UNIVERSITY OF MINNESOTA

ST. ANTHONY FALLS LABORATORY
Engineering, Environmental and Geophysical Fluid Dynamics

Project Report No. 603

Biofiltration Media Optimization
FINAL REPORT

by

Andrew J. Erickson, Jessica L. Kozarek, Kathryn A. Kramarczuk, and Laura Lewis

St. Anthony Falls Laboratory, University of Minnesota,
2 Third Avenue SE Minneapolis, MN 55455

Prepared for
University of Minnesota Water Resources Center,
Minnesota Stormwater Research Council

December 2022
Minneapolis, Minnesota

Cite as: Erickson, AJ, Kozarek, JL, Kramarczuk, KA, and Lewis, L. (2022). "Biofiltration Media Optimization: Final Report." Project Report No. 603, St. Anthony Falls Laboratory, University of Minnesota, Minneapolis, MN. December 2022.

This project was supported by the Minnesota Stormwater Research and Technology Transfer Program administered by the University of Minnesota Water Resources Center. Financial support was provided through an appropriation from the Clean Water Fund established by Minnesota Clean Water Land and Legacy Amendment and from the Minnesota Stormwater Research Council with financial appropriations from:

- Barr Engineering Company
- Capitol Region Watershed District
- City of Bloomington
- City of Edina
- City of Minnetonka
- City of Woodbury
- Comfort Lake-Forest Lake Watershed District
- Emmons & Olivier Resources, Inc.
- Mississippi Watershed Management Organization
- Nine Mile Creek Watershed District
- Ramsey-Washington Metro Watershed District
- South Washington Watershed District
- Stantec
- Upper Mississippi River Source Water Protection Project
- Valley Branch Watershed District

For more information about the Center and the Council, visit https://wrc.umn.edu/msrc

For more information about the Minnesota Clean Water, Land and Legacy Amendment, visit https://www.legacy.mn.gov/about-funds

Water Resources Center
University of Minnesota
Driven to Discover℠

MINNESOTA STORMWATER RESEARCH COUNCIL

CLEAN WATER LAND & LEGACY AMENDMENT

ACKNOWLEDGEMENTS

The authors thank the Minnesota Stormwater Research and Technology Transfer Program, the University of Minnesota Water Resources Center, the Minnesota Stormwater Research Council, the Clean Water Fund, the Minnesota Clean Water Land and Legacy Amendment and affiliated entities for provided funding to support this project. Support and assistance from several organizations and individuals are listed below and is greatly appreciated. Support and assistance for the contracting process was provided by John Bilotta, Jodi Rahn (Fritsch), Kari Lamp, and Jenni Larson.

Support and assistance from several organizations and individuals are listed below and is greatly appreciated. Input and feedback were provided by the project team members:

- Barbara Heitkamp, University of Minnesota (UMN), St. Anthony Falls Lab (SAFL)
- John Chapman, UMN, Biosystems and Bioproducts Engineering
- Mike Isensee, Carnelian-Marine-St. Croix Watershed District
- Mike Trojan, Minnesota Pollution Control Agency (MPCA)
- David Fairbairn, MPCA
- Erin Anderson Wenz, Barr Engineering

The authors also thank several individuals for providing input during listening sessions on February 1, 2019 (Curt Coudron, Dakota County Soil and Water Conservation District; Dwayne Stenlund, Minnesota Department of Transportation; John Gulliver, UMN SAFL; Kayla Walsh, MPCA; Kerry Glader, Plaisted Companies; Tom Halbach, UMN Department of Soil, Water, and Climate), February 8, 2021 (Dave Saftner, UMD Civil Engineering; Mike Trojan, MPCA Stormwater Program; Erin Anderson Wenz, Barr Engineering Co; Dwayne Stenlund, MnDOT stormwater and erosion prevention unit; Bryan Pynn, Washington Conservation District; Bridget Ulrich, UMD NRRI; Erik Larson, UMD Facilities Management Civil Engineer; Tom Johnson, City of Duluth Stormwater Engineer; Curt Coudron, Dakota County SWCD; Tiffany A Sprague, Stormwater Research Coordinator with NRRI (UMD) and Resilience Extension Educator with MN Sea Grant; Ed Matthiesen, Water resource engineer at Stantec; Eric Watkins, UMN Horticultural Science (turfgrass science); Keith Pilgrim, Barr Engineering; Chan Lan Chun, UMD Civil Engineering and Natural Resources Research Institute; John Gulliver, St. Anthony Falls Laboratory, Univ. of Minnesota; Andy Novak, Landscape Restoration Specialist at Washington Conservation District; Greg Wilson, Barr Engineering Company; Peggy Jones, American Peat Technology, Aitkin, MN; Mike Isensee, Carnelian Marine St. Croix Watershed District; John Chapman, UMN BBE department and Erosion and Stormwater Management Certification Program), and/or throughout the project from 2019 through 2022.

The authors thank Plaisted Companies (Elk River, MN) for donating materials for the experiments. The authors also which to thank Creekside Soils (Hutchinson MN), Empire Mulch (Rosemount MN), Cologne Compost (Cologne MN), Cottage Grove Compost (Cottage Grove MN) and the Shakopee Mdewakanton Sioux Community (SMSC) Organics (Shakopee MN) for donating materials or allowing us to purchase materials for this project.

Support provided by St. Anthony Falls Laboratory (SAFL) staff and students include Aaron Ketchmark, Alyson Skoglund, Anna Healy, Ben Erickson, Eleanor Arpin, Haleigh Ziebol, Jack Lorentzen, John Carlson, Katherine Cloherty, Katie Kemmitt, Melissa Friese, Nam Nguyen, Nick Bentelspacher, Nick Findley, Noah Gallagher, Sam Hirsch, Sam Wang, Sandra Larson, Tanner Schubert, Tasha Spencer, Tiara Burton, Yiling Chen, among others.

TABLE OF CONTENTS

FIGURES

TABLES

CHAPTER 1: INDOOR TESTS AND METRICS EVALUATION EXPERIMENTS

The indoor tests and metrics evaluation experiments described in this chapter was originally published in Erickson, AJ, Kozarek, JL, Kramarczuk, KA, and Lewis, L. (2021). "Biofiltration Media Optimization – Phase 1 Final Report." Project Report No. 593, St. Anthony Falls Laboratory, University of Minnesota, Minneapolis, MN. January 2021.

The purpose of these tests is to identify characteristics, tests, or metrics that can be used by designers in specifications to ensure biofiltration media components provide adequate filtration rate, support plant growth and microbial function, and do not release phosphate from the media. Tests or metrics are considered "simple" if they can be performed or measured in the field (preferable) or quickly in a lab, such that material characteristics do not change between the time of measurement and the time of installation. For these experiments, several compost materials were obtained, and the phosphate release was measured with simple batch experiments. The phosphate release was then compared to several laboratory techniques and 'at-home' or 'in-field' test kits to determine correlations.

1.1 METHODS AND MATERIALS

1.1.1 Compost Collection

Seven compost samples were collected on August 19th, 2020, from five different sites; Creekside Soils in Hutchinson MN, Empire Mulch in Rosemount MN, Cologne compost site in Cologne MN, Cottage Grove Compost Site in Cottage Grove, MN, and the Shakopee Mdewakanton Sioux Community (SMSC) Organics in Shakopee MN. Two sites (named Site 1 and Site 2 hereafter) provided a yard waste sample and an organic (food residue) sample. Two other sites (Sites 3 & 4) provided only yard waste samples. One site (Site 5) provided a blended sample, which is a mix of yard waste and food residue.

1.1.2 Compost Sample Preparation

A sub-sample of each compost sample was added to 273mL of MilliQ water (18.2MΩ-cm) in a 500mL acid-washed glass bottle in three different amounts: 1 gram, 2 grams, and 4 grams of compost for three triplicates of each mixture. Each bottle was placed on an orbital shaker table for 15 minutes at 1500rpm and then a 50mL subsample of the supernatant was poured into a 50mL conical tube and centrifuged at 2800rpm for 15 minutes. Using a syringe and a 0.45µm filter, the 50-mL samples were filtered into three 15mL subsamples and stored in conical tubes.

1.1.3 Solvita Compost Maturity Index

Two sub-samples from each compost (14 total) were collected and processed according to the Solvita manufacturer instructions (Brinton, 2019). The methodology produces ordinal number values for CO_2 respiration, NH_3 respiration, and Solvita Maturity Index. The ordinal values for CO_2 respiration and NH_3 respiration correspond to concentrations of their respective gasses as described in Table 1 (Brinton and Evans, 2006). The ordinal values for CO_2 respiration (varies from 1 to 8) and NH_3 respiration (varies from 1 to 5) are then used to determine the Solvita Maturity Index according to Figure 1.

Table 1: Solvita Ordinal Numbering of Visual Optical Scale in Relation to Concentration of Gases (Brinton and Evans 2006).

Ordinal Number for CO_2 Test Result	Approximate CO_2 Concentration (mg/L)	Ordinal Number for NH_3 Test Result	Approximate NH_3 Concentration (mg/L)
8	2,000	N/A	N/A
7	5,000	N/A	N/A
6	10,000	N/A	N/A
5	20,000	5	<100
4	40,000	4	800
3	75,000	3	2,000
2	140,000	2	8,000
1	200,000	1	25,000

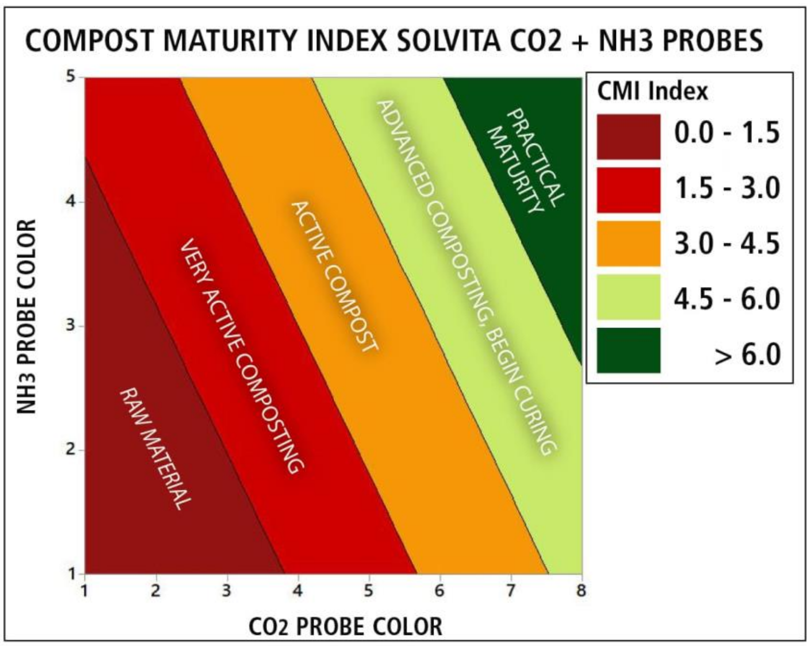

Figure 1: Compost Maturity Index as a function of CO2 and NH3 Ordinal Values (https://solvita.com/cmi-calculator/).

1.1.4 Wet Chemistry Analytical Techniques

Phosphate concentration was measured using flow injection analysis by the Lachat Instruments (Milwaukee, Wisconsin) Quick-Chem model 8000, Method 10-115-01-1-M with a statistically determined detection limit (as determined in water) of 5 µg P/L. The Lachat was calibrated using prepared phosphorus standards ranging from 0 - 2000 µg P/L. These standards were also used during the sample run as check standards; one for every 10 samples. Analysis was repeated (duplicated) on one sample after every 10 samples for quality assurance and quality control (QA/QC). The system was rinsed with

Disodium ethylenediaminetetraacetic acid (EDTA) and MilliQ (high degree deionized water) immediately after calibration standards and after each check standard or sample duplicate. Calibration curves and measured concentrations were recorded digitally by the Lachat software and exported into Excel for additional data and QA/QC analysis.

1.1.5 Compost Soil Analytical Techniques

A single sample of each compost (seven total) was submitted to Research Analytical Laboratories (http://ral.cfans.umn.edu/) for analysis of Olsen Phosphorus (mg/kg media), Bray Phosphorus (mg/kg media), 27-Element analysis by ICP-OES including phosphorus (mg/kg), and Mehlich III Phosphorus (mg/kg media). The methods for these analyses are provided below, quoted from the RAL website at http://ral.cfans.umn.edu/tests-analysis/soil-analysis (Accessed 11/12/2020):

- *"Phosphorus, Bray-1 Extractable, for non-calcareous soils: Phosphorus is extracted by shaking 1 g of air dried soil in 10 mL of 0.025 M HCl and 0.03 M NH_4F for 5 minutes. Phosphorus is determined on the filtrate by the molybdate-blue method using ascorbic acid as a reductant. Color development is measured at 880 nm on a Brinkmann PC 900 probe colorimeter.* [Source] *Frank K., D. Beagle and J. Denning. Phosphorus. p.21-29. in Recommended Chemical Soil Test Procedures for the North Central Region. North Central Regional Research Publication No. 221 (Revised). Jan. 1998. Missouri Agricultural Experiment Station SB 1001.*
- *"Phosphorus, Olsen Bicarbonate Extractable, for calcareous soils: Phosphorus is extracted by shaking 1 g of air dried soil in 20 ml of 0.5 M $NaHCO_3$, pH 8.5, for 30 minutes. Phosphorus is determined on the filtrate by the molybdate-blue method using ascorbic acid as a reductant. Color development is measured at 900 nm on a Brinkmann PC 900D probe colorimeter.* [Source] *Frank K., D. Beagle and J. Denning. Phosphorus. p.21-29. in Recommended Chemical Soil Test Procedures for the North Central Region. North Central Regional Research Publication No. 221 (Revised). Jan. 1998. Missouri Agricultural Experiment Station SB 1001.*
- *"Total phosphorus (S31): Total P Microwave Procedure: digest 0.5 g of air dried soil with 10 mL of HNO_3 in a 50 mL quartz vessel using microwave digestion for 6.5 minutes at 175°C. Determination of P, K, Na, Ca, Mg, Al, B, Fe, Mn, Cu, Zn, Cd, Ca, Ni, Pb, Co, Mo, Si, S, As, Ti, Be, Sr, Rb, Li, V, and Ba by ICP-AES.* [Source] *Tadon, H., M.P. Cuescas, and E.H. Tyner. 1968. An acid-free vanadate-molybdate reagent for the determination of total P in soils. Soil Sci. Soc. Am. Proc. 32:48-51.*
- *"Mehlich III Extractable Nutrients: A 3 g sample of air dried soil is shaken with 30 mL of Mehlich III extracting solution [0.2 N CH_3COOH, 0.25 N NH_4NO_3, 0.015 N NH_4F, 0.013 N HNO_3, and 0.001 NEDTA] for 5 min. and then centrifuged. The supernatant is analyzed for Ca, Cu, K, Mg, Mn, P, and Zn by ICP-AES.* [Sources] *(Mehlich A. 1984. Mehlich 3 soil test extractant: a modification of mehlich 2 extractant. Commun. in Soil Sci. Plant Anal. 15:1409-1416.) (Fassel, V.A., and R.N. Kniseley. Nov. 1974. Inductively Coupled Plasma Optical Emission Spectroscopy. Anal. Chem. 46 (13):1110A-1120A. Also: Dahlquist, R.L. and J.W. Knoll. 1978. Inductively Coupled Plasma-Atomic Emission Spectrometry: Analysis of biological materials and soils for major trace, and ultra-trace elements. Appl. Spectroscopy 32:1-30. ICP: ARL (Fisons) Model 3560 ICP-AES Thermo Instrument Systems Inc. (Fisons Instruments Inc. Division) 81 Wyman Street PO Box 9046 Waltham, MA 02254."*

1.2 RESULTS AND DISCUSSION

1.2.1 Batch Tests for Phosphorus Release

The phosphate release from each compost in controlled batch test experiments is illustrated in Figures 2, 3, and 4. There are two important conclusions to draw from this information:

1) the amount of phosphate released (µg) increases as the amount of compost (g) increases; and
2) the amount of phosphate release varies by site and by compost type (yard vs. food residue).

For all composts, the amount of phosphate released increases as the compost mass increases, as indicated by the positive slope of the linear regression between phosphate release (µg) and compost mass (g). It is important to note, though, that the intercept of the linear regression does not equal zero for any of the fits. The intercept represents the phosphate release when compost mass equals zero and is a positive value for all regressions. The positive intercept for all regressions and lack of fit for some experiments suggests that the relationship between phosphate release and compost mass may not be linear for all compost masses. It is likely nonlinear as the compost mass approaches zero.

Two sites provided food residue composts as shown in Figure 2. The regressions appear to fit the data well ($R^2 \geq 0.9$) and all trials appear to provide consistent data (95% confidence interval bounds are close to the regression line). The slope of the linear regressions represents the amount of phosphate released (µg) per mass of compost (g). These values (21.6 µg P / g compost for Site 1; 63.5 µg P / g compost for Site 2) will be used in later analysis. Compared to each other, Site 2 released more phosphate than Site 1, which demonstrates the variability in phosphate release from different sites.

