

# Implications of Using Different Water Sources When Hydrologically Compacting Bioretention Columns

Clayton A. Stahnke<sup>1</sup>, Cara J. Poor<sup>1\*</sup>

**ABSTRACT:** Many investigators have conducted research on bioretention systems both in the laboratory and field. There is little consensus on which sources of water are best suited to hydrologically compact bioretention columns. Water with low ionic strength can leach ions from soil media, resulting in a different soil chemistry environment than would be found in typical bioretention applications. Soil columns were hydrologically compacted with three different water sources often used in column studies: deionized water, tap water, and rain water. Influent and effluent samples for each water source were measured for pH, conductivity, copper, zinc, and phosphate. On average, deionized water yielded larger percentage increases between influent and effluent for pH, conductivity, copper, and zinc, indicating that deionized water leaches out more ions from bioretention media than tap water or rain water. To maintain soil chemistry similar to the field, rain water or tap water should be used in column studies. *Water Environ. Res.*, 89, 451 (2017).

**KEYWORDS:** bioretention, column studies, soil chemistry, metals, green stormwater infrastructure, stormwater treatment.  
doi:10.2175/106143017X14839994523505

## Introduction

Research on the utilization and optimization of bioretention systems has increased significantly in the past decade (Herrera, 2015; Lynn et al., 2015; Palmer et al., 2013). Bioretention systems typically consist of an aggregate drainage layer, a 30- to 60-cm thick layer of soil media, and a top layer of mulch and vegetation, with a shallow depression or ponding zone to collect stormwater runoff (City of Portland, 2014). Bioretention systems are often used in urban areas with higher intensity stormwater runoff that can carry higher concentrations of contaminants, including bacteria, hydrocarbons, heavy metals, and sediments

from roadways (Herrera, 2015). As stormwater infiltrates into the soil structure, bioretention systems use passive mechanisms to improve water quality, including filtration, plant uptake, and microbial reduction.

However, as a relatively new, developing technology, there has been a lack of uniformity in design parameters (e.g., media composition, basin geometry, vegetation selection, drainage configuration, etc.) as well as methods used in research applications (Davis et al., 2009; Lim et al., 2015; Rycewicz-Borecki et al., 2015). While lack of uniformity in applied design parameters can be readily addressed through standardized designs based on the climatic tendencies and infrastructural necessities between cities and states, the differences in bioretention research study methods can make interpretation and comparison of results challenging, particularly when making design decisions and establishing standards.

One area of discrepancy is the type of water used to hydrologically compact bioretention columns prior to testing. Compacting soils by running water through columns is a common method used to decrease infiltration rates to those typically observed in field applications. Applying water to the columns prior to testing also allows for proper plant establishment. Although it is clear that hydrologic compaction is integral to accurately representing bioretention systems, it is not clear what type of water to use. Water sources have ranged from tap water to deionized (DI) water in column studies (Lim et al., 2015; Palmer et al., 2013; Rycewicz-Borecki et al., 2015). Rycewicz-Borecki et al. (2015) hydrologically compacted bioretention columns with tap water before conducting tests with synthetic stormwater, accounting for effluent metals concentrations from both events for their final calculations. Palmer et al. (2013) and Lim et al. (2015) hydrologically compacted columns with DI water before conducting tests with stormwater. Using different water sources may impact interpretation of results from various studies.

Water sources that have low ionic strength may have the capacity to alter the chemical composition of bioretention media (Mullane et al., 2015), raising the question of whether or not

<sup>1</sup> School of Engineering, University of Portland.

\* School of Engineering, University of Portland, 5000 N. Willamette Blvd., Shiley Hall, Portland, Oregon 97203-5798; telephone: 503-943-8743; e-mail: poor@up.edu.

