbon (PAH) concentrations in μg/L (standard error) for influent waters (clean water and influent stormwater runoff) and triplicate effluent waters from each of the three treatment depths plus the clean water control for Water Year 6. One-half of the value of the detection limit was substituted for the value of non-detects in calculating means; used when the compound was detected in at least one replicate for the treatment. Values following '<' are equal to the detection limit.

PAHs	Clean Water	Influent	6"	12"	18"	18" CWC
1-Methylnaphthalene	0.003	0.02	0.021 (0.002)	0.019 (0.003)	0.017 (7e-04)	0.002 (3e-04)
2-Chloronaphthalene	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001
2-Methylnaphthalene	0.003	0.015	0.005 (0.003)	0.007 (0.005)	0.003 (6e-04)	0.002 (3e-04)
Naphthalene	0.007	0.046	0.01 (0.002)	0.006 (3e-04)	0.006 (7e-04)	0.007 (7e-04)
Acenaphthene	<0.003	<0.003	0.003 (0.002)	<0.003	<0.003	<0.003
Acenaphthylene	<0.002	0.004	<0.002	<0.002	<0.002	<0.002
Anthracene	<0.001	0.007	<0.001	<0.001	<0.001	<0.001
Carbazole	<0.001	0.008	0.001 (8e-04)	<0.001	0.001 (2e-04)	<0.001
Dibenzofuran	0.002	0.007	0.002 (0.001)	0.003 (3e-04)	0.003 (3e-04)	0.002 (6e-04)
Fluorene	<0.002	0.021	<0.002	<0.002	0.012 (6e-04)	<0.002
Phenanthrene	0.002	0.062	0.001 (5e-04)	0.002 (0)	0.002 (0)	0.001 (5e-04)
Benz[a]anthracene	<8e-04	0.012	<8e-04	<8e-04	<8e-04	<8e-04
Chrysene	<9e-04	0.037	<9e-04	<9e-04	<9e-04	<9e-04
Fluoranthene	<0.002	0.084	<0.002	<0.002	<0.002	<0.002
Pyrene	<0.001	0.117	<0.001	<0.001	<0.001	<0.001
Benzo(a)pyrene	<0.002	0.016	<0.002	<0.002	<0.002	<0.002
Benzo(b)fluoranthene	<5e-04	0.033	<5e-04	<5e-04	<5e-04	<5e-04
Benzo(j)fluoranthene	<0.002	0.009	<0.002	<0.002	<0.002	<0.002
Benzo(k)fluoranthene	<0.003	0.013	<0.003	<0.003	<0.003	<0.003
Dibenzo(a,h)anthracene	<0.001	0.002	<0.001	<0.001	<0.001	<0.001
Perylene	<0.006	<0.006	<0.006	<0.006	<0.006	<0.006
Benzo(ghi)perylene	<0.001	0.085	<0.001	<0.001	<0.001	<0.001
Indeno(1,2,3-cd)pyrene	<0.001	0.025	<0.001	<0.001	<0.001	<0.001
Total PAHs	**0.033**	**0.628**	**0.058 (0.005)**	**0.053 (0.006)**	**0.058 (0.001)**	**0.03 (0.001)**

Table A1. 22. Average water chemistry values (standard error) for influent waters (clean water and influent stormwater runoff) and triplicate effluent waters from each of the three treatment depths plus the clean water control for Water Year 7. One-half of the value of the detection limit was substituted for the value of non-detects in calculating means; used when the compound was detected in at least one replicate for the treatment. Values following '<' are equal to the detection limit. n.m. = not measured for this event.

Compound	Units	Clean Water	Influent	6"	12"	18"	18" CWC
Conventional							
Dissolved Organic Carbon	mg/L	<0.08	14.3	16.5 (0.1)^{a}	15 (0.1)^{ab}	13.9 (0.7)^{b}	3.57 (0.03)
Total Suspended Solids	mg/L	0.6	45.6	3 (1)	0.8 (0.6)	5 (2)	2.1 (0.1)
Turbidity	NTU	0.54	51.4	7 (3)	3.2 (0.3)	9 (2)	3.4 (0.8)
Conductivity	S/cm	1500	165.2	171 (2)	167.5 (0.4)	165 (2)	1514 (1)
pH	n.a.	7.68	7.465	7.26 (0.04)	7.12 (0.03)	7.15 (0.04)	7.43 (0.02)
Alkalinity	mg/L as $CaCO_3$	63.9	58.7	50 (3)	56 (6)	44 (1)	59 (1)
Temperature	F	n.m.	37	41.8 (0.8)	36 (1)	37 (1)	37.8 (0.4)
Dissolved Calcium	mg/L	5.72	18.4	17.5 (0.1)^{a}	16.9 (0.3)^{ab}	15.3 (0.2)^{b}	7.3 (0.09)
Dissolved Magnesium	mg/L	19	0.816	1 (0.03)^{a}	0.936 (0.007)^{ab}	0.77 (0.04)^{b}	18.2 (0.1)
Dissolved Sodium	mg/L	224	12.9	11.53 (0.07)^{a}	12.3 (0.1)^{ab}	14 (0.5)^{b}	225.3 (0.9)
Nutrients							
Nitrate/Nitrite	mg/L	0.312	1.2	2.66 (0.02)^{a}	3.04 (0.09)^{ab}	3.3 (0.1)^{b}	1.24 (0.08)
Orthophosphate, as P	mg/L	0.01	0.01	0.037 (0.003)	0.053 (0.003)	0.053 (0.009)	0.18 (0.01)
Microbiology							
Fecal Coliform by MF	CFU/100 mL	n.m.	n.m.	n.m.	n.m.	n.m.	n.m.

MF = membrane filtration

Table A1. 23. Average concentrations of dissolved and total metals in μg/L (standard error) for influent waters (clean water and influent stormwater runoff) and triplicate effluent waters from each of the three treatment depths plus the clean water control for Water Year 7. One-half of the value of the detection limit was substituted for the value of non-detects in calculating means; used when the compound was detected in at least one replicate for the treatment. Values following '<' are equal to the detection limit.

Compound	Clean Water	Influent	6"	12"	18"	18" CWC
Dissolved As	<0.05	1	<0.05	<0.05	<0.05	<0.05
Dissolved Cd	<0.05	<0.05	<0.05	<0.05	<0.05	<0.05
Dissolved Cu	<0.05	23.5	11.9 (0.5)	8 (0.1)	10 (1)	2.1 (0.2)
Dissolved Pb	<0.079	<0.079	<0.079	<0.079	<0.079	<0.079
Dissolved Ni	<0.2	2.8	2.1 (0.1)	1.7 (0.7)	1.6 (0.3)	0.4 (0.1)
Dissolved Zn	<0.19	45.3	10 (5)	6.2 (0.6)	14 (4)	0.6 (0.3)
As	<0.05	1.2	<0.05	<0.05	<0.05	<0.05
Cd	<0.05	<0.05	<0.05	<0.05	<0.05	<0.05
Cu	<0.2	42.5	14 (2)	8.93 (0.03)	11 (2)	3.7 (0.3)
Pb	<0.079	3	<0.079	<0.079	<0.079	<0.079
Ni	<0.2	4.4	2.2 (0.2)	2.2 (0.4)	2.2 (0.1)	0.8 (0.3)
Zn	<0.19	114	15 (7)	6.8 (0.3)	18 (6)	0.6 (0.3)

Table A1. 24. Average polycyclic aromatic hydrocarbon (PAH) concentrations in μg/L (standard error) for influent waters (clean water and influent stormwater runoff) and triplicate effluent waters from each of the three treatment depths plus the clean water control for Water Year 7. One-half of the value of the detection limit was substituted for the value of non-detects in calculating means; used when the compound was detected in at least one replicate for the treatment. Values following '<' are equal to the detection limit.