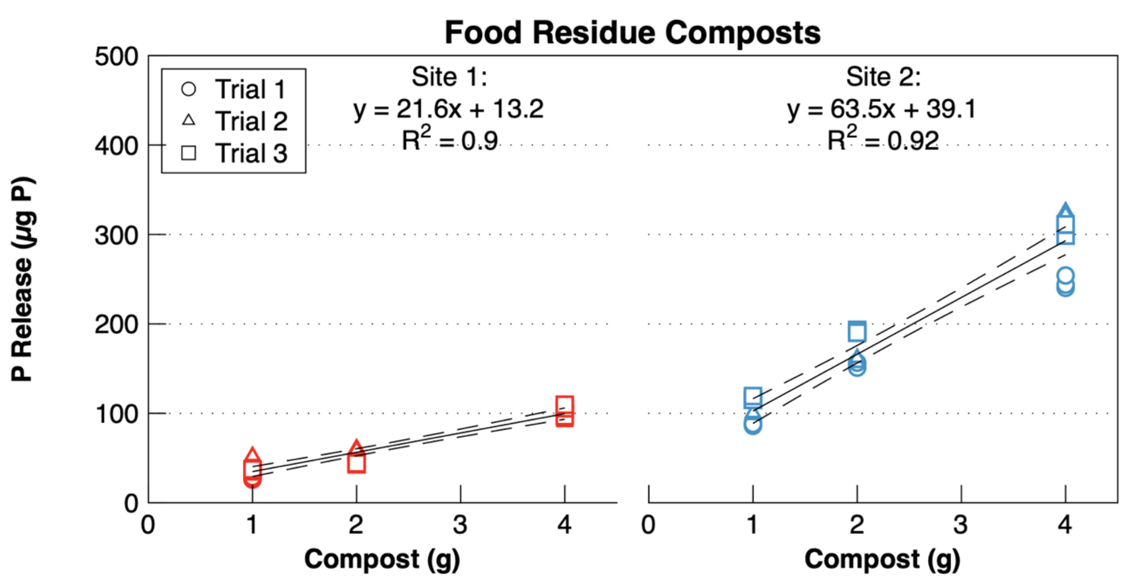

Figure 2: Phosphate release from food residue composts. Dashed lines represent 95% confidence interval on the regression.

Four sites provided yard waste compost as shown in Figure 3. The regressions appear to fit the data well for sites 1 and 2 ($R^2 \geq 0.86$) and all trials appear to provide consistent data (95% confidence interval bounds are close to the regression line). The regressions for Sites 3 and 4 are less predictive of the data ($R^2 \leq 0.66$) and the results from the different trials varied more (large bounds for 95% confidence intervals). Some data were excluded from the regressions for both Sites 3 (compost = 4g, trial 3) and 4 (compost = 2g, trial 1) because they appear to be outliers. The variability is apparent in the remaining data for Sites 3 and 4 and demonstrate the inherent variability within compost from a single site. In other words, a small sample of compost from a specific site may vary compared to a small sample from the same site at the same time. This appears to be true for Sites 3 and 4, and less true for Sites 1 and 2. The slope values (73.5 µg P / g compost for Site 1; 45 µg P / g compost for Site 2; 71.3 µg P / g compost for Site 3; 55.2 µg P / g compost for Site 4) will be used in later analysis.

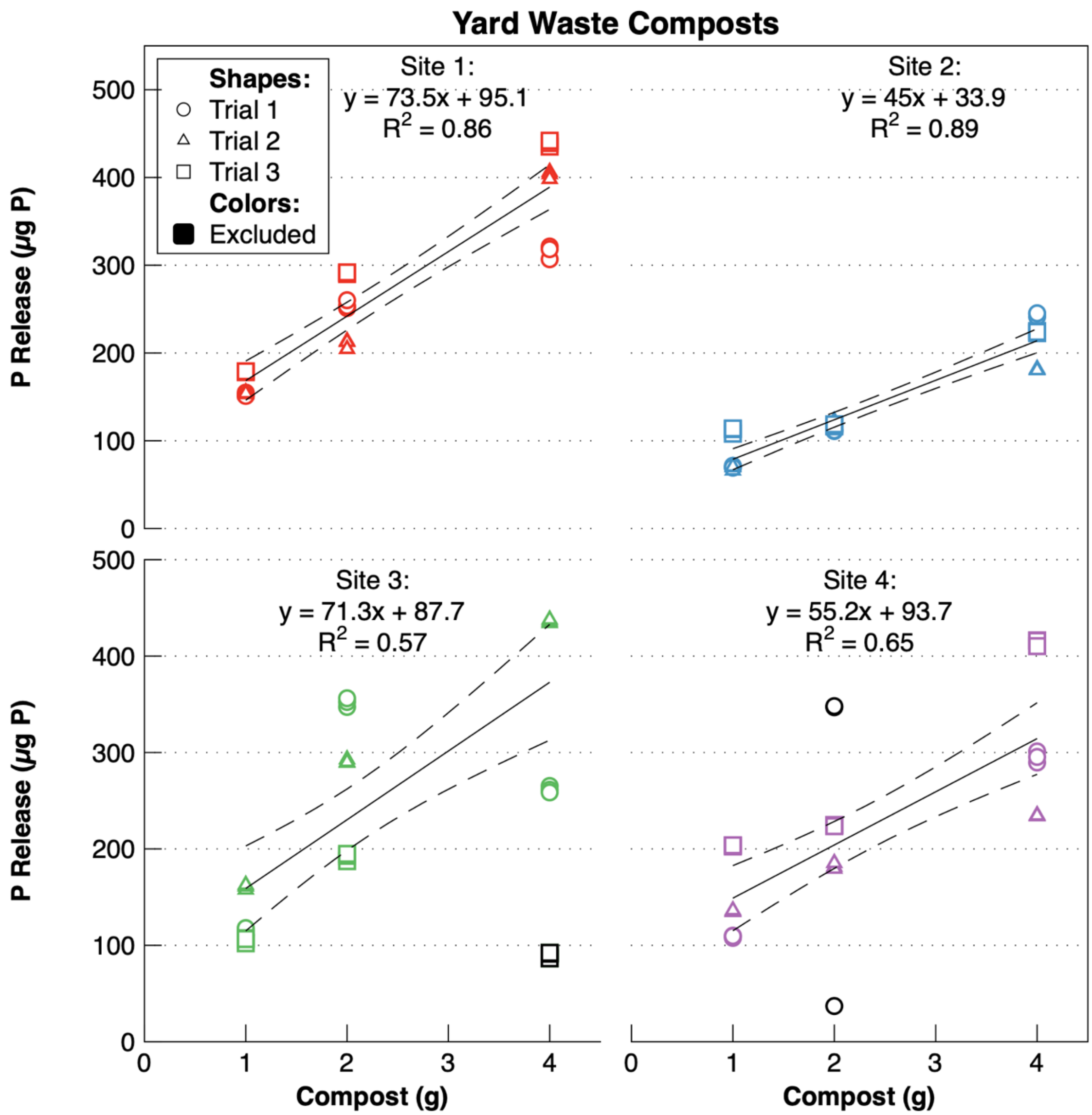

Figure 3: Phosphate release from yard waste composts. Dashed lines represent 95% confidence interval on the regression.

One site provided a blended (food + yard) compost as shown in Figure 4. The regression appears to fit the data well ($R^2 \geq 0.96$) and all trials appear to provide consistent data (95% confidence interval bounds are close to the regression line). The slope value (47.9 µg P / g compost for Site 5) will be used in later analysis.

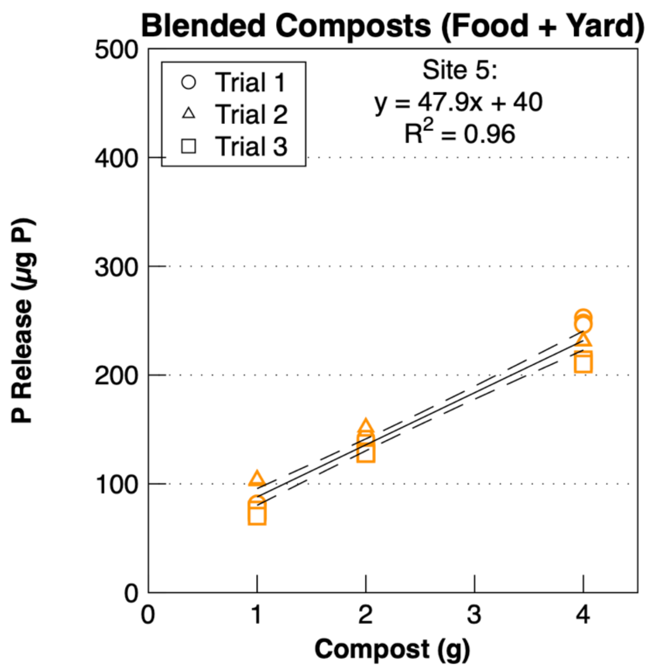

Figure 4: Phosphate release from blended (food + yard) compost. Dashed lines represent 95% confidence interval on the regression.

The three types of compost are compared in Figure 5. The food residue (Figure 5a) composts released less phosphate on average compared to yard waste (Figure 5b), and more than the blended (Figure 5c) compost. Site 1 food residue compost released less phosphate than Site 1 yard waste compost, but the inverse is true for Site 2 (food residue released more phosphate than yard waste). This demonstrates the variability between types of compost and between sites. The variability between sites is further illustrated by the range of phosphate release for each type of compost in Figure 5. For example, the phosphate release for yard waste composts (Figure 5b) ranges from 70 µg to over 200µg at 1g of compost and from 180µg to nearly 450µg for 4g of compost.

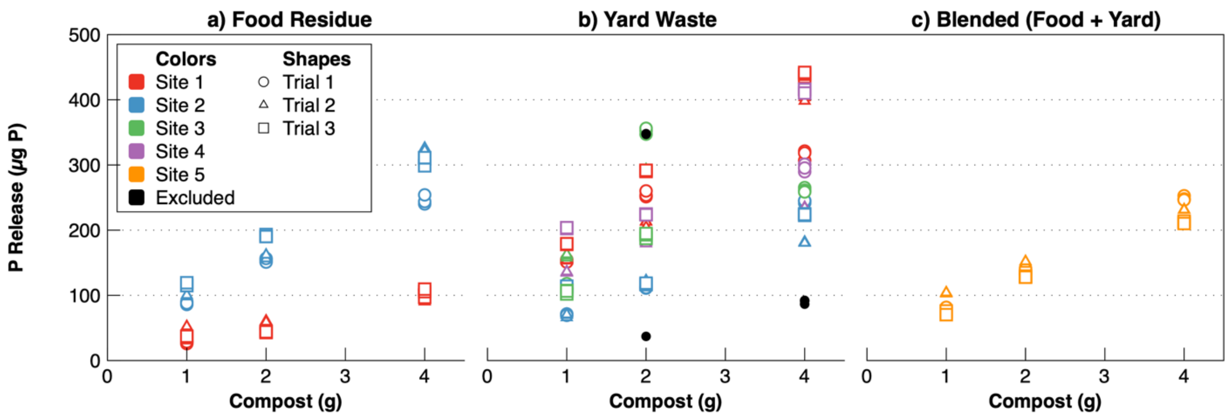

Figure 5: Food Residue Compost (a; 2 sites) compared to Yard Waste compost (b; 4 sites) and Blended (Food + Yard) compost (c; 1 site).

1.2.2 Solvita Tests for Compost Maturity

The Solvita test kits yielded data for CO_2, NH_3, and Solvita Maturity Index, which are illustrated in Figure 6 as a function of the phosphate release in µg P per g of compost. The phosphate release (µg P / g compost) values used in this plot correspond to the slope of the linear regressions from Figures 2, 3, and 4. The CO_2 values varied from 4 to 7 for all samples. One replicate out of 14 total samples produced an NH_3 value of 4, while all other samples produced a value of 5. As such, the NH_3 values had minimal effect on the Solvita Maturity Index Calculation. It's important to note that Solvita values greater than 4.5 are considered "Advanced composting" and values greater than 6 are considered "Practical Maturity," according to Figure 1. The relationship between Solvita Maturity Index and phosphate release (µg P / g compost) is poorly correlated ($R^2 < 0.4$) and the 95% confidence interval on the regression vary between ±1 and ±2 over the range of the data. Thus, Solvita Maturity index is not a good predictor of phosphate release.

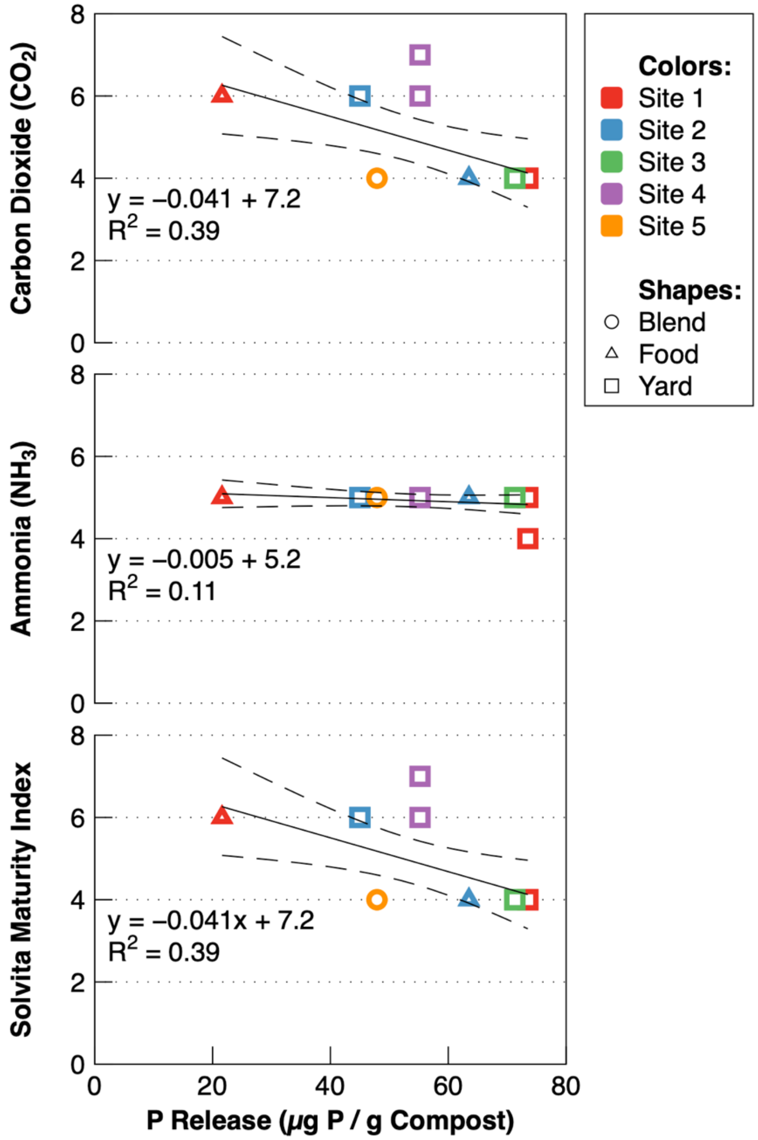

Figure 6: Carbon Dioxide, Ammonia, and Solvita Maturity Index for seven compost samples.

1.2.3 Media Characteristic Tests

The results from Olsen P, Bray P, Mehlich-III P, and ICP-OES P are shown in Figure 7 as a function of the phosphate release in µg P per g of compost. Of these four laboratory analytical techniques, the Mehlich-III P (mg P /kg media) data correlated best with P release (µg P / g compost), though the correlation is weak (R^2 = 0.46), and the 95% confidence interval bounds are large compared to the values of the regression. The correlations between P release (µg P / g compost) and Olsen P, Bray P, ICP-OES P were weak ($R^2 \leq 0.29$).