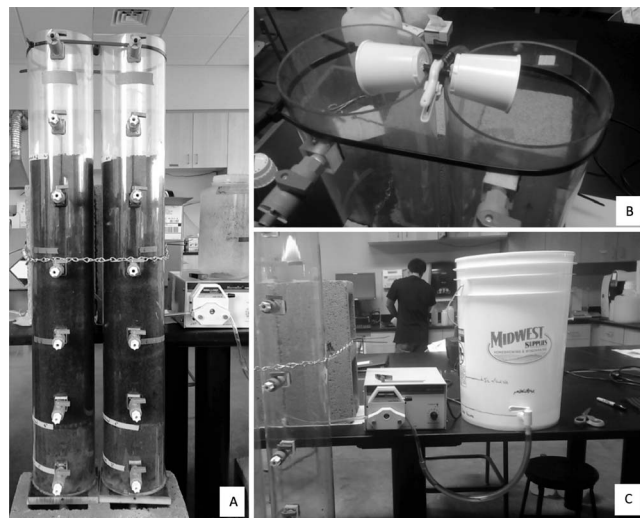
results from column studies using DI water are representative of conditions in the field. Deionized water, with a negligible concentration of dissolved ions, may exhibit a greater tendency to leach ions from bioretention soil media than stormwater or tap water used for irrigation. Deionized water can change the ionic balance of the soil; ions tend to leach from solutes with high concentrations to solutes with low concentrations due to the concentration gradient.

The intent of this study is to determine how different water sources can affect the results of bioretention column studies. The authors hypothesize that DI water, when used to hydrologically compact bioretention columns, can leach out larger quantities of dissolved nutrients and metals or alter the chemical composition of soils in comparison to typical water sources used in field applications, such as tap water or rain water/stormwater. To test this hypothesis, the authors applied DI water, tap water, and rain water to columns and analyzed the influent and effluent for copper, zinc, phosphate, pH, and conductivity to evaluate the impacts of each water source.

## Methods

**Experimental Design.** This experiment was designed to replicate hydrologic compaction procedures used in typical bioretention column studies. Two Plexiglas cylindrical columns were constructed with dimensions similar to columns used in other studies (Barrett et al., 2013; Hsieh and Davis, 2005a; Lynn et al., 2015). Both columns had a diameter of 14.3 cm and a height of 102.9 cm (Figure 1). The columns had a drainage layer consisting of 5 cm of medium river rock and 10 cm of 0.75" minus crushed gravel. After installation of the drainage layer, the bioretention soil media was physically compacted in three 20-cm layer increments for a total of 60 cm of bioretention soil media. A plastic 1-mm mesh filter was placed at the bottom of both columns to prevent clogging. Effluent was collected directly from the bottom of the column. The bioretention soil media consisted of 30 to 40% compost and 60 to 70% loamy sand, as specified in the City of Portland Stormwater Management Manual (City of Portland, 2014).

The three water sources that were collected and used for this study were DI water, tap water, and rain water. A Milli-Q<sup>®</sup> laboratory water purification system was used to produce DI water, and rain water was collected at a downspout from a built-up roof covered with gravel. Thirty liters of each type of water was collected in high-density polyethylene (HDPE) containers for testing. Water was applied to both columns using a peristaltic pump at a flow rate ranging between 75 to 85 mL/min, which equates to a runoff rate of 28.0 to 31.8 cm/hr (11.0–12.5 in./hr). This rate was calculated using dosage methods utilized by Palmer et al. (2013), where the amount of water applied to each column was based on the column's cross-sectional area, a runoff ratio of 10:1, and the 6-month, 24-hour storm in Portland, Oregon (0.83 in. or 2.11 cm). These values result in a total influent volume of approximately 5 L per column. Ten liters of source water was used for each test, which was then split between both columns to administer approximately 5 L to each column (Figure 1).