PAHs	Clean Water	Influent	6"	12"	18"	18" CWC
1-Methylnaphthalene	0.003	0.021	0.014 (9e-04)	0.015 (3e-04)	0.014 (9e-04)	0.002 (0)
2-Methylnaphthalene	0.006	0.014	0.004 (7e-04)	0.003 (3e-04)	0.004 (6e-04)	0.003 (3e-04)
Naphthalene	0.011	0.039	0.008 (0.002)	0.007 (0.001)	0.009 (9e-04)	0.006 (3e-04)
Acenaphthene	<0.003	<0.003	<0.003	<0.003	<0.003	<0.003
Acenaphthylene	<0.002	0.004	0.001 (3e-04)	<0.002	<0.002	<0.002
Anthracene	<0.001	0.007	0.001 (5e-04)	<0.001	<0.001	<0.001
Carbazole	0.002	<0.001	<0.001	0.001 (2e-04)	0.002 (8e-04)	0.001 (2e-04)
Dibenzofuran	0.004	0.008	0.004 (0.002)	0.006 (0)	0.006 (3e-04)	0.003 (3e-04)
Fluorene	<0.002	<0.002	0.003 (0.002)	<0.002	<0.002	<0.002
Phenanthrene	0.003	0.034	0.003 (0.002)	0.002 (3e-04)	0.004 (0.002)	0.001 (2e-04)
Benz[a]anthracene	<8e-04	0.01	0.001 (5e-04)	<8e-04	0.001 (5e-04)	<8e-04
Chrysene	<9e-04	0.029	0.002 (0.001)	<9e-04	0.003 (0.001)	<9e-04
Fluoranthene	<0.002	0.049	0.003 (0.002)	<0.002	0.005 (0.002)	<0.002
Pyrene	<0.001	0.081	0.005 (0.003)	0.002 (3e-04)	0.009 (0.004)	<0.001
Benzo(a)pyrene	<0.002	0.014	<0.002	<0.002	<0.002	<0.002
Benzo(b)fluoranthene	<5e-04	0.018	0.001 (9e-04)	0 (1e-04)	0.002 (5e-04)	<5e-04
Benzo(j)fluoranthene	<0.002	0.005	<0.002	<0.002	<0.002	<0.002
Benzo(k)fluoranthene	<0.003	0.01	<0.003	<0.003	<0.003	<0.003
Dibenzo(a,h)anthracene	<0.001	0.002	<0.001	<0.001	<0.001	<0.001
Perylene	<0.006	<0.006	<0.006	<0.006	<0.006	<0.006
Benzo(ghi)perylene	<0.001	0.044	0.002 (0.002)	<0.001	0.004 (0.002)	<0.001
Indeno(1,2,3-cd)pyrene	<0.001	0.012	0.001 (5e-04)	<0.001	0.002 (5e-04)	<0.001
Total PAHs	**0.044**	**0.407**	**0.062 (0.006)**	**0.05 (0.001)**	**0.075 (0.006)**	**0.03 (6e-04)**

Table A1. 25 Average water chemistry values (standard error) for influent waters (clean water and influent stormwater runoff) and triplicate effluent waters from each of the three treatment depths plus the clean water control for Water Year 8. One-half of the value of the detection limit was substituted for the value of non-detects in calculating means; used when the compound was detected in at least one replicate for the treatment. Values following '<' are equal to the detection limit. n.m. = not measured for this event.

Compound	Units	Clean Water	Influent	6"	12"	18"	18" CWC
Conventional							
Dissolved Organic Carbon	mg/L	<0.08	3.2	2.3 (0.06)	2.3 (0.2)	2.5 (0.2)	1.17 (0.03)
Total Suspended Solids	mg/L	<0.5	15.2	0.4 (0.1)	1.5 (0.9)	5 (2)	0.6 (0.2)
Turbidity	NTU	0.02	24.7	2.3 (0.1)	4 (2)	9 (4)	1.8 (0.3)
Conductivity	S/cm	1499	123.3	114.4 (0.8)	110 (2)	112 (3)	1494 (5)
pH	n.a.	7.914	7.622	7.38 (0.01)	7.27 (0.03)	7.32 (0.09)	7.657 (0.005)
Alkalinity	mg/L as $CaCO_3$	72.2	56.1	48 (3)	50 (7)	44 (2)	77 (10)
Temperature	F	n.m.	n.m.	37 (0.3)	35 (1)	36.2 (0.8)	35.6 (0.6)
Dissolved Calcium	mg/L	7.64	19.6	17.27 (0.09)	16.6 (0.5)	17.8 (0.8)	7.49 (0.06)
Dissolved Magnesium	mg/L	19.5	0.463	0.97 (0.03)[a]	0.9 (0.04)[ab]	0.67 (0.08)[b]	18.6 (0.3)
Dissolved Sodium	mg/L	244	4.87	4.06 (0.02)	3.89 (0.04)	4 (0.3)	239.7 (0.3)
Nutrients							
Nitrate/Nitrite	mg/L	0.22	0.36	0.533 (0.007)	0.583 (0.009)	0.58 (0.02)	0.337 (0.009)
Orthophosphate, as P	mg/L	<0.01	<0.01	0.037 (0.003)	0.043 (0.007)	0.04 (0.01)	0.133 (0.003)
Microbiology							
Fecal Coliform by MF	CFU/100 mL	1	1020	200 (0)	169 (30)	531 (200)	1 (0)

MF = membrane filtration

Table A1. 26. Average concentrations of dissolved and total metals in μg/L (standard error) for influent waters (clean water and influent stormwater runoff) and triplicate effluent waters from each of the three treatment depths plus the clean water control for Water Year 8. One-half of the value of the detection limit was substituted for the value of non-detects in calculating means; used when the compound was detected in at least one replicate for the treatment. Values following '<' are equal to the detection limit.