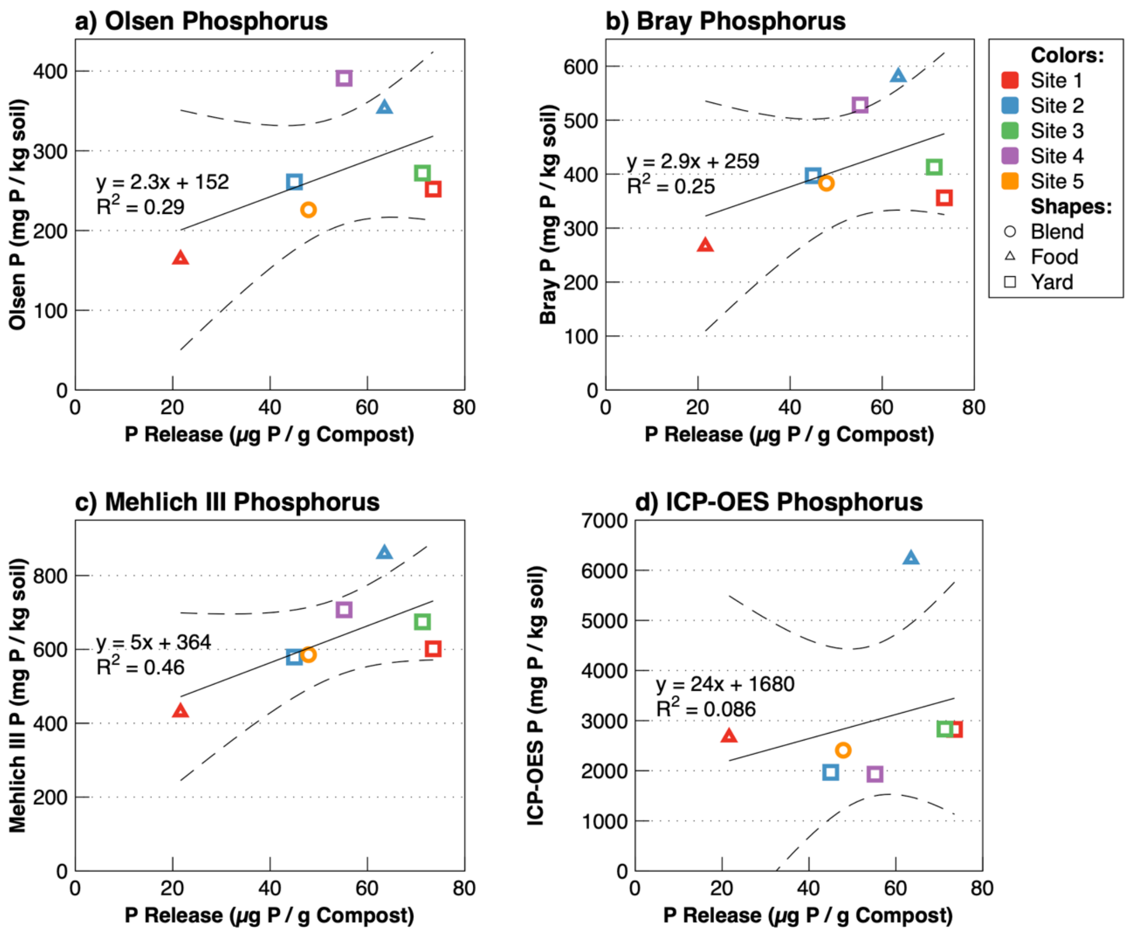

Figure 7: Olsen P, Bray P, Mehlich-III P, and ICP-OES P results compared to phosphate release (µg P / g compost).

1.2.4 In-field Batch Experiments

Two HACH® phosphorus test kits (PO-19A High Turbidity; PO-14) were purchased and used to determine whether in-field batch tests could be used to directly estimate phosphate release from compost samples. Approximately 40 grams (1/4 cup) compost was mixed with 273 mL (1 cup = 8 fl. oz) of water and placed on an orbital shaker table for 15 minutes at 1500rpm. A sample of the water was then processed according to the test kit instructions to determine the P concentration. Two junior scientists conducted the experiments independently and neither could determine the P concentration. While the instructions were easy to follow, the provided filtration system (filter paper) could not filter the sample via gravity due to a large amount of suspended compost particles. For an unfiltered sample, the resulting color of the sample could not be matched to the example colors on the provided color wheel. It is suspected that the turbidity affected the color of the samples, making it challenging to match the sample to the color wheel. It is also possible that tannins from the compost also influence the color. Both junior scientists recommended that this procedure not be considered for practitioners.

CHAPTER 2: Outdoor Mesocosm Experiments

2.1 METHODS AND MATERIALS

To evaluate the impact of biofiltration media mixes on filtration rate, nutrient output, and vegetation growth, outdoor mesocosm experiments were completed in the decommissioned spillway between the St. Anthony Falls spillway and St. Anthony Falls Laboratory in Minneapolis, Minnesota. In total, experiments were conducted for a total of four years (Y1, Y2, Y3, and Y4) with 14 simulated rainfall events from July 18 through October 24, 2019 (Y1); 8 simulated rainfall events from August 21 through October 15, 2020 (Y2); 12 events from July 22 until October 14, 2021 (Y3), and 5 events from August 3 through August 31, 2022 (Y4). The experimental setup, data recording, acquisition, processing, and analytical methods are described in the following sections.

2.1.1 Mesocosm Construction and Mixes

The media mixes were derived from local Minnesota guidance within the Minnesota Stormwater Manual, Design Criteria for Bioretention (MPCA 2022). In general, clean washed construction sand (e.g., AASHTO M-6 or ASTM C-33) is the primary granular filter media for nearly all biofiltration media mixes. Compost or peat was added as the organic material, and various amendments or enhancements were added to increase pollutant removal.

The mesocosm experiments were performed on ten different biofiltration media mixes in Y1 and Y2. Plaisted Companies, Inc. (https://plaistedcompanies.com/, Elk River, MN) provided the following materials: sand, leaf compost, sphagnum peat, reed sedge peat, biochar (#4 size provided by Plaisted Companies, sourced from Royal Oak Charcoal), and iron filings (d_{50} ~0.75mm; provided by Plaisted Companies, sourced from Connelly GPM, Inc.). Food compost (The Mulch Store; Rosemount, MN, https://www.mulchstoremn.com/empire.html) and spent lime (St. Paul Regional Water Treatment Facility, https://www.stpaul.gov/departments/water-services) were provided by other suppliers. Plaisted Companies, Inc. mixed the sand and amendments at various proportions (by volume) as listed in Figure 8. A total of thirty mesocosms were constructed; 10 different media mixes and three replicates of each mix.

In Y3, some media mixes were removed, and new mixes were constructed and evaluated. Referring to Figure 8, the mixes that were removed included the 10% and 20% food residue compost mixes, the 10% leaf compost mix, and 20% reed sedge peat mix, and the 5% iron mixed with 20% leaf compost mix. The new mixes were derived from Y1 and Y2 results in pursuit of one or more 'optimized' media mixes that could meet all project objectives, and included mixes of 100% clean washed sand (Y3 new baseline), 10% leaf compost, 10% spent lime + 10% leaf compost, 10% sphagnum peat + 10% leaf compost, and a layered mix in which 10% leaf compost mixed with 90% sand comprises the top layer while 5% iron mixed with 95% sand comprises the bottom layer (see Figure 9). A total of fifteen mixes were removed after Y2 (5 media mixes with 3 replicates of each) and fifteen new mixes were added for Y3 (five new mixes with three replicates each).

4	3	5	1	2	10	6	1	4	9	1	2	7	8	5	6	3	8	7	7	8	10	5	4	6	9	3	9	10	2

Y1 & Y2 Mix ratios by volume:
1: 100% Clean Washed Sand
2: 90% Sand + 10% Leaf Compost
3: 90% Sand + 10% Food Compost
4: 80% Sand + 20% Leaf Compost
5: 80% Sand + 20% Food Compost
6: 80% Sand + 20% Sphagnum Peat
7: 80% Sand + 20% Reed/Sedge Peat
8: 75% Sand + 20% leaf + 5% iron
9: 75% Sand + 20% leaf + 5% spent lime
10: 65% Sand + 20% leaf + 15% biochar

Figure 8: Diagram of media mix randomization scheme treatment placement (top), biofiltration media mixes (bottom left), and photo of completed setup (bottom right) for Y1 and Y2.

4	3	7	1	5	10	6	1	4	9	1	8	7	7	2	6	7	5	3	8	2	10	3	4	6	9	5	9	10	2

Y3 Mix ratios by volume:
1: 100% Clean Washed Sand (from Y1 Y2)
2: 100% Clean Washed Sand (new Y3)
3: 90% Sand + 10% Leaf Compost (new Y3)
4: 80% Sand + 20% Leaf Compost (from Y1 Y2)
5: 80% Sand + 10% Leaf Compost + 10% Sphagnum (new Y3)
6: 80% Sand + 20% Sphagnum Peat (from Y1 Y2)
7: 80% Sand + 10% Leaf Compost + 10% Spent Lime (new Y3)
8: 90% Sand + 10% Leaf Compost Layer OVER 95% Sand + 5% Iron Layer (new Y3)
9: 75% Sand + 20% leaf + 5% spent lime (from Y1 Y2)
10: 65% Sand + 20% leaf + 15% biochar (from Y1 Y2)

Figure 9: Diagram of media mix randomization scheme treatment placement (top), biofiltration media mixes (bottom left), and photo of completed setup (bottom right) for Y3.

For Y4, some mesocosms were removed and new mixes were added for the purpose of testing the impacts of salt loading on nutrient release from biofiltration media mixes. All mixes from Y1 and Y2 were removed while some mixes from Y3 were kept, including 100% sand (two replicates), 10% leaf compost (two replicates), and the iron-compost layered mix (three replicates). A total of 23 mesocosms were removed after Y3. The new mixes added for Y4 included 100% sand (six replicates), 10% leaf compost (six replicates), and the iron-compost layered mix (six replicates). A total of 18 mesocosms were added in Y4; six replicates for each new mix were added such that three replicates can receive salt loading while the other three replicates do not (see Figure 10).

9	5	1	6	7	4	5	6	9	4	1	8	7	6	2	5	8	4	3	8	2	9	3	7	1

Y4 Mix ratios by volume:
1: 100% Clean Washed Sand (new Y4)
2: 100% Clean Washed Sand (Y3)
3: 90% Sand + 10% Leaf Compost (Y3)
4: 90% Sand + 10% Leaf Compost (new Y4)
5: 100% Clean Washed Sand w/ Salt Loading (new Y4)
6: 90% Sand + 10% Leaf Compost w/ Salt Loading (new Y4)
7: 90% Sand + 10% Leaf Compost Layer OVER 95% Sand + 5% Iron Layer (new Y4)
8: 90% Sand + 10% Leaf Compost Layer OVER 95% Sand + 5% Iron Layer (Y3)
9: 90% Sand + 10% Leaf Compost Layer OVER 95% Sand + 5% Iron Layer w/ Salt Loading (new Y4)

Figure 10: Diagram of media mix randomization scheme treatment placement (top), biofiltration media mixes (bottom left), and photo of completed setup (bottom right) for Y4.

To construct the mesocosms, thirty 22-gallon cylindrical plastic containers with a top diameter of 15.75 inches and a height of 30 inches were purchased, scrubbed with phosphorus-free soap, rinsed with ultrapure water (Milli-Q, 18.2 MΩ·cm), and lined up on an outdoor platform which elevated them between 22.5 and 27 inches above the ground. A plastic bulkhead was installed by drilling a hole into the bottom of each barrel, and approximately 20 inches of UV-resistant black PVC tubing (McMaster-Carr, https://www.mcmaster.com/) with 3/4 inches inner diameter was attached to the bulkhead. Black tubing was selected over clear tubing to minimize photosynthetic reactions which would inhibit water flow. Approximately 4 inches of pre-washed pea gravel (d_{50} = 0.5 cm; Plaisted Companies), was added to the bottom of each column for drainage. Each dry biofiltration media mix treatment was added in approximately four 4.5-inch lifts until approximately 18 inches of media had settled in each column. In between each lift, the sides of the column were tapped with a rubber mallet to promote settling of particles. For each mixture, the same mass was added for two additional replicates (Y1, Y3, and Y3) or for six replicates (Y4). A schematic of the mesocosm construction and dimensions is shown in Figure 11.

The experimental setup for simulated runoff events consisted of 5-gallon influent buckets elevated approximately 13 inches above the surface of each mesocosm, with 11-inch-long black tubing (5/8-inch diameter) and an inline valve (~4 inches from the end of the tubing). The tubing from the influent buckets were placed into flow dissipators, which were placed on the surface of each mesocosm to prevent the media surface from scouring (Figure 12). Flow dissipators were constructed by securing 8-inch-tall mesh wire (1/4 inch openings) around 4-inch plastic caps and filling with pea gravel placed around 1-inch diameter PVC pipes, which would stabilize influent tubing during experiments. Each dissipator was stabilized with wooden dowels pressed against the inner sides of the mesocosm.

In Y1, thirty digital scales (ACCUTECK- All-in-1 series W-820) were placed below and in front of the mesocosms on concrete tiles. Each scale was leveled, set to kilograms, and tared. The scales had an average error of 0.32%. A wooden brace was placed on top of the scale to level and stabilize the effluent collection buckets. The effluent buckets with their lids were placed on the scales and the black tubing from each mesocosm bulkhead was inserted into the hole within the lid. The scales measured the weight of the effluent buckets continuously, but the readout was recorded on printed datasheets at regular intervals throughout the experiments.

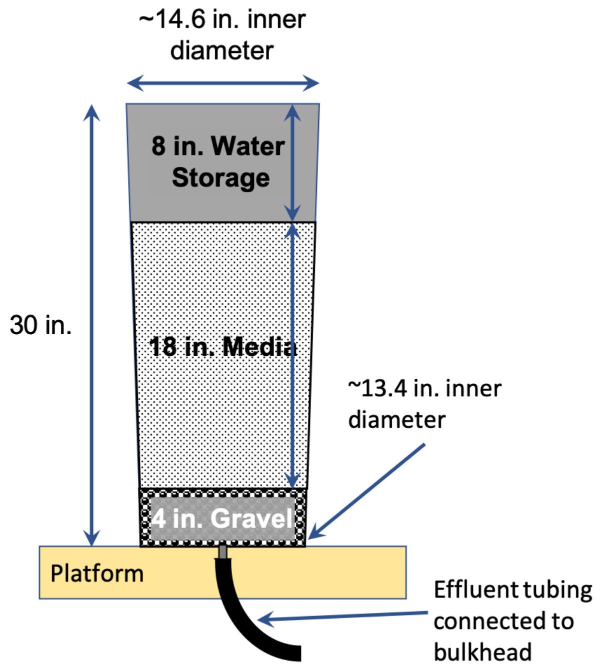

Figure 11: Mesocosm schematic with approximate dimensions (not to scale).

Figure 12: Side (left) and aerial (right) photos of dissipater columns.

For Y2-Y4, the digital scales were replaced with electronic load cells (Pull Pressure Force S-Type Load Cell Sensor 30KG and SparkFun Electronics model SEN-13261) were installed above the influent and effluent collection buckets and data were continuously recorded using a Raspberry Pi® system (Raspberry Pi 3 Model B+ Board (3B+). Data was downloaded regularly and backed up on University-controlled data management and storage.

Each mesocosm was seeded with 0.4 g of pure live switchgrass *(Panicum virgatum)* seed (seeding rate = 0.29g/ft^2, Prairie Restorations, Princeton, MN, http://www.prairieresto.com/) by evenly scattering across four 0.34 ft^2 quadrants. After seeding, mesocosms were watered twice weekly until experiments began with 1 L of Mississippi River water, collected near downtown Minneapolis, MN, USA. In Y1, Y2, and Y4, each new mesocosm was covered with transparent plastic until the start of experiments to prevent mesocosm disruption from rainfall and to promote seed germination.

Between simulated events, filtered rainwater and any residual experiment effluent that drained from each mesocosm was collected in clear, 6-quart plastic containers (13 5/8 inches long, 8 1/4 inches wide, and 4 7/8 inches tall, Figure 13). The effluent tubing from each mesocosm - 31 inches in length - was fitted through a hole drilled into each container's lid. In Y1, an additional lid-less container was placed beside the experimental setup to collect rainwater between August 6, 2019, and October 28, 2019. Rainwater was not measured in Y2-Y4.

Figure 13: Rainwater collection bins underneath mesocosms.

2.1.2 Simulated Stormwater Runoff Events

To prepare for the events, a 550-gallon storage tank was cleaned and scrubbed with phosphorus free soap and rinsed with Mississippi River water. The tank was filled with approximately 250 gallons of Mississippi River water extracted with a hose and pump. During the filling process, three 50 mL samples of the Mississippi River water were collected and placed in the freezer. Based on previously measured background soluble reactive phosphorus (SRP) concentration in the Mississippi River water, K_2PO_4 (99.9%, J.T. Baker) was added to the tank when it was approximately half full to achieve a target SRP concentration of 300 µg/L. Previous studies in the Twin Cities have shown average dissolved P concentrations in stormwater to be 200 µg/L and 90 µg/L (Brezonik & Stadelmann, 2002; Janke et al. 2017, respectively). We used a target concentration of 300 µg/L to increase the difference between the influent and the level of detection (10 µg/L). A kayak paddle was used to fully mix and dissolve the K_2PO_4, and then the lid was placed on the tank, and it was allowed to equilibrate overnight.

To simulate the runoff events, influent buckets were filled to the edge of the buckets (20.4L ± 0.2L) with water pulled from the tank using a sump pump and hose system. Plastic sheets were placed on top of the mesocosms and behind the mesocosms while the influent buckets were filled to prevent any water that was spilled during the filling process from entering the mesocosms. Three samples were collected while filling buckets, one at the start, one at the middle and one at the end of filling. In Y4, approximately 155g of salt (NaCl, Morton© Table Salt) was added to each influent bucket (~7.75 g NaCl per L) corresponding to a mesocosm designated to receive salt loading. The salt was added while the bucket was filled such that filling the bucket and swirling would dissolve the salt as much as possible. The temperature and pH of the tank was recorded after all the buckets were filled.