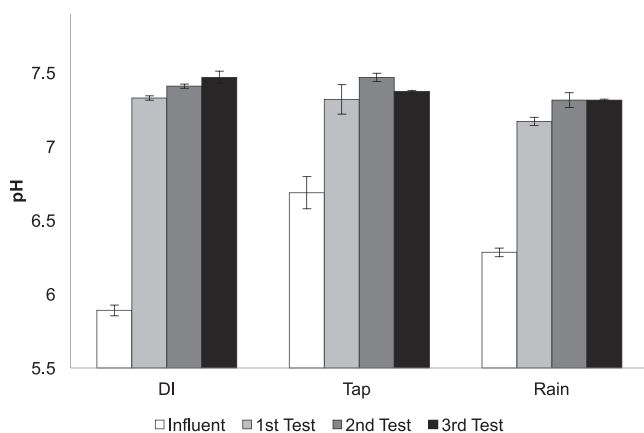
**Figure 1**—Experimental setup, including soil columns (A), the flow splitter use to distribute water to the columns (B), and the bucket and peristaltic pump used to deliver water to columns (C).

For each water source analyzed, three tests were performed. Application of water during each test took approximately 65 minutes. Both columns were then allowed to drain, and effluent was collected in HDPE carboys. Time between tests was 24 to 48 hours. Tests were conducted in a temperature-controlled laboratory (constant temperature of 20 °C). All sample bottles were acid-washed and samples were stored according to standard methods (Rice et al., 2012). In between testing of each water source, the soil and gravel was discarded and the columns were cleaned and repacked with new, clean soil.

Conductivity was measured using a Vernier probe, and pH was measured using a HACH probe. A composite sample of the influent and effluent from each test was collected for analysis. Phosphate was analyzed using U.S. Environmental Protection Agency Method 365.1. Because sample turbidity was high, samples had to be diluted 3:1 for phosphate analysis. Copper and zinc were analyzed using a Shimadzu AAS (Method 3111; Rice et al., 2012). The Wilcoxon signed rank test (Helsel and Hirsch, 2002) was used to determine whether there was a significant difference between influent and effluent, and whether there was a significant difference among the three water sources.

## Results and Discussion

Results for all constituents measured are shown in Figures 2 through 6. The influent and effluent concentrations were significantly different ( $p < 0.05$ ) for all constituents except zinc. Influent and effluent zinc concentrations were significantly different for tests with DI water, but no statistical difference was found for tests with tap and rain water. The percent difference between influent and effluent was notably different between water sources (Table 1). Because the influent concentration of copper and zinc in the DI water were below detection limits, percent difference for these constituents were calculated assuming an influent copper and zinc concentration of 1 ppb.

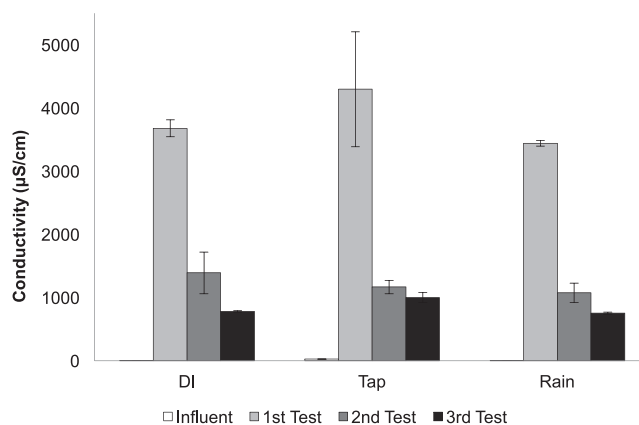


**Figure 2—Influent and effluent pH for the three different water sources. Uncertainty bars = 1 standard deviation.**

The concentrations for DI, tap, and rain water were significantly different ( $p < 0.05$ ) for all constituents measured except phosphate, in which no statistical difference was found. Trends varied for each water quality constituent, and are discussed in the following sections.