Compound	Clean Water	Influent	6"	12"	18"	18" CWC
Dissolved As	4.48	<0.05	<0.05	<0.05	0.5 (0.4)	4.8 (0.4)
Dissolved Cd	<0.05	<0.05	<0.05	<0.05	<0.05	<0.05
Dissolved Cu	0.81	6.94	4.6 (0.3)	4.2 (0.5)	5 (1)	2.9 (0.1)
Dissolved Pb	<0.079	<0.079	<0.079	<0.079	<0.079	<0.079
Dissolved Ni	<0.2	1.32	1.01 (0.03)	1 (0.1)	1.02 (0.06)	0.7 (0.06)
Dissolved Zn	<0.19	15.6	3 (0.2)	4 (2)	9 (3)	<0.19
As	0.099	1.27	0.69 (0.01)	0.63 (0.02)	0.8 (0.1)	0.45 (0.02)
Cd	0.067	0.05	0.027 (0.002)	<0.05	0.04 (0.01)	<0.05
Cu	<0.2	18.1	6.3 (0.2)	5 (1)	9 (3)	2.5 (0.2)
Pb	<0.079	2.2	0.25 (0.009)	0.3 (0.1)	0.8 (0.4)	0.083 (0.005)
Ni	0.211	1.74	0.543 (0.006)	0.7 (0.2)	0.9 (0.2)	0.49 (0.02)
Zn	<0.19	51.9	4.4 (0.2)	7 (4)	19 (10)	<0.19

Table A1. 27. Average polycyclic aromatic hydrocarbon (PAH) concentrations in μg/L (standard error) for influent waters (clean water and influent stormwater runoff) and triplicate effluent waters from each of the three treatment depths plus the clean water control for Water Year 8. One-half of the value of the detection limit was substituted for the value of non-detects in calculating means; used when the compound was detected in at least one replicate for the treatment. Values following '<' are equal to the detection limit.

PAHs	Clean Water	Influent	6"	12"	18"	18" CWC
1-Methylnaphthalene	0.002	0.005	0.003 (0)	0.004 (3e-04)	0.004 (6e-04)	0.002 (0)
2-Methylnaphthalene	0.003	0.006	0.002 (0)	0.003 (6e-04)	0.004 (9e-04)	0.003 (0)
Naphthalene	0.006	0.011	0.005 (3e-04)	0.007 (0.002)	0.01 (0.002)	0.006 (6e-04)
Acenaphthene	<0.003	<0.003	<0.003	<0.003	<0.003	<0.003
Acenaphthylene	<0.002	0.002	<0.002	<0.002	<0.002	<0.002
Anthracene	<0.001	0.003	<0.001	<0.001	0.002 (5e-04)	<0.001
Carbazole	<0.001	0.007	<0.001	<0.001	0.002 (0.001)	<0.001
Dibenzofuran	<0.002	0.003	0.001 (3e-04)	0.002 (3e-04)	0.003 (3e-04)	<0.002
Fluorene	<0.002	0.005	<0.002	0.001 (3e-04)	0.002 (9e-04)	<0.002
Phenanthrene	0.003	0.017	0.002 (0)	0.003 (0.001)	0.007 (0.004)	0.002 (0)
Benz[a]anthracene	<8e-04	0.005	<8e-04	0.001 (5e-04)	0.002 (0.001)	<8e-04
Chrysene	0.002	0.014	0.002 (3e-04)	0.003 (0.001)	0.007 (0.003)	0.002 (0)
Fluoranthene	<0.002	0.026	<0.002	0.003 (0.002)	0.01 (0.005)	<0.002
Pyrene	0.001	0.043	0.002 (0)	0.006 (0.004)	0.017 (0.009)	0.002 (0)
Benzo(a)pyrene	<0.002	0.006	<0.002	<0.002	0.003 (9e-04)	<0.002
Benzo(b)fluoranthene	0.001	0.01	0.001 (2e-04)	0.001 (9e-04)	0.004 (0.002)	0.001 (3e-05)
Benzo(j)fluoranthene	<0.002	0.003	<0.002	<0.002	0.001 (3e-04)	<0.002
Benzo(k)fluoranthene	<0.003	0.004	<0.003	<0.003	<0.003	<0.003
Dibenzo(a,h)anthracene	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001
Perylene	<0.006	<0.006	<0.006	<0.006	<0.006	<0.006
Benzo(ghi)perylene	<0.001	0.021	<0.001	0.002 (0.002)	0.008 (0.004)	<0.001
Indeno(1,2,3-cd)pyrene	<0.001	0.005	<0.001	0.001 (5e-04)	0.002 (0.001)	<0.001
Total PAHs	**0.033**	**0.201**	**0.031 (6e-04)**	**0.048 (0.006)**	**0.1 (0.01)**	**0.033 (6e-04)**

Table A1. 28 Average water chemistry values (standard error) for influent waters (clean water and influent stormwater runoff) and triplicate effluent waters from each of the three treatment depths plus the clean water control for Water Year 9. One-half of the value of the detection limit was substituted for the value of non-detects in calculating means; used when the compound was detected in at least one replicate for the treatment. Values following '<' are equal to the detection limit. n.m. = not measured for this event.

Compound	Units	Clean Water	Influent	6"	12"	18"	18" CWC
Conventional							
Dissolved Organic Carbon	mg/L	1.41	3.44	3.1 (0.1)a	4.12 (0.04)b	4.5 (0.1)b	2.2 (0.3)
Total Suspended Solids	mg/L	1.6	38.8	1.2 (0.2)	1.5 (0.5)	1.4 (0.6)	1.5 (0.3)
Turbidity	NTU	0.03	36.3	6.5 (0.1)	6 (0.4)	6.4 (0.4)	2.4 (0.4)
Conductivity	S/cm	1478	82.8	120 (2)	142 (2)	155 (3)	1502 (2)
pH	n.a.	7.836	7.605	7.91 (0.01)	7.837 (0.006)	7.842 (0.006)	7.66 (0.04)
Alkalinity	mg/L as $CaCO_3$	78.9	37.5	36 (3)	40 (2)	41.1 (0.6)	63 (2)
Temperature	F	n.m.	37.2	38.6 (0.3)	35.1 (0.5)	35 (1)	35.1 (0.3)
Dissolved Calcium	mg/L	7.22	11.7	0.23 (0.02)a	0.33 (0.02)b	0.43 (0.03)c	8 (0.04)
Dissolved Magnesium	mg/L	23.6	0.349	0.3 (0.3)	0.018 (0.002)	0.03 (0.006)	22.3 (0.2)
Dissolved Sodium	mg/L	321	2.79	23.6 (0.3)a	28.7 (0.4)b	30.7 (0.3)c	320 (1)
Nutrients							
Nitrate/Nitrite	mg/L	0.265	0.2	0.328 (0.003)a	0.37 (0.010)a	0.479 (0.007)b	0.43 (0.05)
Orthophosphate, as P	mg/L	<0.01	<0.01	0.12 (0.006)a	0.257 (0.009)b	0.41 (0.02)c	0.147 (0.003)
Microbiology							
Fecal Coliform by MF	CFU/100 mL	n.m.	n.m.	n.m.	n.m.	n.m.	n.m.