Valves were opened consecutively, the first valve to be opened was randomly selected. Once the water had drained from the influent buckets freely, the buckets were manually tipped to ensure all water had been drained completely. In Y1, effluent mass recording began as soon as possible after the valves had been opened. For Y2-Y4, influent and effluent mass were recorded continuously by the load cell and Raspberry Pi system. An example of the inflow and outflow characteristics from Y2 is shown in Figure 14, in which the blue line represents the bucket being filled, the influent volume when full, and the quick decrease when the valve was opened. The red line represents the effluent bucket being filled over time. For Y1-3, statistical analyses of the results showed significant, but small differences between the measured filtration rate for different mixes. On average, 50% of the influent volume passed within 5-15 minute and 75% passed within 20 minutes (data not shown; Erickson et al. 2021).

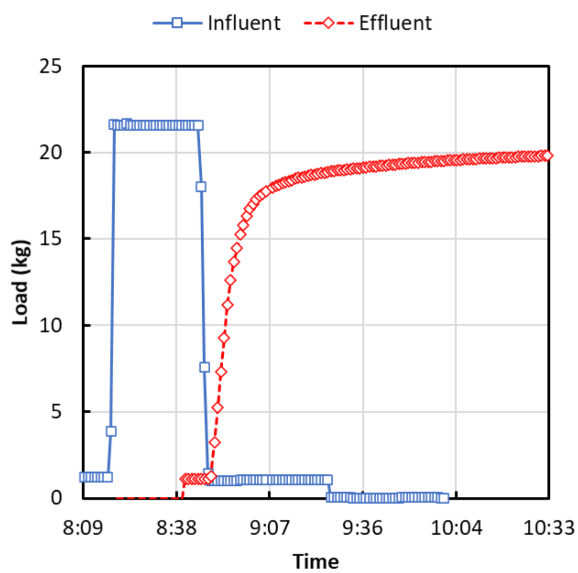

Figure 14: A graph depicting the typical influent and effluent flow characteristics of the mixes.

The mesocosms were allowed to collect effluent for two hours and then triplicate 50 mL samples in new plastic tubes were collected from the effluent buckets. Water was collected, swirled and dumped three times before saving the final sample. The temperature of each effluent bucket was recorded. In Y3 and Y4, pH and conductivity were measured in addition to temperature. After conclusion of the experiment, influent buckets were rinsed with River water, effluent buckets were washed with phosphorus free soap, rinsed with ultrapure water (Milli-Q, 18.2 MΩ·cm) and all buckets were stored on raised platforms and secured with netting until the next event.

2.1.3 Monitoring In Between Simulated Events

2.1.3.1 Vegetation

Overhead photos of each mesocosm were taken on a weekly basis to document vegetation growth. To monitor growth throughout the experiment, the number of switchgrass sprouts, the height of the tallest sprout, and the height of the shortest sprout in each mesocosm were measured. A more robust determination of media's impact on vegetation success involved gathering the above ground biomass in each mesocosm in paper bags at senescence in Y1-Y3, dehydrating it in an oven until it was completely desiccated, and massing it. Any non-switchgrass biomass was measured separately.

2.1.3.2 Settling

Consolidation (i.e., settling) of the media within the mesocosms was tracked by measuring the depth of the media surface relative to the top of the mesocosm on the same weekly interval as the vegetation measurements. Overall, there was no statistical difference in settling between the treatments and settling within all columns averaged 0.73 inches ± 0.35 inches (standard deviation) in Y1. Settling in new mesocosms in Y3 and Y4 was similar magnitude. Settling in mesocosms after the first year was minimal.

2.1.3.3 Rainfall

Plastic tubs placed below the mesocosms in between experiments allowed for rough tracing of water and nutrient concentration movement through each column. Both rainfall and residual experimental water contributed to these. Either scales or graduated cylinders with thermometers were employed for water mass measurements in Y1. The load cell and Raspberry Pi system was used for rainfall measurement in Y2-4. Standard sampling protocol applied when enough water was in the tubs for Y1.

2.1.4 Data Recording, Analysis, and Exclusions

In Y1, field data sheets were used to collect filtration rate, water temperature, and observational data on experiment days. They were then scanned, entered, and error corrected. All other notes, observations, and data regarding rainfall, vegetation, and sediment settling measurements were recorded in a field notebook. In Y2, mass of influent and effluent buckets was recorded using the load cell and Raspberry Pi system.

Following a thorough quality assurance/quality control review, some exclusions and adjustments were needed to complete the data analysis. For phosphorus analysis, the tank concentration for event 14 of Y1 was missing and therefore the tank concentration was estimated by taking the average of all other tank values measured during Y1. In Y1, events 1 and 2 were not included in analyses and in Y2 event 7 was excluded due to unreliable concentration measurements from the Lachat (i.e., poor QA/QC results). For nitrate analysis, event 1 from Y3 was removed from the data set due to measurements that were 2-20 times higher across treatments than measurements from the later events and other years. For vegetation biomass analysis, values less than 0.01g were assumed to be 0.01g (measurement limit of the scale) for mesocosm 8 (sand) in Y2 and Y3 and mesocosm 30 (new sand) in Y3. Mesocosm 1 (20% leaf) showed little vegetation growth and was excluded from analysis for Y1 and Y2. In Y3, mesocosms 14 (10% spent lime/10% leaf), 25 (sphagnum), 27 (10% sphagnum/10% leaf) and 29 (biochar) were removed from analysis due to a loss of vegetation before harvest. Biomass was normalized to 100 growing days by taking the biomass measurement, dividing it by the number of growing days and then multiplying that by 100. For vegetation height analysis, the average maximum height value for each mesocosm for the entire year was used to compile the data set and the data was not transformed. The pH data was not transformed. The phosphate, nitrate and biomass measurements were natural log transformed.

2.1.5 Wet Chemistry Analytical Techniques

During Y1, influent and effluent samples were frozen after they were collected. To prepare for analysis, they were either allowed to thaw at room temperature for approximately 12 hours or allowed to thaw in the refrigerator at 5°C for 48-72 hours. The samples were then centrifuged for 15 minutes at 2800rpm. Immediately following the centrifuge, the samples were filtered using a 10mL syringe and 0.45-micron filter. The samples were sub-sampled into 3 different 15mL conical tubes; one for phosphate, one for nitrate and one supplemental. The nitrate and supplemental samples were placed back into the freezer. The phosphate samples were then analyzed or stored at 5°C and analyzed within two days. For Y2-Y4, the samples were centrifuged and filtered immediately following the simulated event. The phosphate samples were then either immediately analyzed or stored at 5°C and analyzed within two days.

Phosphate concentrations were analyzed using flow injection analysis by a Lachat Instrument (Milwaukee, Wisconsin) Quick-chem model 8000, method 10-115-01-1 with a statistically determined detection limit of 5 µg P/L. Prepared phosphorus standards were used to calibrate the Lachat before analysis ranging from 0-1500 µg P/L. Check standards were run every 15 samples using the calibration standards. Analytical duplicates were run every 10 samples for quality assurance and quality control. After every check standard and every analytical duplicate, the instrument was flushed with Disodium ethylenediaminetetraacetic acid (EDTA) and ultrapure water (Milli-Q, 18.2 MΩ·cm). Calibration curves and measured concentrations were recorded digitally from the Lachat software and exported into Excel for further QA/QC and statistical analysis. A subset of effluent samples was tested for Nitrate (nitrate+nitrite, mg/L) concentration using the cadmium reduction method (Events 5, 7, 9 and 13 of 14 total events were sampled in Y1; events 2, 4, 6 and 8 of 8 total events were sampled in Y2; and events 1, 3, 5 and 7 of 12 total events were sampled in Y3). Media characteristics (for Y3 only) and nitrate analysis was performed by the Research Analytical Laboratory on the St. Paul campus of UMN (https://ral.cfans.umn.edu/). Media characteristic analysis included Mehlich-III P, Bray P, Olsen P, LOI, extracted nitrate, total nitrogen and total organic carbon (Table A1 in Kramarczuk 2022).

2.2 RESULTS AND DISCUSSION

2.2.1 Phosphate

Phosphate is a primary concern of stormwater managers with impaired freshwater lakes. Because biofiltration systems have underdrains, any phosphate that is released from biofiltration media can be delivered into the storm sewer system, which commonly drains to lakes and rivers. As such, a question from stormwater managers is what media components (e.g., compost, peat, biochar) affect phosphate concentration, both capture and release. The mesocosm experiments were designed to investigate possible answers to this question. Table 2 lists the different media mixes that were installed in the mesocosms, which years the media mixes were tested, and short names for each mix that corresponds to text and figures in the remainder of this report. For a thorough explanation of statistical results, refer to Erickson et al. 2021 and Kramarczuk 2022.

Table 2: Summary of mixes for all years, sorted alphabetically by short names.

Media Mix	Years Tested	Short Name
90% sand, 10% food compost	Y1-Y2	10% food
90% sand, 10% leaf compost	Y1-Y2	10% leaf
90% sand, 10% leaf compost (added)	Y3-Y4	10% leaf (Y3)
90% sand, 10% leaf compost (no salt)	Y4	10% leaf (Y4 no salt)
90% sand, 10% leaf compost (salt added)	Y4	10% leaf (Y4 salt)
80% sand, 10% spent lime, 10% leaf compost (added)	Y3	10% lime/10% leaf
80% sand, 10% leaf compost, 10% sphagnum peat (added)	Y3	10% sphag/10% leaf
100% Sand	Y1-Y3	100% sand (Y1)
100% Sand	Y3-Y4	100% sand (Y3)
100% Sand (no salt)	Y4	100% sand (Y4 no salt)
100% Sand (salt added)	Y4	100% sand (Y4 salt)
65% sand, 20% leaf compost, 15% biochar	Y1-Y3	15% biochar
80% sand, 20% food compost	Y1-Y2	20% food
80% sand, 20% leaf compost	Y1-Y3	20% leaf
75% sand, 20% leaf compost (by volume), 5% iron (by weight)	Y1-Y2	5% iron
75% sand, 20% leaf compost, 5% spent lime	Y1-Y3	5% spent lime
90% sand & 10% leaf compost (top half) layered over 95% sand & 5% iron (bottom half) (no salt)	Y4	compost-iron layered (no salt)
90% sand & 10% leaf compost (top half) layered over 95% sand & 5% iron (bottom half) (salt added)	Y4	compost-iron layered (salt)
90% sand & 10% leaf compost (top half) layered over 95% sand & 5% iron (bottom half)	Y3-Y4	compost-iron layered (Y3)
80% sand, 20% reed/sedge peat	Y1-Y2	reed sedge
80% sand, 20% sphagnum peat	Y1-Y3	sphagnum

The phosphate concentration from the mesocosm media mixes in Y1 is shown in Figure 15, with simulated events 3 through 14 shown as individual bars (error bars indicate one standard deviation) and each media mix shown as a color-coded group. As shown by the dashed black line across the plot and group of grey bars on the right side, the average inflow concentration into the mesocosms was ~200 µg/L in Y1. When plotting the outflow from the mesocosms, effluent concentrations that are greater than the inflow concentration indicate phosphate (P) release or export. Outflow concentrations that are less than the inflow concentration indicate phosphate capture within the mesocosm media. As shown in Figure 15, the 10% food, 10% leaf, 20% food, 20% leaf, 15% biochar, and 5% spent lime media mixes increased the phosphate concentration from inflow to outflow, indicating phosphate release. These mixes also increase effluent phosphate concentration compared to 100% sand. Therefore, the compost, biochar, and/or spent lime must be the source of the exported phosphate.

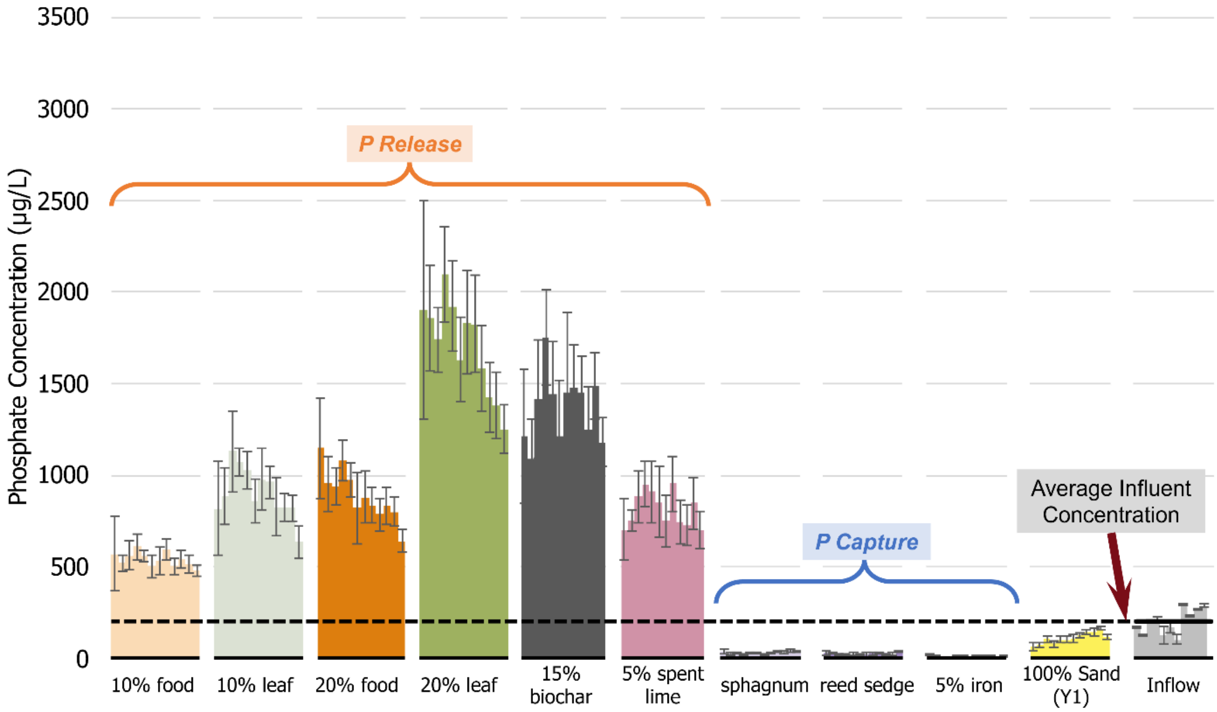

Figure 15: Phosphate concentration (bars) from mesocosm media mixes (color-coded groups) in Y1.

Figure 15 shows that the leaf-based compost exported more phosphate than the food residue compost in these mesocosm experiments, though not all treatments were statistically different (see Erickson et al. 2021 and Kramarczuk 2022). As noted in CHAPTER 1:, phosphate release from leaf-based or food residue compost varies substantially based on supplier and likely several other factors. The media mixes with 15% biochar and 5% spent lime exported phosphate, but the effluent concentrations were less than the effluent from 20% compost. If biochar or spent lime exported phosphate, we would expect the effluent concentration from the combination of biochar and compost to be greater than 20% leaf compost, which was not observed. Because the 15% biochar and 5% spent lime mixes also contained 20% leaf compost and these mixes release phosphate concentrations less than 20% compost, the biochar and spent lime appear to capture some phosphate that was released from the leaf compost. The biochar and spent lime at these mix ratios, however, were not sufficient to reduce the phosphate concentration below the inflow concentration, and thus exhibited a net release of phosphate.

The 20% leaf compost was replaced with 20% sphagnum peat or 20% reed sedge peat in the sphagnum and reed sedge media mixes, respectively. Both the sphagnum and reed sedge media mixes reduced the phosphate concentration from the inflow of ~200 µg/L to below 45 µg/L, which was unexpected and unexplained.

The 5% iron media mix reduced the phosphate concentration to below 20 µg/L, which is substantially less than the inflow concentration (200 µg/L) and the effluent from the 20% leaf compost (~1700 µg/L). The 5% iron mix also contains 20% leaf compost, and therefore these results indicate that 5% iron mixed throughout a granular media with 20% leaf compost can reduce phosphate concentration from the inflow and any phosphate that may be released from the compost. 100% sand provided minimal benefit which decreased over time. Previous research has shown that silica sand can capture some phosphate immediately after construction but has limited long-term capacity (Erickson et al. 2007, 2012).

The phosphate concentration from the mesocosm media mixes in Y2 is shown in Figure 16. The average inflow concentration into the mesocosms was ~320 µg/L in Y2 (dashed black line and group of grey Inflow bars). Similar to Y1, the 10% food, 10% leaf, 20% food, 20% leaf, 15% biochar, and 5% spent lime media mixes increased the phosphate concentration from inflow to outflow in Y2, indicating phosphate release. The media mixes with sphagnum or reed sedge peat, and the mixes with iron mixed with compost reduced phosphate concentration in Y2, similar to Y1. Because the climate conditions were different in Y2 compared to Y1 and were not thoroughly measured, comparing results from Y1 and Y2 would not be a conclusive determination of performance due to media aging. However, the trends observed in Y2 are similar to the trends observed in Y1, and thus the results appear consistent in both years despite differing climate conditions.

The phosphate concentration from the mesocosm media mixes in Y3 is shown in Figure 17. The average inflow concentration into the mesocosms was ~295 µg/L in Y3. Five media mixes from Y1 and Y2 were kept in Y3 (20% leaf, 15% biochar, 5% spent lime, sphagnum, and 100% Sand (Y1)), and five new media mixes were constructed in Y3. Of the mixes that were kept, the trends in effluent concentration for Y3 were similar to trends observed in Y1 and Y2 (20% leaf compost, 15% biochar, and 5% spent lime exported phosphate; sphagnum captured phosphate). This further demonstrates that the results for these mixes are consistent for the period of time over which they were measured.