Exfiltration rates, or the flow rate per cross-sectional area exiting the column, and volume of water recovered increased with testing. Average exfiltration rates during the initial tests was 25.4 cm/hr (10.0 in./hr) and 37.4 cm/hr (14.7 in./hr) during subsequent tests. Exfiltration rates were generally low during each initial test, which was likely due to the bioretention media being initially dry. The average percentage of water recovered by volume was 81% for the initial tests, and 93% during the final tests. During testing, the authors observed the pore spaces in the bioretention media initially filling with water, which likely resulted in a lower exfiltration rate and a higher volume of water stored in the columns.

**pH and Conductivity.** The influent pH of the three water sources was expectedly varied, with DI water at a lower pH (5.9) than tap and rain water (6.7 and 6.3, respectively). The pH of effluent samples for all tests were similar, with an average of 7.4 (Figure 2). This relatively consistent effluent pH is likely due to the high buffer capacity of natural soils and compost (Mullane et al., 2015). The highest change in pH was observed during the DI water tests, with a pH increase ranging between 24.7 and 27.5%



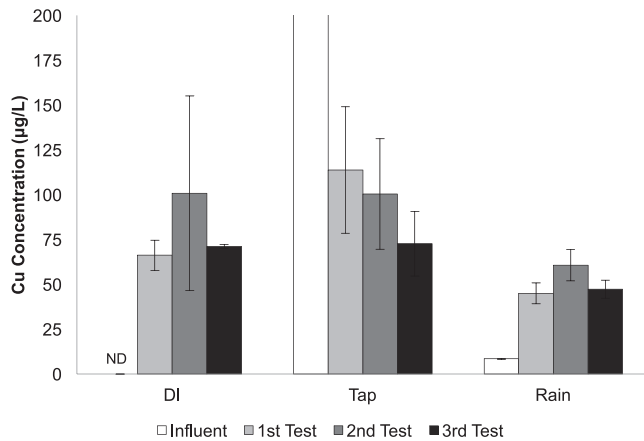
**Figure 3—Influent and effluent conductivity for the three different water sources. Uncertainty bars = 1 standard deviation.**

(Table 1). Natural waters are largely controlled by the carbonate system. Sources of carbonate include atmospheric carbon dioxide, respiration, and carbonate solids (Morel and Hering, 1993). Carbon dioxide is a weak acid, which lowers the pH of natural waters. When no other ions are present, the theoretical pH of water is about 5.6 when open to the atmosphere. As more ions are dissolved in water, the pH increases. This indicates that the amount of dissolved ions in the DI water increased as it traveled through the soil column.

Influent conductivity for tap, DI, and rain water was 27.7, 1.97, and 6.8  $\mu\text{S/cm}$ , respectively (Figure 3). During the initial tests, conductivity increased substantially and effluent conductivity was comparable for all water sources (4298, 3680, and 3442  $\mu\text{S/cm}$  for tap, DI, and rain water, respectively). Conductivity is directly proportional to the total dissolved solids, or ions, in a solution (Morel and Hering, 1993). Therefore, water with a higher conductivity generally contains a higher concentration of ions. Water that has a low ionic strength will have a higher tendency to leach ions from surrounding media. When compared to rain water, DI water yields a slightly higher effluent conductivity per test, suggesting that more ions may leach out with DI water than rain water (Figure 3). Additionally, the percent increase in conductivity between influent and effluent was considerably higher with DI water tests (Table 1). These results correspond to the higher changes in pH

**Table 1—Percentage differences between influent and effluent concentrations for each test. The difference between influent and effluent was statistically significant ( $p < 0.05$ ) for all water sources and constituents except for tap and rain water zinc levels. The difference between water sources was statistically significant ( $p < 0.05$ ) for all constituents except phosphorus.**