MF = membrane filtration

Table A1. 29. Average concentrations of dissolved and total metals in μg/L (standard error) for influent waters (clean water and influent stormwater runoff) and triplicate effluent waters from each of the three treatment depths plus the clean water control for Water Year 9. One-half of the value of the detection limit was substituted for the value of non-detects in calculating means; used when the compound was detected in at least one replicate for the treatment. Values following '<' are equal to the detection limit.

Compound	Clean Water	Influent	6"	12"	18"	18" CWC
Dissolved As	0.367	0.577	1.18 (0.02)a	1.46 (0.06)b	2 (0.03)c	0.92 (0.06)
Dissolved Cd	<0.05	<0.05	<0.05	<0.05	<0.05	<0.05
Dissolved Cu	<0.05	3.74	5.1 (0.2)	5.3 (0.6)	6.28 (0.05)	1.7 (0.2)
Dissolved Pb	<0.079	<0.079	<0.079	<0.079	<0.079	<0.079
Dissolved Ni	<0.2	<0.2	<0.2	0.5 (0.2)	0.68 (0.03)	0.5 (0.4)
Dissolved Zn	1.12	22.3	4.5 (0.9)	4.7 (0.6)	6 (1)	1.5 (0.4)
As	0.45	0.87	1.18 (0.01)a	1.57 (0.04)b	2.12 (0.07)c	0.94 (0.04)
Cd	<0.05	<0.05	<0.05	<0.05	<0.05	<0.05
Cu	0.583	17.7	6.8 (0.1)a	7.5 (0.5)ab	8.8 (0.3)b	3.1 (0.3)
Pb	<0.079	2.89	<0.079	<0.079	<0.079	<0.079
Ni	<0.2	1.63	<0.2a	0.59 (0.04)a	1.04 (0.05)b	0.3 (0.2)
Zn	0.683	84.7	7.3 (0.1)a	6.8 (0.2)ab	6.9 (0.2)b	2.1 (0.6)

Table A1. 30. Average polycyclic aromatic hydrocarbon (PAH) concentrations in µg/L (standard error) for influent waters (clean water and influent stormwater runoff) and triplicate effluent waters from each of the three treatment depths plus the clean water control for Water Year 9. One-half of the value of the detection limit was substituted for the value of non-detects in calculating means; used when the compound was detected in at least one replicate for the treatment. Values following '<' are equal to the detection limit.

PAHs	Clean Water	Influent	6"	12"	18"	18" CWC
1-Methylnaphthalene	0.003	0.007	0.004 (6e-04)	0.004 (3e-04)	0.005 (0)	0.005 (0.002)
2-Methylnaphthalene	0.004	0.011	0.004 (3e-04)	0.004 (3e-04)	0.004 (3e-04)	0.005 (0.002)
Naphthalene	0.007	0.02	0.005 (3e-04)	0.006 (0)	0.005 (6e-04)	0.004 (3e-04)
Acenaphthene	<0.003	<0.003	<0.003	<0.003	<0.003	<0.003
Acenaphthylene	<0.002	0.004	<0.002	<0.002	<0.002	<0.002
Anthracene	<0.001	0.004	0.002 (5e-04)	0.001 (2e-04)	0.001 (8e-04)	<0.001
Carbazole	<0.001	0.006	<0.001	<0.001	<0.001	0.001 (2e-04)
Dibenzofuran	0.002	0.005	0.002 (3e-04)	0.002 (0)	0.003 (3e-04)	0.001 (3e-04)
Fluorene	<0.002	0.007	0.002 (3e-04)	<0.002	<0.002	0.001 (3e-04)
Phenanthrene	0.002	0.038	0.003 (3e-04)	0.002 (0)	0.003 (3e-04)	0.001 (5e-04)
Benz[a]anthracene	<8e-04	0.01	<8e-04	<8e-04	<8e-04	<8e-04
Chrysene	<9e-04	0.029	0.001 (3e-04)	0.001 (3e-05)	0.001 (0)	<9e-04
Fluoranthene	<0.002	0.056	0.002 (3e-04)	0.001 (3e-04)	0.001 (3e-04)	<0.002
Pyrene	<0.001	0.083	0.004 (3e-04)	0.004 (0.002)	0.002 (3e-04)	0.001 (2e-04)
Benzo(a)pyrene	<0.002	0.013	<0.002	<0.002	<0.002	<0.002
Benzo(b)fluoranthene	<5e-04	0.022	0.001 (0)	0.001 (7e-05)	0.001 (3e-05)	<5e-04
Benzo(j)fluoranthene	<0.002	0.005	<0.002	<0.002	<0.002	<0.002
Benzo(k)fluoranthene	<0.003	0.01	<0.003	<0.003	<0.003	<0.003
Dibenzo(a,h)anthracene	<0.001	0.002	<0.001	<0.001	<0.001	<0.001
Perylene	<0.006	<0.006	<0.006	<0.006	<0.006	<0.006
Benzo(ghi)perylene	<0.001	0.059	0.003 (3e-04)	0.02 (0.02)	0.002 (0)	<0.001
Indeno(1,2,3-cd)pyrene	<0.001	0.013	<0.001	<0.001	<0.001	<0.001
Total PAHs	**0.033**	**0.408**	**0.042 (0.001)**	**0.05 (0.02)**	**0.041 (0.001)**	**0.032 (0.003)**

Table A1. 31. Average water chemistry values (standard error) for influent waters (clean water and influent stormwater runoff) and triplicate effluent waters from each of the three treatment depths plus the clean water control for Water Year 10. One-half of the value of the detection limit was substituted for the value of non-detects in calculating means; used when the compound was detected in at least one replicate for the treatment. Values following '<' are equal to the detection limit. n.m. = not measured for this event.

Compound	Units	Clean Water	Influent	6"	12"	18"	18" CWC
Conventional							
Dissolved Organic Carbon	mg/L	<0.5	3.5	3.53 (0.03)	3.60 (0.07)	3.93 (0.09)	0.848 (0.04)
Total Suspended Solids	mg/L	0.8	48.8	<0.5	0.617 (0.2)	1.02 (0.5)	0.567 (0.3)
Turbidity	NTU	0.03	18.28	5.90 (3.4)	5.88 (0.1)	6.54 (0.9)	1.79 (0.4)
Conductivity	S/cm	1436	49.1	52.7 (0.8)	51.8 (1)	52.4 (0.4)	1465 (5.8)
pH	n.a.	7.64	6.93	6.90 (0.07)	6.67 (0.02)	6.66 (0.04)	7.40 (0.02)
Alkalinity	mg/L as $CaCO_3$	56.6	13	13.4 (0.3)	12 (0.3)	11.7 (0.06)	53 (0.2)
Temperature	F	n.m	n.m	38.6 (0.3)	37.9 (0.4)	37.4 (1)	36.3 (0.9)
Dissolved Calcium	mg/L	6.42	4.51	4.82 (0.1)a	3.24 (0.1)b	1.89 (0.4)c	7.42 (0.07)
Dissolved Magnesium	mg/L	20	0.535	0.532 (0.04)a	0.58 (0.05)a	0.29 (0.05)b	20.7 (0.2)
Dissolved Sodium	mg/L	256	3.76	4.82 (0.1)a	6.30 (0.3)a	8.69 (0.6)b	264 (1)
Nutrients							
Nitrate/Nitrite	mg/L	0.106	0.352	0.612 (0.02)a	0.684 (0.03)ab	0.803 (0.05)b	0.290 (0.07)
Orthophosphate, as P	mg/L	<0.004	0.008	0.041 (0.001)a	0.067 (0.002)a	0.095 (0.01)b	0.124 (0.003)
Microbiology							
Fecal Coliform by MF	CFU/100 mL	n.m.	n.m.	n.m.	n.m.	n.m.	n.m.