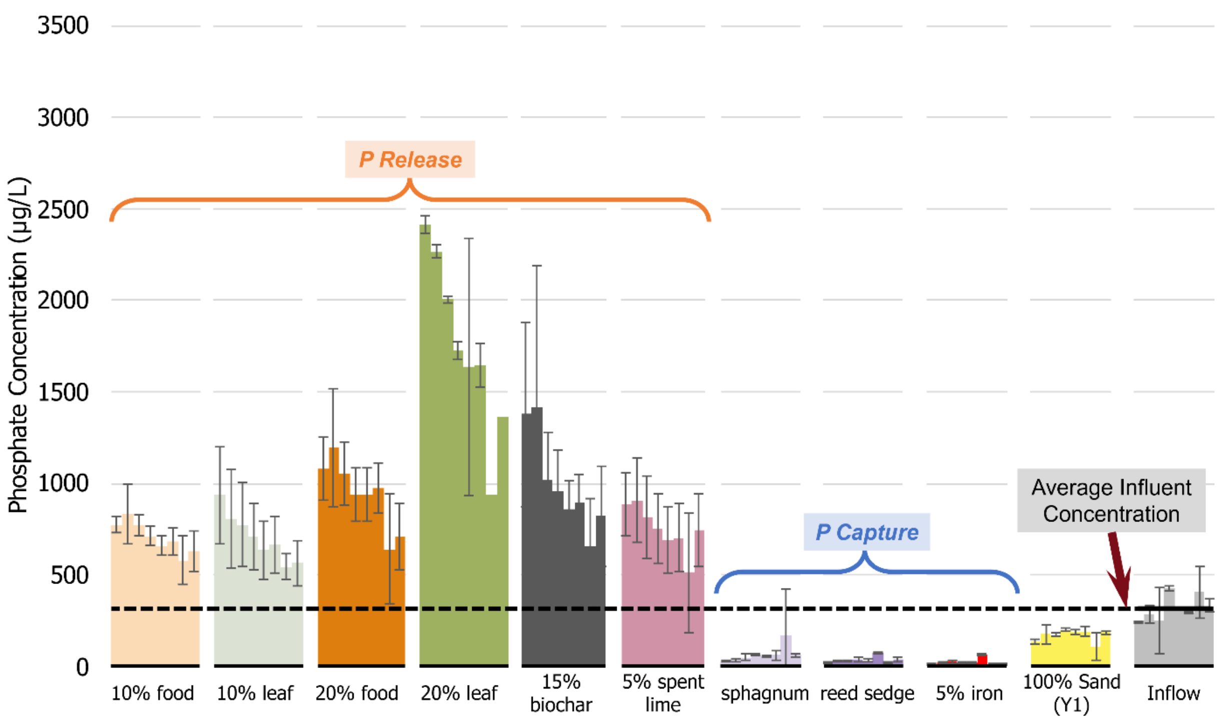

Figure 16: Phosphate concentration (bars) from mesocosm media mixes (color-coded groups) in Y2.

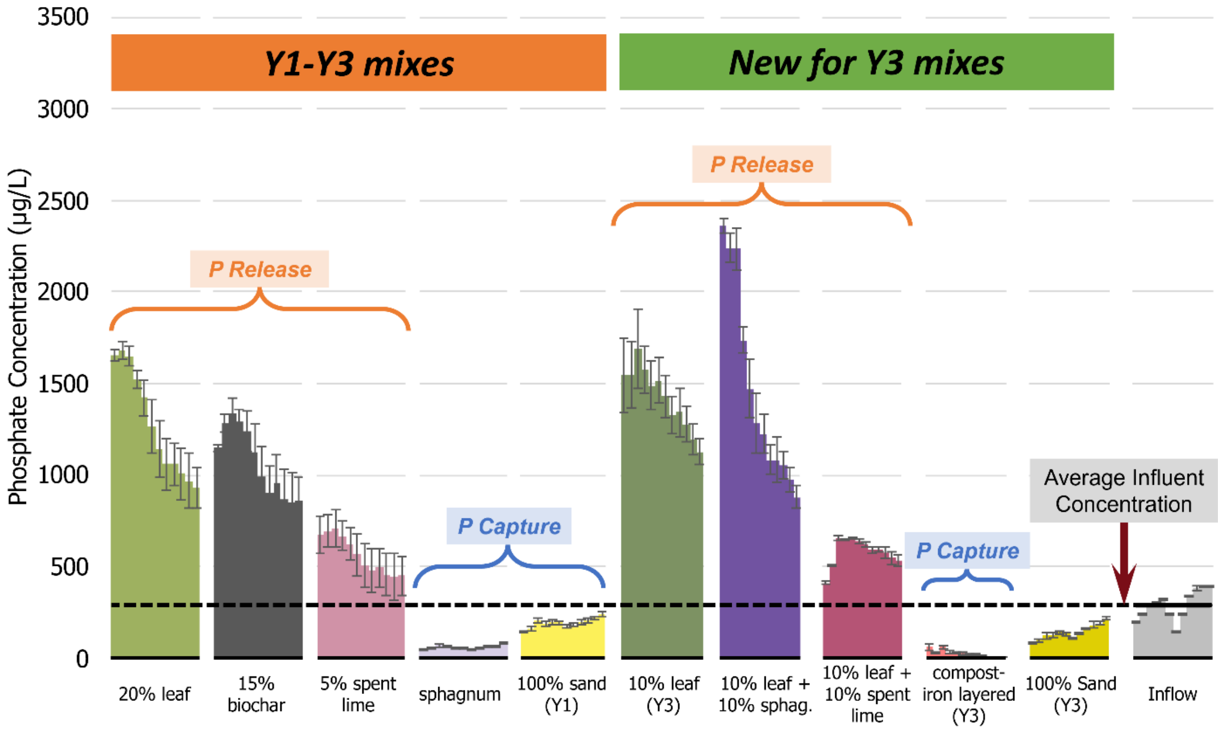

Figure 17: Phosphate concentration (bars) from mesocosm media mixes (color-coded groups) in Y3.

The new mixes added in Y3 included 10% leaf, a mix of 10% leaf compost + 10% sphagnum peat, a mix of 10% leaf compost + 10% spent lime, a layered mix of 10% compost over 5% iron, and a new 100% sand mix. In the pursuit of biofiltration media optimization, it was clear from Y1 and Y2 results that 20% compost had the potential to substantially increase the phosphate concentration and export phosphate. Thus, 10% compost was selected a new 'baseline' design mix and the new mixes for Y3 were based on 10% leaf compost. In Y3, 10% leaf compost exported phosphate such that the effluent concentration was ~1440 µg/L on average, which is a substantial increase from the average inflow of 295 µg/L. This is consistent with results from Y1 and Y2.

In Y1 and Y2, sphagnum peat reduced the phosphate concentration. Thus, it was hypothesized that a mix of 10% leaf compost with 10% sphagnum peat could result in effluent concentration near the inflow concentration. As shown in Figure 17, the 10% leaf + 10% sphagnum mix substantially increased phosphate concentration, which was not expected. Thus, this mix of leaf compost and sphagnum peat is not an optimal mix for reducing phosphate pollution. It should be noted, however, that the sphagnum peat and the leaf compost obtained in Y3 is not the same as the peat and compost obtained in Y1, even though they were sourced from the same supplier. It is unclear why the mix of sphagnum and compost exhibited phosphate release, but it could be due to variations in peat and/or compost composition or properties.

In Y1 and Y2, 5% spent lime mixed with 20% leaf compost produced effluent concentrations that were less than 20% compost, but greater than inflow concentrations. It was hypothesized that a mix of 10% leaf compost with 10% spent lime could result in effluent concentration near the inflow concentration. As shown in Figure 17, the 10% leaf + 10% spent lime mix increased the effluent phosphate concentration compared to the inflow. Thus, this mix of leaf compost and spent lime is not an optimal mix for reducing phosphate pollution.

In Y1 and Y2, 5% iron mixed with 20% compost reduced the effluent concentration from the inflow to below 25 µg/L on average. It was hypothesized that an optimal mix may comprise a 'growth layer' of 10% compost mixed with sand on top of a 'treatment layer' of 5% iron mixed with sand. As shown in Figure 17, the compost-iron layered mix reduced the phosphate concentration to an average of less than 30 µg/L. Thus, a compost-iron layered mix may be an optimal biofiltration media mix for reducing phosphate pollution.

The phosphate concentration from the mesocosm media mixes in Y4 is shown in Figure 18. The average inflow concentration into the mesocosms was ~360 µg/L in Y4. Three media mixes from Y3 were kept in Y4 (10% leaf (Y3), compost-iron layered (Y3), and 100% Sand (Y3)), and three new media mixes were constructed in Y3 with six replicates of each media mix. Of the mixes that were kept, the trends in effluent concentration for Y4 were similar to trends observed in Y3 (10% leaf compost exported phosphate; compost-iron layered mix captured phosphate). This further demonstrates that the results for these mixes are consistent for the period of time over which they were measured.

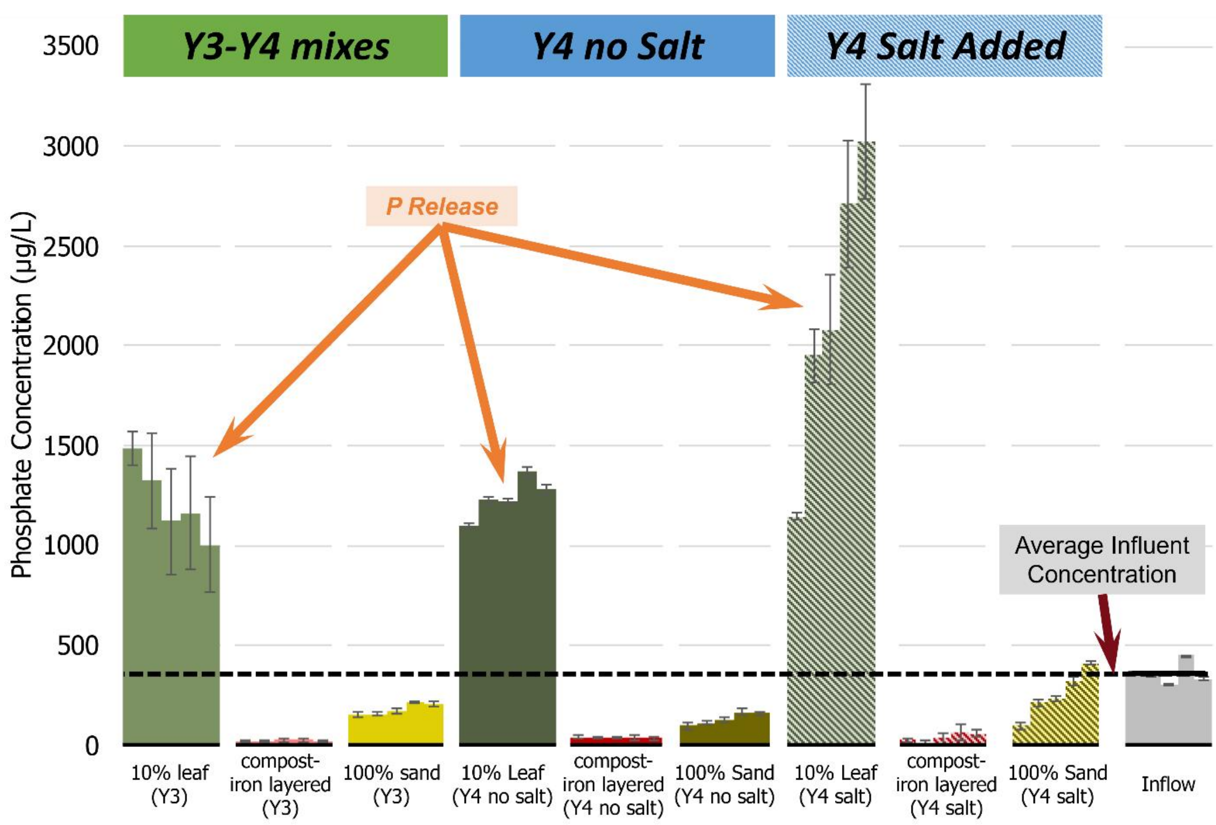

Figure 18: Phosphate concentration (bars) from mesocosm media mixes (color-coded groups) in Y4. Bars with hashed pattern are mesocosm media mixes that received salt.

A primary objective of Y4 was to test whether dissolved salt within the inflow would impact the phosphate concentration in the effluent from select biofiltration media mixes. In cold climates such as

Minnesota, salt in the form of sodium chloride (NaCl), magnesium chloride (MgCl), or calcium chloride (CaCl) is often applied on sidewalks, parking lots, and roadways to melt snow and ice, thus reducing vehicular and pedestrian accidents and injuries. When dissolved, this salt can mobilize ions that are attached to solids in a process similar to water softening rinse cycles in which salt-laden water is rinsed through the ion-absorption resin to release hardness-causing ions from the resin. Road salt can be concentrated in stormwater during winter melt events and spring snowmelt events. It was hypothesized that road salt in stormwater runoff could mobilize phosphate from biofiltration media mixes.

Three media mixes were selected for testing with and without salt in the inflow: 10% leaf compost, compost-iron layered, and 100% sand. For the mixes that did not receive salt, the 10% leaf compost released phosphate, which was expected given the results from Y1-Y3. The compost-iron layered mix reduced phosphate concentration, similar to Y3. The 100% sand also captured some phosphate. For the mixes that received salt (Figure 18, hashed bar plots), the effluent concentration for 10% compost was greater for the mix that received salt compared to the mix that did not receive salt. Statistical significance was not calculated for this report, but the effluent phosphate concentration is substantially greater in the mix that received salt for events 2-5 in Y4 for 10% leaf compost. This suggests that salt in the inflow may exacerbate phosphate release from compost.

Minimal difference is observed between the compost-iron layered mix that received salt and the mix that did not receive salt (Figure 18). This suggests that salt at ~7.75 g NaCl per L in the inflow is not likely to mobilize phosphate that is captured by iron enhanced sand when the iron-sand layer is below the compost-sand layer. Other media mix designs (e.g., compost, iron, and sand mixed throughout such as in Y1-Y3) were not tested with salt and other salt concentrations were not tested, so other designs and other salt conditions may or may not produce different results.

2.2.2 Nitrate

A similar analysis was conducted for a subset of the events (four events in Y1; four events in Y2, and three events in Y3) to measure nitrate+nitrite as N (hereafter referred to as nitrate) capture or release. The nitrate concentration for Y1 is shown in Figure 19. The average influent concentration was ~0.5 mg/L. The outflow from 100% sand mimicked the inflow nitrate concentration, as expected. The 10% food, 10% leaf, 20% food, 20% leaf, 15% biochar, 5% spent lime, and 5% iron increased the nitrate concentration, indicating an export of nitrate. The food, leaf, biochar and spent lime mixes were expected to release nitrate because these mixes also released phosphate, as described above.

The export from spent lime during the first event was substantial (~9 mg/L), as shown in Figure 19. It's possible that the strong buffering capacity of spent lime may have reacted with the acidic properties of the compost, causing the release during the first event. This phenomenon was not observed during subsequent events. To test whether acid/base interactions were affecting water chemistry within the spent lime media mixes, pH was measured in Y1 and found to not vary significantly between media mixes (see Erickson et al. 2021 and Kramarczuk 2022).

The 5% iron also released nitrate because it contains 20% leaf compost. The nitrate concentration exported from the 5% iron media mix is less than the nitrate concentration from the 20% leaf compost, however, which suggests that not all nitrate is released from compost or that some unknown or unexpected mechanism in the 5% iron media mix can reduce how much nitrate is released. The sphagnum and reed sedge media mixes did not appear to impact nitrate concentration, mimicking both the inflow concentration and the 100% sand concentration.