	DI Water			Tap Water			Rain Water		
	1st Test	2nd Test	3rd Test	1st Test	2nd Test	3rd Test	1st Test	2nd Test	3rd Test
<b>pH</b>	24.7%	25.0%	27.5%	8.4%	10.7%	12.4%	13.8%	16.1%	17.0%
<b>Conductivity</b>	96703%	173775%	60177%	14979%	4224%	3516%	52059%	15990%	10513%
<b>Zinc</b>	3602%	6309%	7664%	-22.6%	31.6%	-16.9%	-36.7%	2.4%	7.0%
<b>Copper</b>	6527%	9980%	7021%	-78.9%	-72.6%	-87.0%	411.2%	634.9%	456.3%
<b>Phosphate</b>	1533%	3958%	3494%	1221%	3608%	3400%	1262%	3925%	5115%

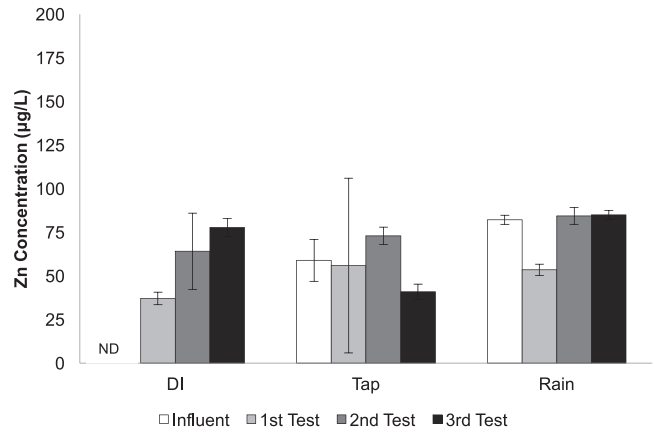


**Figure 4—Influent and effluent copper concentrations for the three different water sources. Copper concentration for influent tap water (489.3 ± 106.6 µg/L) was cut off for graphical purposes. Uncertainty bars = 1 standard deviation, ND = non-detect.**

observed with DI water. The effluent conductivity with tap water is slightly higher than the conductivity with DI water during the first and third tests, and lower during the second test. Although the conductivity was higher, the change in conductivity was much higher for DI water than tap water (Table 1). High conductivity observed in all effluent may be due to the initial washout of solids observed in newer bioretention facilities (Hsieh and Davis, 2005b), and not related to the leaching capability of water sources.

**Copper and Zinc.** Effluent copper concentrations were higher than influent for the tests with DI and rain water (Figure 4), whereas a decrease in copper was observed with tap water (average effluent concentrations of 96 µg/L and a 73 to 87% decrease for all tests). Copper concentrations in the tap water were very high (influent concentration of 489 µg/L). This is likely due to the older copper piping in the building. Although copper levels in the tap water influent were high, effluent copper concentrations were similar to concentrations observed during the DI water tests and the increase in copper concentrations was much larger with DI water than the other water sources (Table 1). Similar to conductivity results, copper concentrations were higher in the DI water effluent than rain water effluent, indicating that DI water can leach out more copper than other natural water sources.

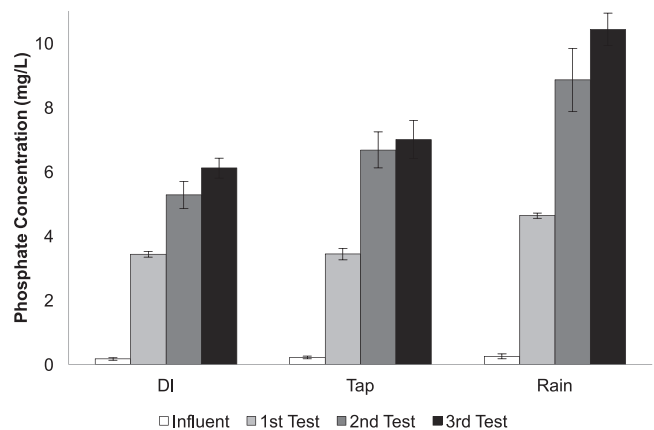
Zinc results illustrate different trends than copper measurements, though similar conclusions related to leaching can be made (Figure 5). No statistical difference was found between the influent and effluent zinc concentrations for tap water and rain water, indicating that little to no zinc is leaching from the soil media. A large increase of zinc in the effluent was observed during DI water tests. The influent zinc concentration in the DI water was below detection limits, thus the zinc observed in the effluent must be from the soil media. Zinc is leaching from the columns flushed with DI water, but not from the columns flushed with tap or rain water.