MF = membrane filtration

Table A1. 32. Average concentrations of dissolved and total metals in μg/L (standard error) for influent waters (clean water and influent stormwater runoff) and triplicate effluent waters from each of the three treatment depths plus the clean water control for Water Year 10. One-half of the value of the detection limit was substituted for the value of non-detects in calculating means; used when the compound was detected in at least one replicate for the treatment. Values following '<' are equal to the detection limit.

Compound	Clean Water	Influent	6"	12"	18"	18" CWC
Dissolved As	<0.2	0.355	0.512 (0.008)	0.499 (0.02)	0.593 (0.04)	0.362 (0.02)
Dissolved Cd	<0.2	<0.2	<0.2	<0.2	<0.2	<0.2
Dissolved Cu	0.346	10.1	6.79 (0.2)	5.82 (0.2)	6.20 (0.5)	1.90 (0.09)
Dissolved Pb	<0.5	<0.5	<0.5	<0.5	<0.5	<0.5
Dissolved Ni	<0.5	0.832	0.510 (0.009)	0.671 (0.3)	0.752 (0.03)	<0.5
Dissolved Zn	2.12	43.6	6.12 (0.9)	6.01 (0.1)	5.34 (0.5)	2.55 (0.5)
As	<0.2	0.768	0.675 (0.02)	0.684 (0.02)	0.751 (0.02)	0.510 (0.02)
Cd	<0.2	<0.2	<0.2	<0.2	<0.2	<0.2
Cu	1.31	31.7	9.39 (0.3)	8.37 (0.3)	8.31 (0.3)	3.23 (0.05)
Pb	<0.5	3.46	<0.5	<0.5	<0.5	<0.5
Ni	<0.5	2.42	0.864 (0.06)	1.32 (0.4)	1.16 (0.03)	0.705 (0.08)
Zn	1.24	116	8.07 (2)	7.54 (1)	5.97 (0.2)	3.71 (0.6)

Table A1. 33. Average polycyclic aromatic hydrocarbon (PAH) concentrations in μg/L (standard error) for influent waters (clean water and influent stormwater runoff) and triplicate effluent waters from each of the three treatment depths plus the clean water control for Water Year 10. One-half of the value of the detection limit was substituted for the value of non-detects in calculating means; used when the compound was detected in at least one replicate for the treatment. Values following '<' are equal to the detection limit.

PAHs	Clean Water	Influent	6"	12"	18"	18" CWC
1-Methylnaphthalene	0.004	0.009	0.003 (6e-04)	0.007 (6e-04)	0.002 (0)	0.003 (6e-04)
2-Methylnaphthalene	0.004	0.016	0.005 (3e-04)	0.004 (3e-04)	0.004 (0)	0.004 (0)
Naphthalene	0.008	0.018	0.007 (3e-04)	0.007 (6e-04)	0.007 (6e-04)	0.006 (3e-04)
Acenaphthene	0.01	0.01	0.01 (0)	0.01 (0)	0.01 (0)	0.01 (0)
Acenaphthylene	0.01	0.004	0.01 (0)	0.01 (0)	0.01 (0)	0.01 (0)
Anthracene	0.01	0.006	0.005 (0.003)	0.008 (0.002)	0.008 (0.002)	0.007 (0.003)
Carbazole	0.01	0.01	0.01 (0)	0.01 (0)	0.01 (0)	0.01 (0)
Dibenzofuran	<0.002	0.005	0.01 (0.003)	0.007 (0.003)	0.007 (0.003)	0.007 (0.003)
Fluorene	0.01	0.005	0.005 (0.003)	0.01 (0)	0.01 (0)	0.01 (0)
Phenanthrene	0.002	0.036	0.004 (9e-04)	0.005 (0.002)	0.006 (0.003)	0.005 (7e-04)
Benz[a]anthracene	0.001	0.012	0.001 (3e-04)	0.005 (0.003)	0.001 (0)	0.007 (0.003)
Chrysene	0.002	0.036	0.003 (7e-04)	0.004 (9e-04)	0.003 (3e-04)	0.004 (0.003)
Fluoranthene	<0.002	0.057	0.003 (6e-04)	0.01 (0.002)	0.005 (0.002)	0.005 (0.003)
Pyrene	0.002	0.089	0.004 (0)	0.008 (0.003)	0.005 (0.002)	0.005 (0.003)
Benzo(a)pyrene	0.01	0.016	0.01 (0)	0.01 (0)	0.01 (0)	0.01 (0)
Benzo(b)fluoranthene	0.001	0.023	0.002 (3e-04)	0.002 (6e-04)	0.002 (3e-04)	0.004 (0.003)
Benzo(j)fluoranthene	0.01	0.011	0.01 (0)	0.01 (0)	0.01 (0)	0.01 (0)
Benzo(k)fluoranthene	0.01	0.013	0.01 (0)	0.01 (0)	0.01 (0)	0.01 (0)
Dibenzo(a,h)anthracene	0.01	0.003	0.007 (0.003)	0.01 (0)	0.01 (0)	0.01 (0)
Perylene	0.01	0.01	0.01 (0)	0.01 (0)	0.01 (0)	0.01 (0)
Benzo(ghi)perylene	0.01	0.043	0.003 (7e-04)	0.003 (9e-04)	0.005 (0.003)	0.01 (0)
Indeno(1,2,3-cd)pyrene	0.01	0.013	0.002 (7e-04)	0.004 (0.003)	0.001 (0.003)	0.01 (0)
Total PAHs	**0.158**	**0.492**	**0.14**	**0.169**	**0.158**	**0.177**

Table A1. 34. Average water chemistry values (standard error) for influent waters (clean water and influent stormwater runoff) and triplicate effluent waters from each of the three treatment depths plus the clean water control for Water Year 11. One-half of the value of the detection limit was substituted for the value of non-detects in calculating means; used when the compound was detected in at least one replicate for the treatment. Values following '<' are equal to the detection limit. n.m. = not measured for this event.