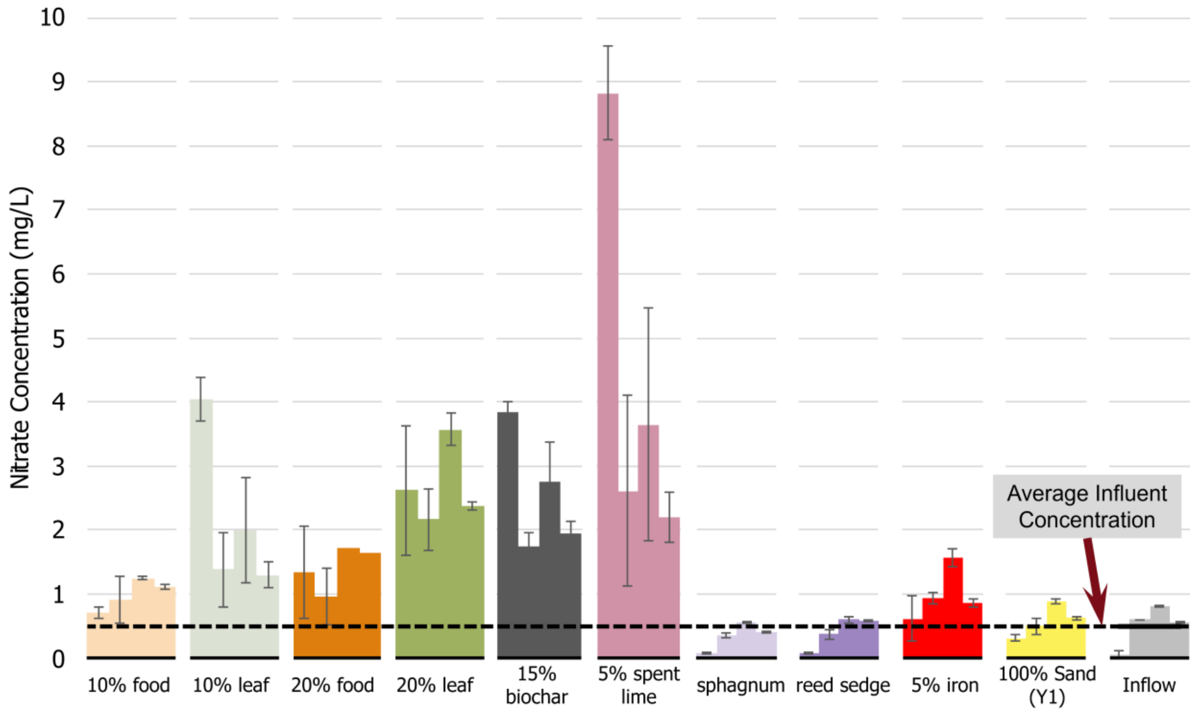

Figure 19: Nitrate + nitrite as N concentration (mg/L) concentration (bars) from mesocosm media mixes (color-coded groups) in Y1.

The nitrate concentration from the mesocosm media mixes in Y2 is shown in Figure 20. The average inflow concentration into the mesocosms was ~0.16 mg/L in Y2. Similar to Y1, the 10% food, 10% leaf, 20% food, 20% leaf, 15% biochar, 5% spent lime, and 5% iron media mixes increased the nitrate concentration from inflow to outflow in Y2, indicating nitrate release. The sphagnum and reed sedge media mixes did not appear to impact nitrate concentration, mimicking both the inflow concentration and the 100% sand concentration.

The nitrate concentration from biochar relative to other mixes (e.g., 20% leaf) in Y2 was less than biochar in Y1 relative to other mixes (Figure 20). It's possibly that the micropore structure of biochar allowed for microbial growth and interstitial anaerobic conditions, creating an opportunity for denitrifying bacteria to grow within the biochar and reduce nitrate release. The 5% iron media mix released nitrate compared to 100% sand and the inflow. More nitrate was released from 5% iron in Y2 relative to other mixes compared to the nitrate release from 5% iron in Y1, relative to other mixes. As iron continues to rust, the redox conditions will continue to change in response, which may affect nitrate transport or formation.

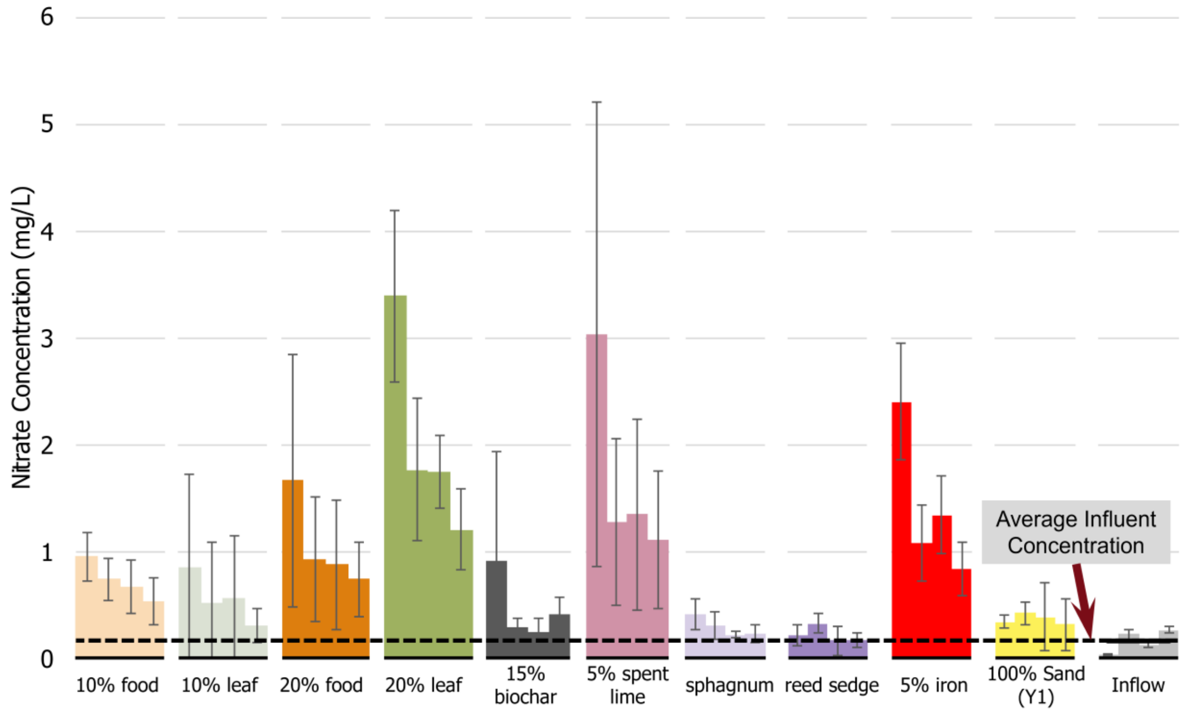

Figure 20: Nitrate concentration (mg/L) concentration (bars) from mesocosm media mixes (color-coded groups) in Y2.

The nitrate concentration from the mesocosm media mixes in Y3 is shown in Figure 21. The average inflow concentration into the mesocosms was ~0.14 mg/L in Y3. Five media mixes from Y1 and Y2 were kept in Y3 (20% leaf, 15% biochar, 5% spent lime, sphagnum, and 100% Sand (Y1)), and five new media mixes were constructed in Y3. Of the mixes that were kept, the trends in nitrate concentration for Y3 were similar to trends observed in Y1 and Y2, except for biochar. In Y1, biochar released nitrate at the same magnitude as 20% leaf compost (Figure 19), but in Y2 biochar released much less nitrate (Figure 20). In Y3, biochar performed similar to Y1 with release at the same magnitude as 20% leaf compost. It's unclear whether the processes described for Y2 were not occurring in Y3, or if Y2 was an abnormal year for other reasons. Another difference in Y3 was that the sphagnum peat media mixes released more nitrate in Y3 compared to the inflow. The effluent nitrate concentration from sphagnum peat was approximately the same as 100% and the inflow in previous years. It's unclear why nitrate released changed in Y3 for sphagnum peat.

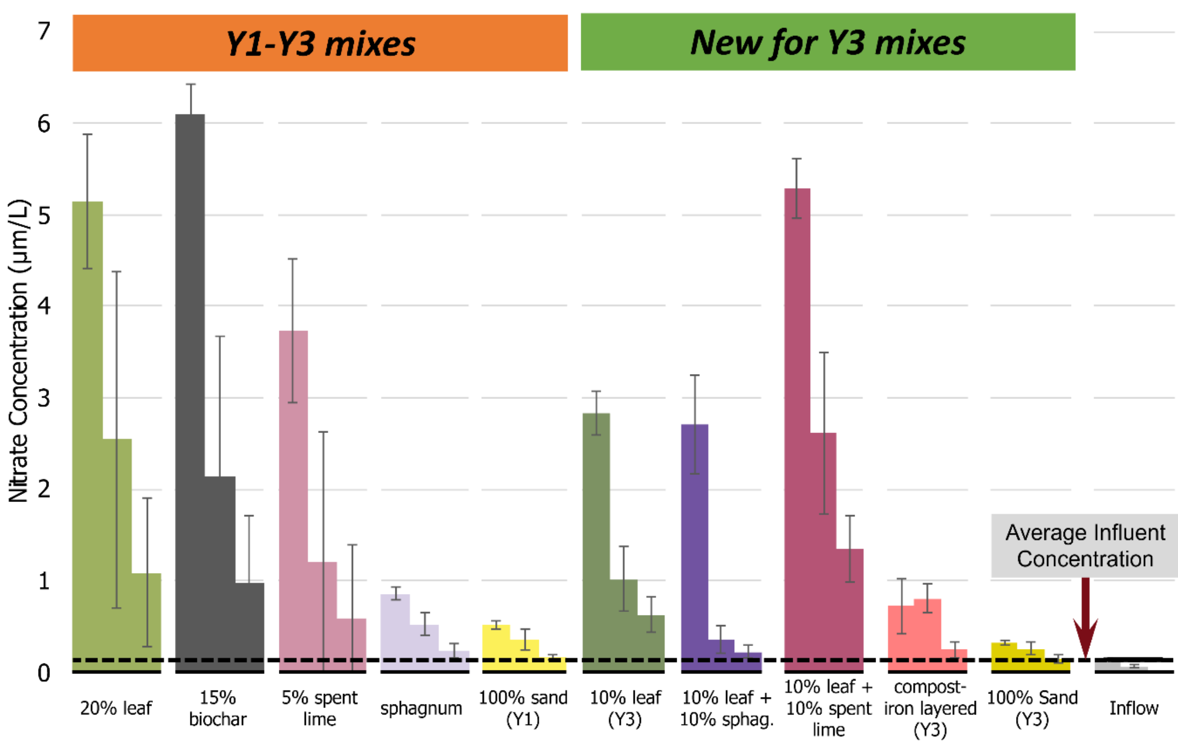

Figure 21: Nitrate concentration (mg/L) concentration (bars) from mesocosm media mixes (color-coded groups) in Y3.

The new 10% leaf media mix added in Y3 performed similar to 10% leaf media mixes from previous years; releasing nitrate. The new mix of 10% leaf compost with 10% sphagnum peat also released nitrate for the first event but mimicked the inflow and 100% sand media mix in subsequent events. It's possible that an initial release of nitrate from the leaf compost produced the high initial nitrate concentration, but the mix of leaf compost and peat reduced the overall release of nitrate.

The new mix of 10% leaf compost with 10% spent lime released the highest nitrate concentration in Y3, of the new media mixes. As was observed in Y1 and Y2, spent lime produces a substantial release of nitrate compared to other mixes. That phenomenon was also observed in the 10% leaf compost with 10% spent lime media mix.

In Y3 a new mix comprising 10% compost mixed with sand in a layer on top of a layer of 5% iron mixed with sand showed less nitrate release than 10% compost, but more nitrate release than 100% sand or the inflow. This is consistent with 5% iron media mixes from Y1 and Y2; nitrate is released from iron enhanced mixes.

2.2.3 Vegetation height and biomass

Vegetation growth over time was measured approximately weekly by counting the number of stems and recorded the maximum and minimum height. Except for mesocosm 1 (20% leaf mixture), all mesocosms had vegetation growth. Vegetation height varied between the media mixes and replicates, and statistical significance was not observed between most media mixes (Erickson et al. 2021 and Kramarczuk 2022).

Visual observation of Figure 22 suggests two groups of media mixes; one in which vegetation grew well and another group which did not.

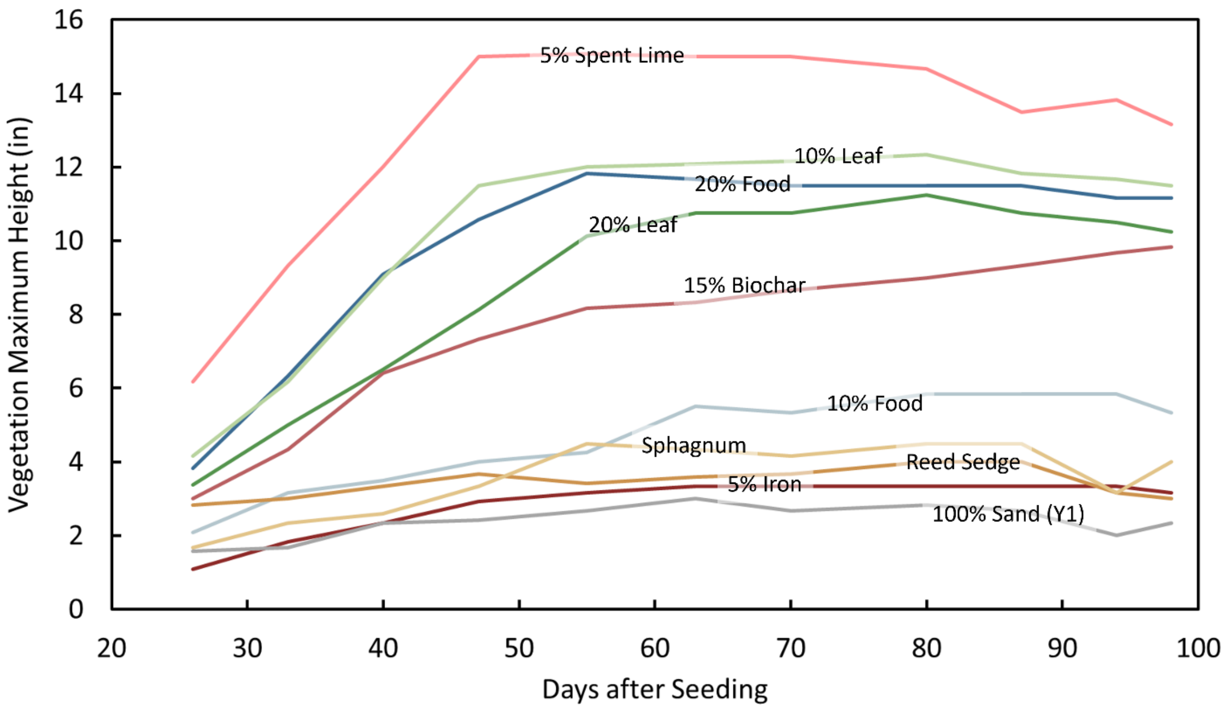

Figure 22: Average (n = 3) maximum height (inches) of switchgrass for each media mix in Y1.

The average maximum height for each media mix increased until approximately 50 days after seeding in Y1 and then continued at approximately peak height until senescence as shown in Figure 22. The 5% spent lime media mix exhibited the largest maximum vegetation height, though vegetation from 10% leaf, 20% food, 20% leaf, and 15% biochar all grew from 25 to 50 days after seeding and peaked at max heights at least twice as tall as when measurements began. As shown in Figure 15 and Figure 19, the 10% food, 10% leaf, 20% food, 20% leaf, 15% biochar, and 5% spent lime media mixes exported phosphate and nitrate in Y1. Thus, there was abundant soluble phosphate and nitrate in the pores of these media mixes to support vegetation growth. The other media mixes (10% food, 20% sphagnum, 20% reed sedge, 5% iron, and 100% sand (Y1) did not grow vegetation as well, however. These mixes also exhibited phosphate capture and minimal impact on nitrate concentration, except for 10% food. The 10% food media mix, however, released the least amount of phosphate and nitrate of the media mixes that released nutrients. It's possible the amount of phosphate and nitrate available in the 10% food media mix was not sufficient to stimulate large vegetation growth.

Vegetation measurements did not begin until ~45 days after seeding in Y2, which was after the vegetation reached its peak of max height, as shown in Figure 23. Despite the lack of measurements during the growth period, there is a clear visual distinction between three groups of vegetation growth performance: 'high' growth (10% leaf, spent lime, biochar, 20% food, and 20% leaf), 'low' growth (reed sedge, iron, 10% food, and sphagnum), and a no growth group (100% sand). In Y2, spent lime performed similar to other media mixes in the high growth group, unlike Y1 in which spent lime was approximately 30% taller than all other media mixes. The media mixes in the low growth group in Y2 are the same media mixes that exhibited minimal vegetation growth in Y1, though the low growth group appeared to grow switchgrass better than 100% sand in Y2. The interrelationships between nutrient release and vegetation growth that were observed in Y1 (i.e., high nutrient release → tall vegetation growth) was also observed in Y2.