**Figure 5—Influent and effluent zinc concentrations for the three different water sources. Uncertainty bars = 1 standard deviation, ND = non-detect.**

**Phosphate.** Phosphate leached from the soil media with all three water sources (Figure 6). Similar trends were observed for all three water sources, with no statistical difference found between water sources (Table 1). The columns flushed with rain water leached the most phosphate. Samples were visually very turbid, indicating a high suspended solids concentration. Phosphate is known to be associated with suspended solids (Petry et al., 2002). The observed leaching of phosphate could be a result of initial movement of suspended solids during hydrologic compaction, or from leaching due to the water source.

**Solids Washout.** A large initial washout of solids was observed, which has also been observed in bioretention field applications with underdrains (Hsieh and Davis, 2005b). As water runs through bioretention soil that is not compacted, some soil media will wash out of the system. This washout can affect the export of phosphate and other ions from the columns, as observed with the high conductivity and phosphate concentrations in the effluent. These tests also indicate there were large quantities of phosphate, copper, and zinc in the soil media.



**Figure 6—Influent and effluent phosphate concentrations for the three different water sources. Uncertainty bars = 1 standard deviation.**



Phosphate is likely from the compost (Herrera, 2015). If the goal is to reduce copper, zinc, and phosphate to low levels for protection of sensitive waters, the soil media must be free of these pollutants.

**Water Source Implications.** Many investigators use DI water because it is free of ions and potential contaminants. If DI water is used, contaminants will not be added in a column study. However, DI water can also leach ions from the soil media. Bioretention systems in the field are often irrigated with tap water the first two years to maintain vegetation, and/or exposed to rain water and surface water. Tap or rain water more accurately mimic field conditions, although there are drawbacks to these two water sources as well. Tap water and rain water from a roof may have high levels of metals and/or other ions, as was observed with copper and zinc (Figures 4 and 5). Tap water could contain high levels of ions due to the source of water or the piping material. The roof material can affect roof runoff quality (Kingett Mitchell and Diffuse Sources, 2004; Mendez et al., 2010); metals have been detected in rain water collected from roofs in other studies (Lye, 1992; Quek and Förster, 1993). Potential contamination or leaching could be avoided by using a synthetic rain water made with DI water and salts (i.e., Mullane et al., 2015). Ionic strength can be controlled with synthetic rain water, although the large volumes required for large studies is not always feasible.

## Conclusions

Columns are often compacted with water before testing, but there is a discrepancy, or nonuniformity, on the type of water used. The authors flushed columns with three types of water: DI, tap, and rain water. Results indicate that DI water has a higher tendency to leach ions from the soil media due to the low ionic strength of DI water, which may change the soil chemistry. To more closely mimic field conditions, column studies should use tap or rain water to hydrologically compact columns. If tap or rain water from a roof is used, water quality tests should be conducted to ensure it does not have high levels of metals or other ions before applying to columns. High levels of metals and other ions in water used to hydrologically compact a column may impact the actual stormwater testing results. A synthetic rain water can be used if there is concern that ion levels are too high in tap or collected rain water.

## Acknowledgments

The University of Portland provided funding to complete this study. The authors thank Jacob Amos and Allen Hansen for assistance with column construction.

*Submitted for publication June 13, 2016; revised manuscript submitted October 26, 2016; accepted for publication November 25, 2016.*

## References

- Barrett, M. E., Limouzin, M., Lawler, D. F. (2013) Effects of Media and Plant Selection on Biofiltration Performance. *J. Environ. Eng.*, **139**, 462–470.
- City of Portland (2014) *Stormwater Management Manual (SWMM)*; Portland, Oregon.