Compound	Units	Clean Water	Influent	6"	12"	18"	18" CWC
Conventional							
Dissolved Organic Carbon	mg/L	0.57	64.41	53.17 (3.4)	55.57 (4.2)	54.66 (3.6)	1.89 (0.1)
Total Suspended Solids	mg/L	<0.8	126	2.67 (0.7)	2 (0.6)	3.33 (0.9)	<0.8
Turbidity	NTU	0.02	77.6	1.66 (1)	2.56 (0.5)	1.39 (0.2)	2.13 (0.07)
Conductivity	S/cm	937	584	526.7 (11)	529 (8.5)	519.7 (8.8)	862.3 (3.3)
pH	n.a.	7.359	7.458	7.393 (0.02)^{a}	7.249 (0.06)^{ab}	7.137 (0.04)^{bc}	7.54 (0.02)
Alkalinity	mg/L as $CaCO_3$	47.1	116	108.7 (0.9)	105 (1.7)	103.7 (0.7)	47 (0.2)
Temperature	F	n.m.	n.m.	n.m.	n.m.	n.m.	n.m.
Dissolved Calcium	mg/L	3.63	6.89	24.7 (1.5)	23.7 (0.06)	24.3 (0.7)	4.25 (0.04)
Dissolved Magnesium	mg/L	11.9	1.02	2.62 (0.09)^{a}	2.01 (0.2)^{b}	1.92 (0.03)^{b}	12 (0.06)
Dissolved Sodium	mg/L	136	41.3	56.4 (5)	53.8 (5.4)	56.2 (4.6)	139 (1.5)
Nutrients							
Nitrate/Nitrite	mg/L	0.193	0.063	0.216 (0.08)	0.15 (0.08)	0.288 (0.09)	0.292 (0.009)
Orthophosphate, as P	mg/L	<0.004	0.073	0.02 (0.0009)^{a}	0.03 (0.002)^{b}	0.042 (0.003)^{c}	0.126 (0.003)
Microbiology							
Fecal Coliform by MF	CFU/100 mL	n.m.	n.m.	n.m.	n.m.	n.m.	n.m.

MF = membrane filtration

Table A1. 35. Average concentrations of dissolved and total metals in μg/L (standard error) for influent waters (clean water and influent stormwater runoff) and triplicate effluent waters from each of the three treatment depths plus the clean water control for Water Year 11. One-half of the value of the detection limit was substituted for the value of non-detects in calculating means; used when the compound was detected in at least one replicate for the treatment. Values following '<' are equal to the detection limit.

Compound	Clean Water	Influent	6"	12"	18"	18" CWC
Dissolved As	0.038	0.473	0.298 (0.02)	0.318 (0.03)	0.34 (0.009)	0.359 (0.005)
Dissolved Cd	0.039	0.047	<0.03	<0.03	<0.03	<0.03
Dissolved Cu	<0.173	7.02	5.68 (0.6)	4.41 (0.2)	4.54 (0.7)	2.59 (0.6)
Dissolved Pb	<0.05	0.086	<0.05	0.035 (0.009)	<0.05	0.054 (0.003)
Dissolved Ni	0.199	0.78	0.532 (0.01)	0.89 (0.2)	0.71 (0.1)	0.05 (0.03)
Dissolved Zn	<2.92	33.3	2.13 (0.7)	3.18 (0.05)	<2.92	<2.92
As	<0.0373	1.64	0.324 (0.04)	0.352 (0.02)	0.371 (0.01)	0.405 (0.02)
Cd	0.038	0.202	<0.03	<0.03	<0.03	<0.03
Cu	0.276	45.4	6.64 (0.4)	5.58 (0.5)	5.62 (0.9)	3.27 (0.7)
Pb	<0.05	7.54	0.405 (0.2)	0.154 (0.02)	0.074 (0.02)	0.134 (0.008)
Ni	0.161	5.86	0.758 (0.2)	1.06 (0.4)	0.818 (0.1)	0.546 (0.04)
Zn	<2.92	198	6.89 (1.8)	4.72 (0.8)	2.99 (0.8)	<2.92

Table A1. 36. Average polycyclic aromatic hydrocarbon (PAH) concentrations in μg/L (standard error) for influent waters (clean water and influent stormwater runoff) and triplicate effluent waters from each of the three treatment depths plus the clean water control for Water Year 11. One-half of the value of the detection limit was substituted for the value of non-detects in calculating means; used when the compound was detected in at least one replicate for the treatment. Values following '<' are equal to the detection limit.

PAHs	Clean Water	Influent	6"	12"	18"	18" CWC
1-Methylnaphthalene	0.002	0.008	0.0015 (0.0005)	0.0018 (0.0007)	0.0012 (0.0005)	0.001 (0)
2-Methylnaphthalene	0.003	0.019	0.0023 (0.0003)	0.0017 (0.0003)	0.0023 (0.0003)	0.002 (0)
Naphthalene	0.006	0.006	0.005 (0.0006)	0.005 (0.001)	0.004 (0.0003)	0.004 (0)
Acenaphthene	<0.003	<0.003	<0.003	<0.003	<0.003	<0.003
Acenaphthylene	<0.002	0.011	<0.002	<0.002	<0.002	<0.002
Anthracene	<0.001	0.011	<0.001	<0.001	<0.001	<0.001
Carbazole	<0.001	0.017	<0.001	<0.001	<0.001	<0.001
Dibenzofuran	<0.002	0.01	<0.002	<0.002	<0.002	<0.002
Fluorene	<0.002	0.015	<0.002	<0.002	<0.002	<0.002
Phenanthrene	<0.001	0.131	0.0027 (0.0003)	0.0023 (0.0003)	0.002 (0)	0.0012 (0.0004)
Benz[a]anthracene	<0.0008	0.043	<0.0008	<0.0008	<0.0008	<0.0008
Chrysene	<0.001	0.127	0.001 (0.0005)	0.0012 (0.0004)	0.0007 (0.0002)	<0.001
Fluoranthene	<0.002	0.22	0.0017 (0.0003)	0.0013 (0.0003)	<0.002	<0.002
Pyrene	<0.001	0.317	0.0018 (0.0007)	0.002 (0.0006)	<0.001	<0.001
Benzo(a)pyrene	<0.003	0.061	<0.003	<0.003	<0.003	<0.003
Benzo(b)fluoranthene	<0.0005	0.076	0.0005 (0.0003)	0.0009 (1E-04)	0.0006 (0.0002)	<0.0005
Benzo(j)fluoranthene	<0.002	0.046	<0.002	<0.002	<0.002	<0.002
Benzo(k)fluoranthene	<0.004	0.046	<0.004	<0.004	<0.004	<0.004
Dibenzo(a,h)anthracene	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001
Perylene	<0.007	0.037	<0.007	<0.007	<0.007	<0.007
Benzo(ghi)perylene	<0.002	0.114	<0.002	<0.002	<0.002	<0.002
Indeno(1,2,3-cd)pyrene	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001
Total PAHs	**0.011**	**1.315**	**0.0165**	**0.0144**	**0.0096**	**0.0072**

Table A1. 37. Average water chemistry values (standard error) for influent waters (clean water and influent stormwater runoff) and triplicate effluent waters from each of the three treatment depths plus the clean water control for Water Year 12. One-half of the value of the detection limit was substituted for the value of non-detects in calculating means; used when the compound was detected in at least one replicate for the treatment. Values following '<' are equal to the detection limit. n.m. = not measured for this event.