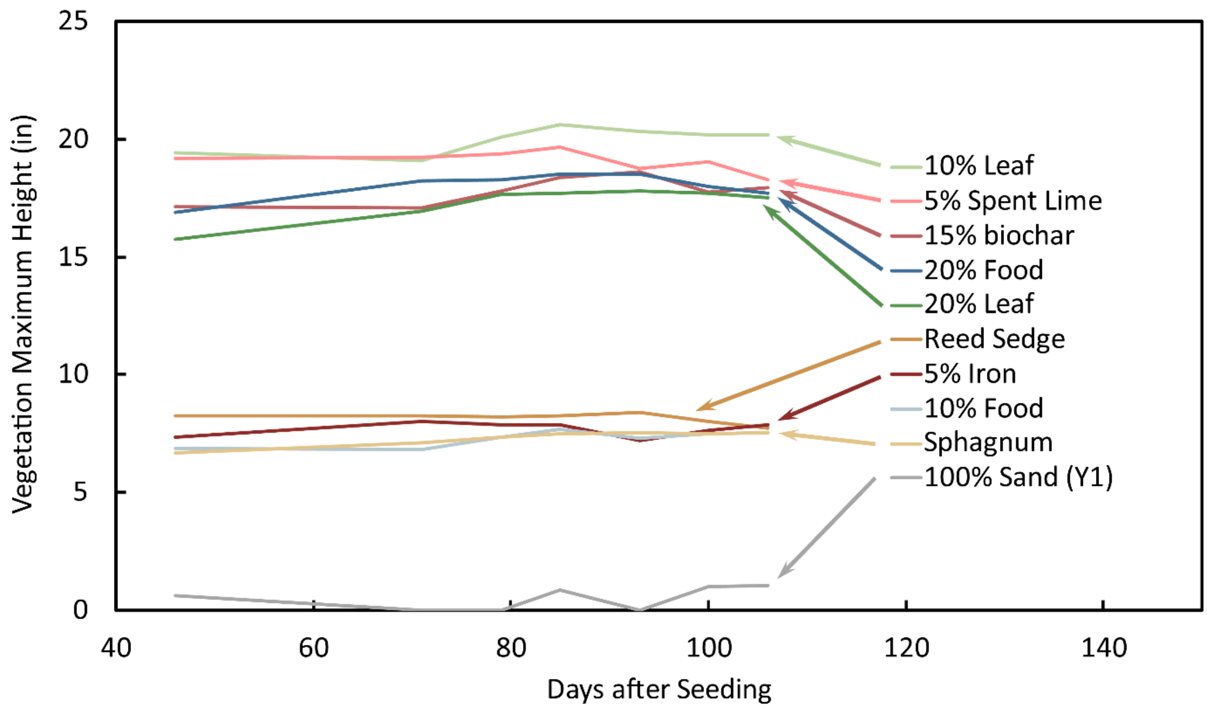

Figure 23: Average maximum height (inches) of switchgrass for each media mix in Y2.

The media mixes were changed in Y3 such that half of the mixes were kept from Y1-Y2 (20% leaf, 15% biochar, 5% spent lime, sphagnum, and 100% Sand (Y1)) and the other half were new mixes (10% leaf, a mix of 10% leaf compost + 10% sphagnum peat, a mix of 10% leaf compost + 10% spent lime, a layered mix of 10% compost over 5% iron, and a new 100% sand mix). The vegetation growth for all mixes in Y3 is shown in Figure 24. The mixes that were kept from Y1-Y2 and into Y3 are shown as solid lines, whereas the new mixes are shown as dashed lines. It's important to note that the solid line mixes (20% leaf, 15% biochar, 5% spent lime, sphagnum, and 100% Sand (Y1)) are media mixes that are growing vegetation in a third season, with new seeding in all three seasons. Thus, the growth in these mixes are not a direct comparison to the dashed line mixes that were added in Y3, and only received seeding in a single season.

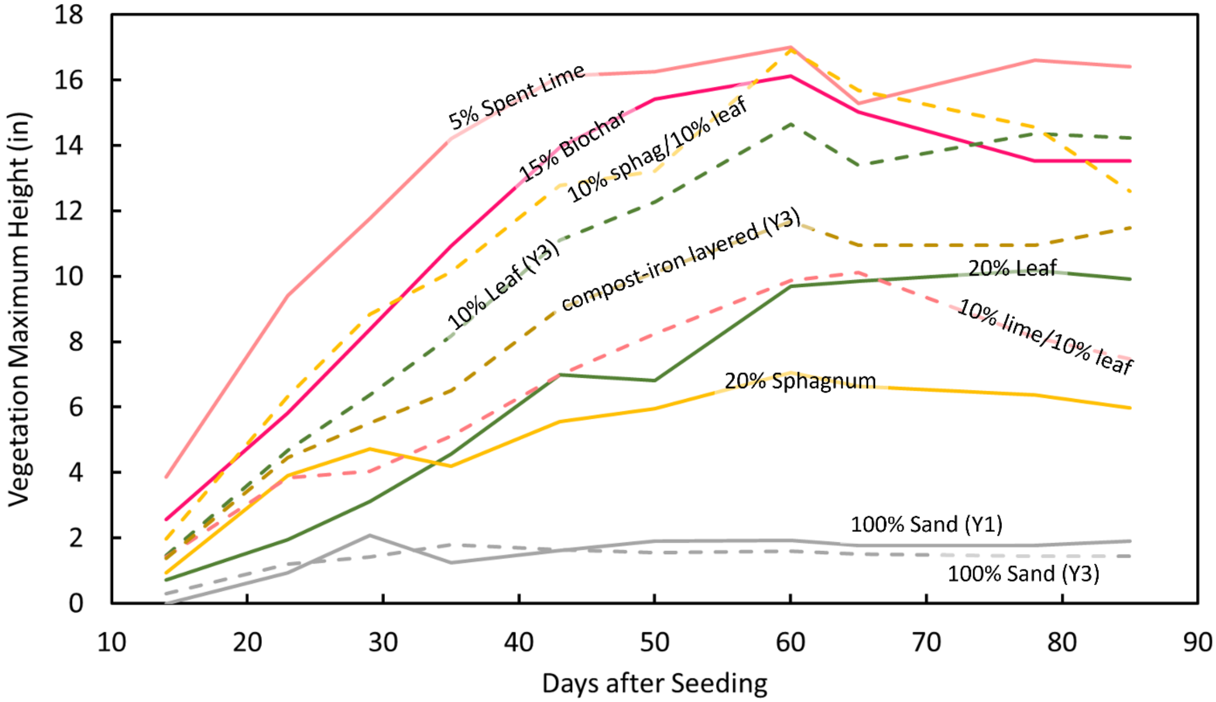

Figure 24: Average maximum height (inches) of switchgrass for each media mix in Y3.

Vegetation in Y3 reached peak maximum height at approximately 60 days after seeding. Of the mixes from Y1-Y3, the spent lime and biochar media mixes grew the tallest vegetation while 20% leaf, sphagnum grew less vegetation, and 100% sand (Y1) grew very little vegetation. These trends were similar to those observed in Y1 and Y2 for these media mixes relative to each other and relative to 100% sand (Y1).

Of the new mixes added in Y3, the 10% leaf + 10% sphagnum mix and 10% leaf (Y3) performed the best, but the compost-iron layered (Y3) and 10% leaf + 10% spent lime performed nearly as well. Only the 100% sand (Y3) grew little vegetation. When compared to mixes in Y1 (Figure 22), the new mixes in Y3 grew taller vegetation than the mixes in the 'low group' and can be considered an improvement in vegetation growth compared to 10% food, 20% sphagnum, 20% reed sedge, 5% iron, and 100% sand (Y1).

Another metric for assessing vegetation growth is above-ground biomass collected at senescence. Biomass measurements vary by orders of magnitude, so the biomass results are plotted on a logarithmic base-10 scale as shown in Figure 25. The yellow shaded box encompasses the new media mixes added in Y3. As described above, the vegetation growth for mixes in Y1-Y3 are not directly comparable to mixes added in Y3 because of the different time scale, seeding conditions, and climate conditions. They are, however, shown on the same figure for conciseness.

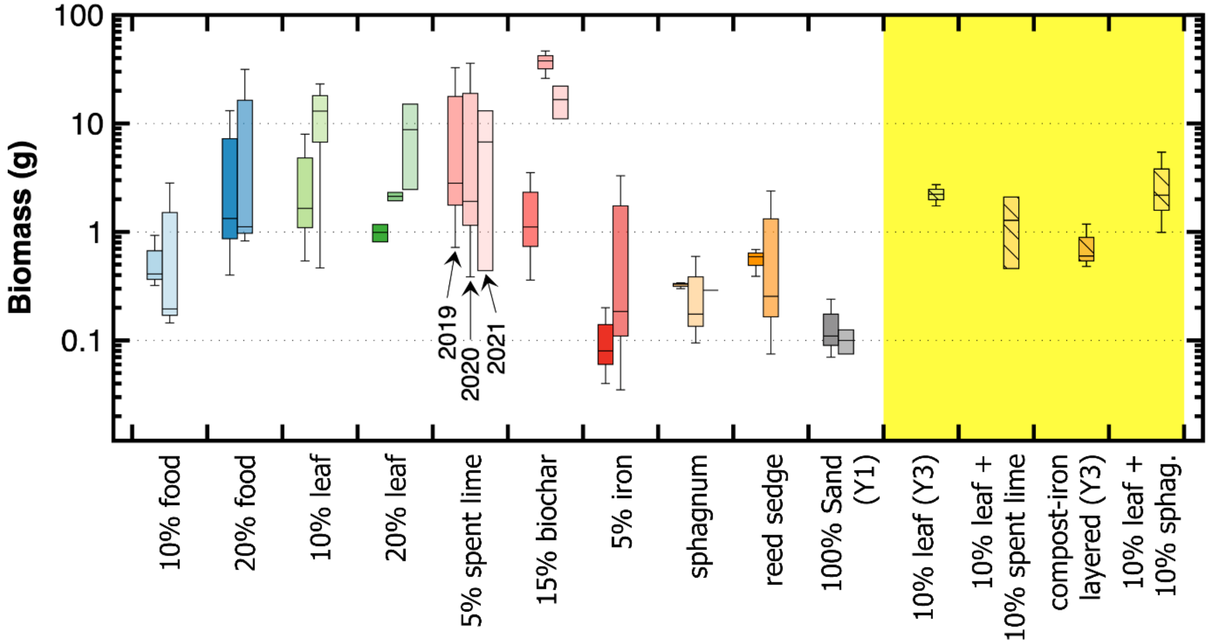

Figure 25: Above-ground biomass (g) collected at senescence for each media mix in Y1, Y2, and Y3. Center line is the median; box boundaries are the interquartile range (IQR); whiskers are the maximum and minimum values.

Similar to vegetation height, several media mixes exhibit substantial vegetation biomass growth of between 1 to more than 10 g; including 20% food, 10% leaf, 20% leaf, spent lime, and biochar. This list of media mixes corresponds well with the 'high' growth group of media mixes that grew tall vegetation in Y1-Y3: 10% leaf, spent lime, biochar, 20% food, and 20% leaf. Thus, these media mixes grew plants that were both tall and exhibited substantial vegetation coverage over the surface area of the mesocosm, as indicated by the biomass. In contrast, the biomass for 10% food, 5% iron, sphagnum, reed sedge, and 100% sand (Y1) was approximately an order of magnitude less than the biomass from the high growth group. This was expected corresponds with the vegetation height data.

For the new media mixes added in Y3, the biomass for the 10% leaf (Y3) and 10% leaf + 10% sphagnum was in the high growth range (1 – 10g) while the biomass for 10% leaf + 10% spent lime and the compost-iron layered (Y3) media mix overlapped the boundary between the low growth range (0.1 – 1 g) and high growth range (1 – 10g). This corresponds well with the vegetation height results.

CHAPTER 3: INDOOR SEED GERMINATION TESTS

3.1 METHODS AND MATERIALS

The outdoor mesocosm experiments (CHAPTER 2:) measured the ability of several media mixes derived from current design standards for 1) releasing or capturing phosphate, and 2) growing vegetation. The results indicate that media mixes that release phosphate also grow plants well. The inverse is also true: media mixes that capture phosphate struggle to grow plants well. A question among stormwater practitioners is this: is there a ratio of compost in the media mix that will release minimal phosphate but be sufficient to grow plants? To answer this question, indoor seed germination tests were developed to measure the capability of several ratios of compost in sand to germinate seeds. While the outdoor mesocosms were a monoculture of switchgrass as a surrogate for native grass seed mixes, the indoor germination tests were developed to measure seed germination of ten different vegetation species so that the results were not limited and biased to a single species.

3.1.1 Testing Protocol

The indoor seed germination tests used ASTM D7322 (ASTM 2017) as a guide, though several modifications were necessary due to differences in the purposes of ASTM D7322 and our intended seed germination test. D7322 is intended to "evaluating the effect of Erosion Control Products (ECPs) on seed germination and vegetation enhancement." We are not evaluating or using erosion control products in this experiment, so any references to ECPs were ignored. Also, D7322 recommends that the size of the containers be 8 inches in diameter and 10 inches in height, while being sub-divided into 2-inch by 2-inch squares on the surface. Our experimental matrix proposes to test approximately 500 different individual squares, so the D7322 methodology required too much space and too many containers. Instead, seedling trays were purchased with 50 individual pots (1 7/8 inches x 1 7/8 inches top area; 1 ¼ inch x 1 ¼ inch bottom area; 2 ¼ inch deep) per tray. D7322 recommends a seeding rate of 0.50 seeds per cm^2 and a seed mix of tall fescue. Because this experiment evaluated seed germination for ten vegetation species (see below) of differing seed size, seeds were added at a rate of 11 seeds per pot in the seeding trays. A seedling light cart was purchased from Johnny Selected Seeds (12 Trays, 384 Watts, https://www.johnnyseeds.com/) to hold and provide light for the germination trays. A small enclosure was constructed of PVC pipe and plastic sheeting to protect the light cart from dust and debris while also containing humidity as much as possible. Temperature within the enclosure throughout the three-week test period varied between 70 – 86°F, light varied between 2900 and 3800 lux (2900 – 3800 lumens / m^2), and relative humidity varied between 35 and 45%. ASTM D7322 recommends a constant temperature of 27 ± 2°C [81 ± 2°F], 45 ± 5 % relative humidity, and Light output = 3300 lm with Color Temperature = 3000 K. D7322 was otherwise followed as closely as possible.

3.1.2 Mixes

A range of compost and sand ratios were tested to determine whether seed germination consistently ceased at a lower limit of compost within the media. The ratios tested were 0% compost with 100% Sand; 3% compost with 97% Sand; 5% compost with 95% Sand; 10% compost with 90% Sand; and 20% compost with 80% Sand. The last two ratios were specifically chosen to correspond with the outdoor mesocosm experiments. Because vegetation experiments are often exhibit large uncertainty in the results, nine replicates for each media mix were prepared.

3.1.3 Vegetation Species

The outdoor mesocosm experiments were limited to a monoculture of switchgrass (*Panicum virgatum*). For the indoor seed germination tests, the technical advisory team was polled for a selection of ten native species that represent the large number of species commonly used in rain gardens in Minnesota. The resulting list of species included: Black-eyed Susan (*Rudbeckia hirta*), Blue Grama (*Bouteloua gracilis*), Brown Fox Sedge (*Carex vulpinoidea*), Canada Wild Rye (*Elymus canadensis*), Creeping Red Fescue (*Festuca rubra*), Partridge Pea (*Chamaecrista fasciculata*), Purple Prairie Clover (*Dalea purpurea*), Sneezeweed (*Helenium autumnale*), Switchgrass (*Panicum virgatum*), Western Yarrow (*Achillea millefolium*). Given that five mix ratios were selected, ten vegetation species plus one unseeded set of pots were used, and nine replicates for each combination were prepared, a total of 495 pots were tested in this 21-day experiment.

3.2 RESULTS AND DISCUSSION

At Day 7, Day 14, and Day 21 after the experiment commenced, the height of the tallest sprout and the number of sprouts were recorded. The sprout height for the five ratios of compost with sand for the ten vegetation species plus the unseeded blank is shown in Figure 26. Two species produced the tallest sprouts after 21 days: Canada Wild Rye and Creeping Red Fescue, for all mix ratios of compost with sand including 0% compost. The sprout height for these species was consistent across all compost mix ratios, and thus not indicative of an optimal minimum mix ratio for compost. This result suggests that Canada Wild Rye and Creeping Red Fescue are resilient species capable of sprouting in low or no compost media mixes.

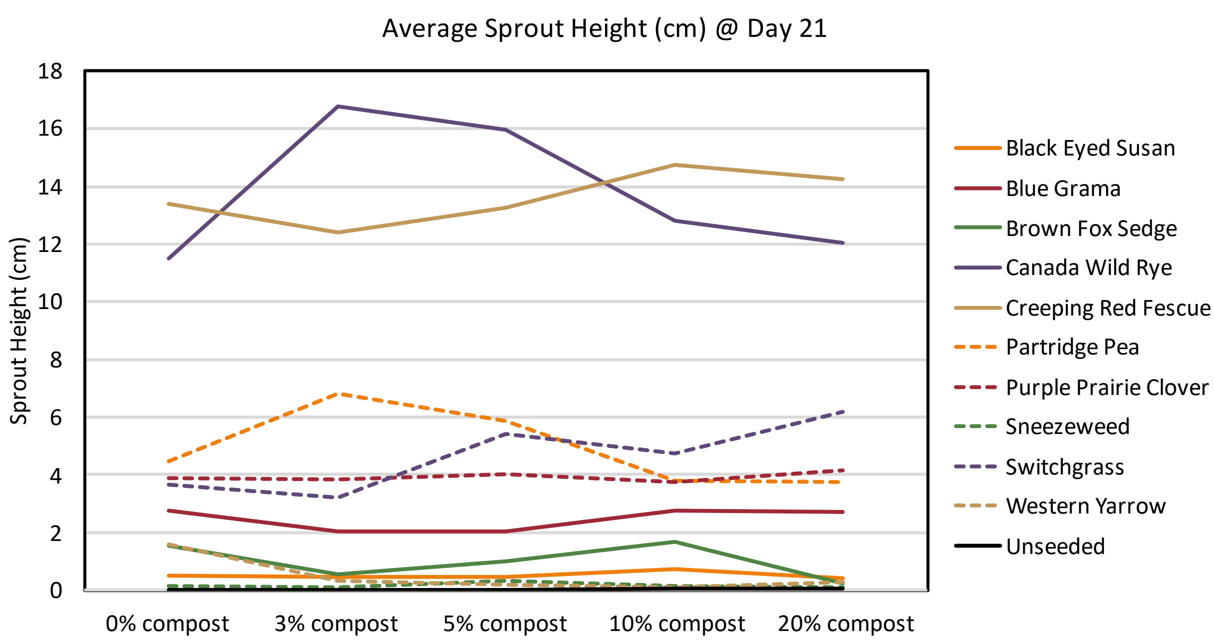

Figure 26: Average (n=9) sprout height (cm) at Day 21 for five ratios of compost with sand and ten vegetation species, plus unseeded.