- Davis, A. P.; Hunt, W. F.; Traver, R. G.; Clar, M. (2009) Bioretention Technology: Overview of Current Practice and Future Needs. *J. Environ. Eng.*, **135**, 109–117.
- Helsel, D. R.; Hirsch, R.M. (2002) *Techniques of Water-Resources Investigations of the U. S. Geological Survey: Statistical Methods in Water Resources*; USGS: Reston, Virginia.
- Herrera Environmental Consultants, Inc. (2015) *Analysis of Bioretention Soil Media for Improved Nitrogen, Phosphorus, and Copper Retention—Final Report*; Herrera Environmental Consultants, Inc.: Seattle, Washington.
- Hsieh, C. H.; Davis, A. P. (2005a) Evaluation and Optimization of Bioretention Media for Treatment of Urban Storm Water Runoff. *J. Environ. Eng.*, **131**, 1521–1531.
- Hsieh, C. H.; Davis, A. P. (2005b) Multiple-Event Study of Bioretention for Treatment of Urban Storm Water Runoff. *Water Sci. Technol.*, **51**, 177–181.
- Kingett Mitchell and Diffuse Sources (2004). *A Study of Runoff Quality in Auckland New Zealand: Implications for Stormwater Management*; Technical Publication 213; Auckland Regional Council: Auckland, New Zealand; January 2004.
- Lim, H. S., Lim, W.; Hu, J. Y.; Ziegler, A.; Ong, S. L. (2015) Comparison of Filter Media Materials for Heavy Metal Removal from Urban Stormwater Runoff Using Biofiltration Systems. *J. Environ. Manage.*, **147**, 24–33.
- Lynn, T. J., Yeh, D. H.; Ergas, S. J. (2015) Performance of Denitrifying Stormwater Biofilters Under Intermittent Conditions. *Environ. Eng. Sci.*, **32**, 796–805.
- Lye, D. J. (1992) Microbiology of Rain Water Cistern Systems: A Review. *J. Environ. Sci. Health, Part A: Environ. Sci. Eng.*, **27**, 2123–2166.
- Mendez, C. B., Afshar, B. R.; Kinney, K.; Barrett, M. E.; Kirisits, M. J. (2010) *Effect of Roof Material on Water Quality for Rain Water Harvesting Systems*; Texas Water Development Board: Austin, Texas; January 2010.
- Morel, F. M. M.; Hering, J. G. (1993) *Principles and Applications of Aquatic Chemistry*; John Wiley & Sons, Inc.: Toronto, Canada.
- Mullane, J. M.; Flury, M.; Iqbal, H.; Freeze, P. M.; Hinman, C.; Cogger, C. G.; Shi, Z. (2015) Intermittent Rainstorms Cause Pulses of Nitrogen, Phosphorus, and Copper in Leachate from Compost in Bioretention Systems. *Sci. Total Environ.*, **537**, 294–303.
- Palmer, E. T.; Poor, C. J.; Hinman, C.; Stark, J. D. (2013) Nitrate and Phosphate Removal through Enhanced Bioretention Media, Mesocosm Study. *Water Environ. Res.*, **85**, 823–832.
- Quek, U.; Förster, J. (1993) Trace Metals in Roof Runoff. *Water, Air, Soil Pollut.*, **68**, 373–389.
- Rice, E. W.; Baird, R. B.; Eaton, A. D.; Clesceri, L. S. (2012) *Standard Methods for the Examination of Water and Wastewater*, 22nd ed.; American Water Works Association, American Public Works Association, Water Environment Federation: Washington, D.C.
- Rycciewicz-Borecki, M.; McLean, J. E.; Dupont, R. R. (2015) Bioaccumulation of Copper, Lead, and Zinc in Six Macrophyte Species Grown in Simulated Stormwater Bioretention Systems Grown in Simulated Stormwater Bioretention Systems. *J. Environ. Manage.*, **166**, 267–275.