Compound	Units	Clean Water	Influent	6"	12"	18"	18" CWC
Conventional							
Dissolved Organic Carbon	mg/L	0.6	7.27	7.44 (0.06)	7.68 (0.2)	8.12 (0.3)	2.03 (0.3) *
Total Suspended Solids	mg/L	<1	61	3 (0)	3 (0.6)	2.33 (0.3)	0.667 (0.3) *
Turbidity	NTU	0.24	21.4	5.87 (1)	6.14 (0.5)	4.53 (0.2)	3.37 (0.07) *
Conductivity	S/cm	831	79.4	81.5 (11)	82.4 (9)	114.8 (9)	855.3 (3.3) *
pH	n.a.	7.734	7.293	7.202 (0.02)	7.219 (0.06)	7.064 (0.02)	7.473 (0.02) *
Alkalinity	mg/L as $CaCO_3$	30.1	20.5	16.67 (0.5) ^{ab}	17.3 (0.7) ^{a}	14.93 (0.5) ^{b}	29 (0.4) *
Temperature	F	n.m.	n.m.	n.m.	n.m.	n.m.	n.m.
Dissolved Calcium	mg/L	1.44	3.5	1.66 (0.7)	1.44 (0.5)	2.43 (0.6)	4.67 (0.6) *
Dissolved Magnesium	mg/L	13.2	0.638	0.412 (0.01) ^{a}	0.195 (0.06) ^{b}	0.258 (0.06) ^{ab}	15.6 (0.05) *
Dissolved Sodium	mg/L	115	5.16	6.16 (0.2) ^{a}	6.39 (0.2) ^{ab}	7.15 (0.3) ^{b}	1.15 (0.3) *
Nutrients							
Nitrate/Nitrite	mg/L	0.112	0.216	1.54 (0.4)	1.1 (0.7)	3.78 (1.1)	2.34 (1.1)
Orthophosphate, as P	mg/L	<0.005	0.058	0.053 (0.003) ^{a}	0.072 (0.002) ^{b}	0.078 (0.005) ^{b}	0.081 (0.005)
Microbiology							
Fecal Coliform by MF	CFU/100 mL	n.m.	n.m.	n.m.	n.m.	n.m.	n.m.

MF = membrane filtration

Table A1. 38. Average concentrations of dissolved and total metals in μg/L (standard error) for influent waters (clean water and influent stormwater runoff) and triplicate effluent waters from each of the three treatment depths plus the clean water control for Water Year 12. One-half of the value of the detection limit was substituted for the value of non-detects in calculating means; used when the compound was detected in at least one replicate for the treatment. Values following '<' are equal to the detection limit.

Compound	Clean Water	Influent	6"	12"	18"	18" CWC
Dissolved As	<0.0373	0.736	0.649 (0.03)	0.656 (0.05)	0.605 (0.03)	0.07 (0.04)*
Dissolved Cd	0.04	<0.03	<0.03	<0.03	<0.03	<0.03
Dissolved Cu	0.185	15.6	12.3 (0.3) ^{a}	11.7 (0.8) ^{ab}	10 (0.2) ^{b}	1.84 (0.1) *
Dissolved Pb	<0.05	0.129	0.119 (0.005)	0.122 (0.006)	0.102 (0.008)	<0.05 *
Dissolved Ni	0.194	1.26	0.976 (0.03)	1.16 (0.2)	1.26 (0.1)	0.544 (0.07) *
Dissolved Zn	<2.92	35.9	4.88 (0.2)	5.41 (0.3)	4.54 (0.2)	<2.92 *
As	<0.0373	1.72	0.778 (0.006)	0.746 (0.006)	0.749 (0.02)	0.046 (0.006) *
Cd	<0.03	0.238	<0.03	<0.03	<0.03	<0.03
Cu	3.71	47.9	15.2 (0.3)	14.5 (1.2)	12.9 (0.3)	2.26 (1) *
Pb	<0.05	5.84	0.208 (0.006)	0.244 (0.04)	0.183 (0.02)	<0.05 *
Ni	0.704	4.68	1.21 (0.06)	1.43 (0.3)	1.48 (0.2)	0.467 (0.1) *
Zn	<2.92	192	6.44 (0.4)	6.37 (0.6)	5.45 (0.2)	3.27 (0.2) *

Table A1. 39. Average polycyclic aromatic hydrocarbon (PAH) concentrations in μg/L (standard error) for influent waters (clean water and influent stormwater runoff) and triplicate effluent waters from each of the three treatment depths plus the clean water control for Water Year 12. One-half of the value of the detection limit was substituted for the value of non-detects in calculating means; used when the compound was detected in at least one replicate for the treatment. Values following '<' are equal to the detection limit.

PAHs	Clean Water	Influent	6"	12"	18"	18" CWC
1-Methylnaphthalene	<0.008	<0.008	0.007 (0.003)	<0.008	<0.008	<0.008
2-Methylnaphthalene	<0.007	0.007	<0.007	<0.007	<0.007	<0.007
Naphthalene	<0.006	0.016	0.006 (0.0003)	0.007 (0.0003)	0.004 (0.001)	0.005 (0.001)
Acenaphthene	<0.004	<0.004	0.007 (0.005)	<0.004	0.003 (0.001)	0.004 (0.002)
Acenaphthylene	<0.005	0.005	<0.005	<0.005	<0.005	<0.005
Anthracene	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005
Carbazole	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005
Dibenzofuran	<0.006	<0.006	<0.006	<0.006	<0.006	<0.006
Fluorene	<0.004	0.004	0.004 (0.002)	<0.004	<0.004	<0.004
Phenanthrene	<0.005	0.024	<0.005	<0.005	<0.005	<0.005
Benz[a]anthracene	<0.005	0.006	<0.005	<0.005	<0.005	<0.005
Chrysene	<0.008	0.03	<0.008	<0.008	<0.008	<0.008
Fluoranthene	<0.006	0.04	<0.006	<0.006	<0.006	<0.006
Pyrene	<0.008	0.058	<0.008	<0.008	<0.008	<0.008
Benzo(a)pyrene	<0.005	0.008	<0.005	<0.005	<0.005	<0.005
Benzo(b)fluoranthene	<0.005	0.019	<0.005	<0.005	<0.005	<0.005
Benzo(j)fluoranthene	<0.005	0.007	<0.005	<0.005	<0.005	<0.005
Benzo(k)fluoranthene	<0.008	<0.008	<0.008	<0.008	<0.008	<0.008
Dibenzo(a,h)anthracene	<0.008	<0.008	<0.008	<0.008	<0.008	<0.008
Perylene	<0.004	<0.004	<0.004	<0.004	<0.004	<0.004
Benzo(ghi)perylene	<0.009	0.036	<0.009	<0.009	<0.009	<0.009
Indeno(1,2,3-cd)pyrene	<0.008	0.01	<0.008	<0.008	<0.008	<0.008
Total PAHs	**0**	**0.303**	**0.024 (0.010)**	**0.007 (0.000)**	**0.007 (0.002)**	**0.009 (0.001)**

Table A1. 40 Average water chemistry values (standard error) for influent waters (clean water and influent stormwater runoff) and triplicate effluent waters from each of the three treatment depths plus the clean water control for Water Year 13. One-half of the value of the detection limit was substituted for the value of non-detects in calculating means; used when the compound was detected in at least one replicate for the treatment. Values following '<' are equal to the detection limit. n.m. = not measured for this event.

Compound	Units	Clean Water	Influent	6"	12"	18"	18" CWC
Conventional							
Dissolved Organic Carbon	mg/L	<0.5	6.62	6.1 (0.08)	6.53 (0.03)	6.81 (0.3)	2.06 (0.2)
Total Suspended Solids	mg/L	53	10	<1	1.33 (0.8)	<1	<1
Turbidity	NTU	0.07	7.96	2.69 (0.4)	2.96 (0.3)	3.05 (0.3)	2.25 (0.1)
Conductivity	S/cm	667	203	130.2 (1)	127.2 (1)	128.2 (0.7)	669.7 (1.9)
pH	n.a.	7.63	7.37	7.27 (0.04)	7.24 (0.06)	7.19 (0.01)	7.6 (0.04)
Alkalinity	mg/L as $CaCO_3$	57.6	38	31.7 (1)	31.4 (0.8)	28.6 (0.4)	53.1 (0.5)
Dissolved Calcium	mg/L	3.15	8.18	6.21 (0.05)	6.00 (0.2)	5.84 (0.2)	3.50 (0.08)
Dissolved Magnesium	mg/L	8.96	0.879	0.907 (0.06)	0.698 (0.02)	0.615 (0.01)	8.94 (0.2)
Dissolved Sodium	mg/L	89.7	10	9.77 (0.4)	9.62 (0.1)	10.09 (0.1)	95.83 (1)
Nutrients							
Nitrate/Nitrite	mg/L	0.204	0.241	0.622 (0.1)	0.42 (0.04)	0.711 (0.2)	0.443 (0.02)
Orthophosphate, as P	mg/L	<0.004	0.004	0.03 (0.0006)	0.03 (0.006)	0.06 (0.006)	0.09 (0.006)
Microbiology							
Fecal Coliform by MF	CFU/100 mL	n.m.	n.m.	n.m.	n.m.	n.m.	n.m.

MF = membrane filtration

Table A1. 41. Average concentrations of dissolved and total metals in μg/L (standard error) for influent waters (clean water and influent stormwater runoff) and triplicate effluent waters from each of the three treatment depths plus the clean water control for Water Year 13. One-half of the value of the detection limit was substituted for the value of non-detects in calculating means; used when the compound was detected in at least one replicate for the treatment. Values following '<' are equal to the detection limit.

Compound	Clean Water	Influent	6"	12"	18"	18" CWC
Dissolved As	<0.0746	0.502	0.467 (0.01)	0.435 (0.01)	0.502 (0.02)	0.257 (0.02)
Dissolved Cd	<0.06	<0.06	<0.06	<0.06	<0.06	<0.06
Dissolved Cu	<0.346	15.2	10.01 (0.3)	8.96 (0.5)	8.12 (0.4)	2.86 (0.5)
Dissolved Pb	<0.103	0.258	0.103 (0.03)	0.145 (0.01)	0.136 (0.003)	0.114 (0.006)
Dissolved Ni	0.18	1.26	0.876 (0.05)	1.00 (0.2)	1.07 (0.07)	0.568 (0.08)
Dissolved Zn	<5.84	29.7	4.05 (1.2)	3.88 (1)	3.93 (1)	<5.84
As	<0.0746	0.64	0.54 (0.005)	0.503 (0.02)	0.551 (0.02)	0.328 (0.01)
Cd	<0.06	<0.06	<0.06	<0.06	<0.06	<0.06
Cu	0.498	19	11.5 (0.6)	10.4 (0.7)	9.66 (0.3)	3.14 (0.3)
Pb	<0.103	7.06	0.189 (0.02)	0.201 (0.01)	0.193 (0.02)	0.143 (0.01)
Ni	0.22	1.6	0.967 (0.04)	1.12 (0.1)	1.25 (0.09)	0.818 (0.06)
Zn	<5.84	49.5	4.23 (1.33)	5.30 (1.2)	6.73 (0.2)	<5.84

Table A1. 42. Average polycyclic aromatic hydrocarbon (PAH) concentrations in µg/L (standard error) for influent waters (clean water and influent stormwater runoff) and triplicate effluent waters from each of the three treatment depths plus the clean water control for Water Year 13. One-half of the value of the detection limit was substituted for the value of non-detects in calculating means; used when the compound was detected in at least one replicate for the treatment. Values following '<' are equal to the detection limit.

PAHs	Clean Water	Influent	6"	12"	18"	18" CWC
1-Methylnaphthalene	<0.008	<0.008	<0.008	<0.008	<0.008	<0.008
2-Methylnaphthalene	<0.007	<0.007	<0.007	<0.007	<0.007	<0.007
Naphthalene	<0.006	0.008	0.004 (0.001)	<0.006	<0.006	<0.006
Acenaphthene	<0.004	<0.004	<0.004	<0.004	<0.004	<0.004
Acenaphthylene	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005
Anthracene	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005
Carbazole	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005
Dibenzofuran	<0.006	<0.006	<0.006	<0.006	<0.006	<0.006
Fluorene	<0.004	<0.004	<0.004	<0.004	<0.004	<0.004
Phenanthrene	<0.005	0.011	<0.005	0.004 (0.001)	<0.005	<0.005
Benz[a]anthracene	<0.006	<0.006	<0.006	<0.006	<0.006	<0.006
Chrysene	<0.008	<0.008	<0.008	<0.008	<0.008	<0.008
Fluoranthene	<0.006	0.012	<0.006	<0.006	<0.006	<0.006
Pyrene	<0.008	0.019	<0.008	<0.008	<0.008	<0.008
Benzo(a)pyrene	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005
Benzo(b)fluoranthene	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005
Benzo(j)fluoranthene	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005
Benzo(k)fluoranthene	<0.008	<0.008	<0.008	<0.008	<0.008	<0.008
Dibenzo(a,h)anthracene	<0.008	<0.008	<0.008	<0.008	<0.008	<0.008
Perylene	<0.004	<0.004	<0.004	<0.004	<0.004	<0.004
Benzo(ghi)perylene	<0.009	<0.009	<0.009	<0.009	<0.009	<0.009
Indeno(1,2,3-cd)pyrene	<0.008	<0.008	<0.008	<0.008	<0.008	<0.008
Total PAHs	**0**	**0.05**	**0.004 (0.001)**	**0.004 (0.001)**	**0**	**0**