Four species consistently produced minimal sprout height after 21 days for all mix ratios, including 20% compost: Black Eyed Susan, Brown Fox Sedge, Sneezeweed, and Western Yarrow. This result does not necessarily indicate that these species will grow poorly in these mixes, but could be a result of the experimental conditions, quality of the live seed, or other factors. The remaining species (Blue Grama, Partridge Pea, Purple Prairie Clover, and Switchgrass) increased in sprout height over the 21-day test period (data not shown), but do not show strong correlations between sprout height and compost mix ratio. Thus, they do not provide information useful for selecting an optimal mix ratio of compost and sand. This result suggests that the compost ratio can vary between 0% and 20% and produce approximately the same sprout height within the first 21 days for these species.

The number of sprouts recorded during the 21-day test period is shown in Figure 27. Three species consistently produced a large number of sprouts compared to the other species: Creeping Red Fescue, Purple Prairie Clover, and Black Eyed Susan. Black Eyed Susan produced fewer sprouts than Creeping Red Fescue and Purple Prairie Clover for 5% compost ad 20% compost. The remaining species produced three or fewer sprouts (< 27% of seeds applied) on average throughout the 21-day test for all mix ratios.

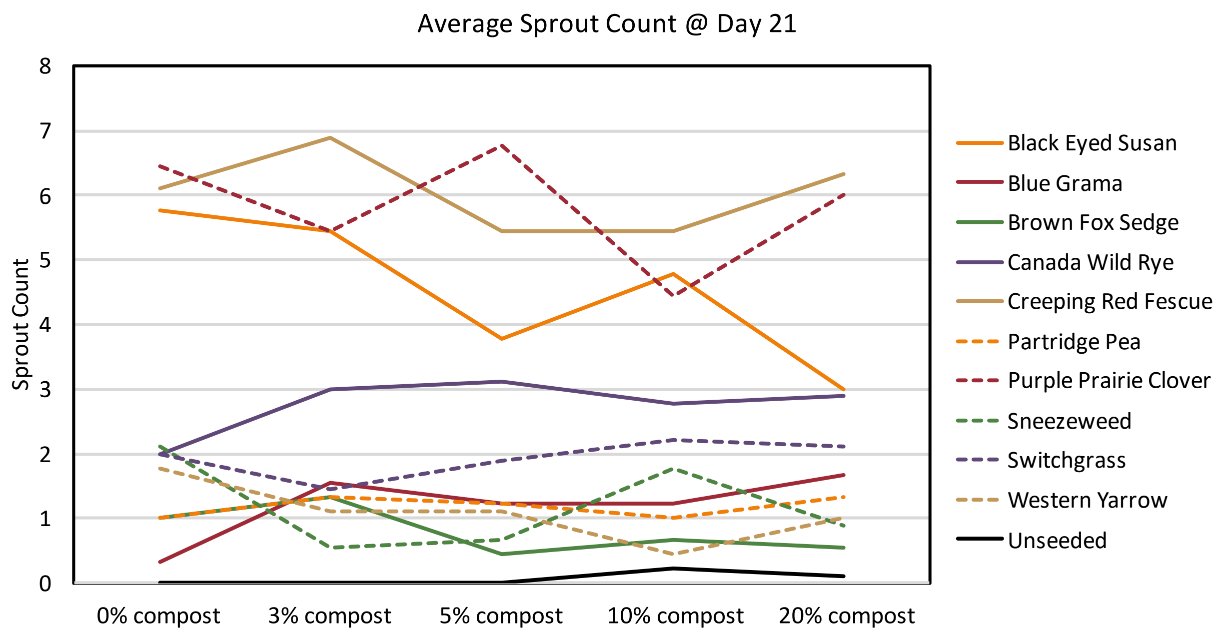

Figure 27: Average (n=9) sprout count at Day 21 for five ratios of compost with sand and ten vegetation species, plus unseeded.

As an estimate of the overall seed germination density, the total number of sprouts was multiplied by the sprout height to determine the 'total height'. This is not an accurate measure of the total vegetation height but provides a relative comparison of germination density. Single tall sprouts could produce a similar total height as several short sprouts. The total height for all species and mix ratios is shown in Figure 28. Creeping Red Fescue produced the largest total height of all species, and the total height was consistent (75 – 90 cm) for all compost mix ratios. This is expected because Creeping Red Fescue produced the tallest sprouts (along with Canada Wild Rye) and the most sprouts (along with Purple Prairie Clover and Black Eyed Susan) for all mix ratios. This further substantiates Creeping Red Fescue as a resilient species that can thrive in low and no compost media mixes.

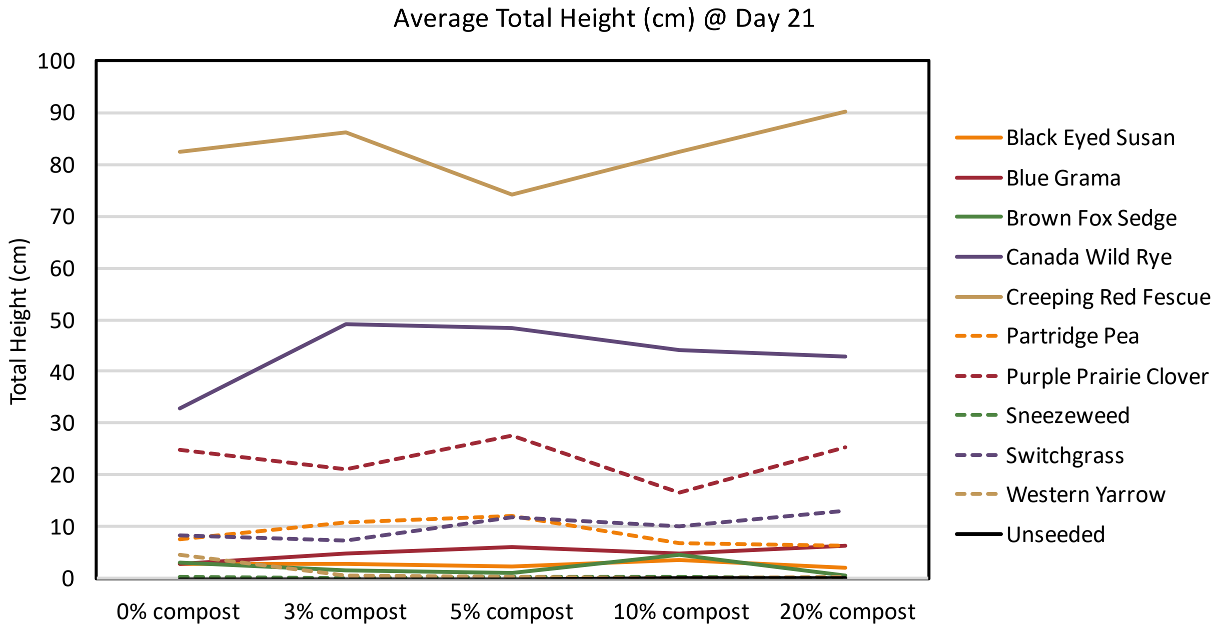

Figure 28: Average (n=9) sprout total height at Day 21 for five ratios of compost with sand and ten vegetation species, plus unseeded.

Canada Wild Rye produced the second largest total height (30 – 50 cm). This is also expected because it was among the tallest sprout height but produced an average number of sprouts between one and three, resulting in a total height less than Creeping Red Fescue. Canada Wild Rye produced a consistent total height of between 40 and 50 cm for all compost mix ratios, except for 0% compost (33 cm). This was the only species that showed any indication that total height could be a function of compost ratio, and only for 0% to 3% compost. Purple Prairie Clover was the species with the third largest total height as a result of its high number of sprouts and average sprout height. The remaining species produced total heights between 0 and 12 cm.

It's important to reiterate that lack of sprout height, number of sprouts, or total height does not necessarily indicate poor performance. This seed germination test was conducted indoors with artificial lights, small scale germination pots, and over a short time period (21-days). This short-term test was devised as a lab-scale discovery experiment, and a more thorough and robust experiment should be conducted to answer questions regarding differences between vegetation species and capability of species to perform in various conditions.

After Day 21, the biomass of each pot was also measured, which is shown in Figure 29. Black Eyed Susan produced the largest biomass for all mix ratios except 20% compost, followed by Purple Prairie Clover and Partridge Pea. The biomass of Black Eyed Susan increased approximately linearly from 0% compost to 5% compost (peak biomass), but decreased for 10% and 20% compost, respectively. A similar trend was observed for Purple Prairie Clover and Partridge Pea, which produced the second and third largest biomass and peaked at 5% compost ratio. This is unexpected and it is unclear why this was observed. At 20% compost, Purple Prairie Clover and Partridge Pea produced more biomass than Black Eyed Susan. The remaining species produced 10mg or less of biomass for all compost mix ratios and did not demonstrate any strong correlations with compost mix ratio.

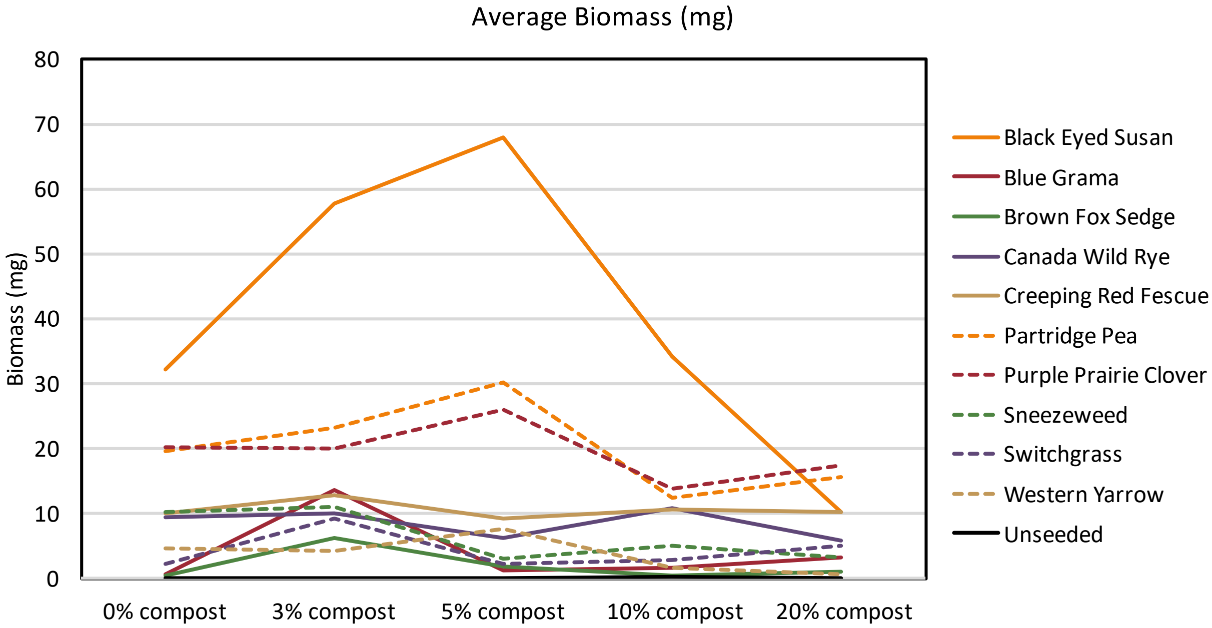

Figure 29: Average (n=9) sprout biomass for five ratios of compost with sand and ten vegetation species, plus unseeded.

Overall, this lab-scale indoor discovery experiment did not indicate that there is a strong correlation between compost ratio and seed germination. In general, a short list of five species produced the tallest sprouts (Canada Wild Rye and Creeping Red Fescue), most sprouts (Creeping Red Fescue, Purple Prairie Clover, and Black Eyed Susan), largest total height (Creeping Red Fescue, Canada Wild Rye, and Purple Prairie Clover), and biomass (Black Eyed Susan, Purple Prairie Clover, and Partridge Pea). A more robust experiment should be developed to more thoroughly compare performance between vegetation species.

CHAPTER 4: CONCLUSIONS

The objectives of this research were to 1) identify which local and sustainable biofiltration media are effective for filtration rate and supporting plant growth and microbial function while not releasing phosphate, and 2) document local sources, simple tests or metrics, and/or design specifications that can be used by practitioners to reliably and repeatably obtain a biofiltration practice that functions as expected.

As described in CHAPTER 1:, batch studies showed that 1) the amount of phosphate released (µg) increases as the amount of compost (g) increases; and 2) the amount of phosphate release varies by site and by compost type (yard vs. food residue). Thus, it's important to understand the potential for an organic material (including topsoil) to release phosphate prior to installation within a stormwater control measure; especially biofiltration practices with underdrains. The batch studies also indicated that results from common media assessment techniques such as Solvita, Olsen P, Bray P, Mehlich-III P, and ICP-OES P did not correlate with phosphate release in batch studies. Thus, these methods may not be a reliable way to predict phosphate release potential in biofiltration media. It was hypothesized that simple batch tests could be performed in the field using water bottles and field test kits for phosphate concentration. Unfortunately, the field test kits could not measure phosphate in samples collected from batch tests due to color interference and turbidity from the media. Thus, other means are necessary for consistently and reliably evaluating phosphate release potential from biofiltration media.

As described in CHAPTER 2:, outdoor mesocosm experiments showed that phosphate is released from biofiltration media mixes commonly designed and specified in Minnesota, including food or leaf compost at ratios of 10% and 20%. Mixes amended with biochar or spent lime reduced the amount of phosphate released from compost-based mixes, but a net export of phosphate (outflow > inflow) was observed. Replacing compost with sphagnum or reed sedge peat resulted in a net phosphate capture (inflow > outflow), which was unexpected and unexplained. Compost-based media amended with 5% iron filings also resulted in net phosphate capture. These trends continued for three rainy seasons of simulated events. Nitrate was also exported from compost-based mixes, including those amended with biochar, spent lime, or iron. Vegetation growth was inversely related to phosphate capture; i.e., mixes that released phosphate grew tall plants with more biomass and mixes that captured phosphate grew small plants and little biomass. Thus, the first three seasons did not determine a mix that could grow plants while also mitigating phosphate release.

In Year 3 of the mesocosm experiments, new mixes were installed to test hypotheses of potential optimal mixes. These mixes included combining leaf compost with sphagnum peat, reducing compost while increasing spent lime, and a layered mix of compost with sand (growth layer) on top of iron with sand (treatment layer). The results indicated that only the compost-iron layered mix could capture phosphate, though it also released nitrate. The compost-iron layered media mix grew plants better than expected, but not as well as 10% leaf compost alone or other mixes in the 'high growth' group.

In Y4, the mesocosm experiments were used to test the impact of road salt on phosphate transport in biofiltration media mixes. Salt at ~7750 mg/L NaCl in the inflow resulted in a nearly twofold increase in phosphate concentration that was exported from 10% leaf compost but did not appear to affect the phosphate capture by the compost-iron layered media mix. Thus, road salt may exacerbate phosphate release from rain gardens and biofiltration practices that are not amended with iron.

As described in CHAPTER 3:, short-term seed germination tests were used to determine whether seed germination could be correlated to compost ratio in the media mix. Overall, this lab-scale indoor discovery experiment did not indicate that there is a strong correlation between compost ratio and seed germination. In general, a short list of five species produced the tallest sprouts (Canada Wild Rye and Creeping Red Fescue), most sprouts (Creeping Red Fescue, Purple Prairie Clover, and Black Eyed Susan), largest total height (Creeping Red Fescue, Canada Wild Rye, and Purple Prairie Clover), and biomass (Black Eyed Susan, Purple Prairie Clover, and Partridge Pea).

The overall conclusions from this study are as follows:

1. Organic materials vary substantially based on supplier, source material and other factors not tested during this research. The results from this study cannot be used to predict the performance of compost, peat, or other organic material due to this variability.
2. Simple tests and metrics do not appear to reliably predict the potential for phosphate release from organic materials.
3. A lab-scale indoor discovery experiment found no correlation between the ratio of compost (0% to 20%) in a media mix and the germination of ten different vegetation species commonly used in Minnesota rain gardens.
4. For the media and materials tested in the mesocosm experiments:
    a. Food-based and leaf-based compost released phosphate and nitrate while also growing switchgrass.
    b. Replacing compost with sphagnum or reed sedge peat resulted in phosphate capture and poor vegetation growth
    c. Leaf-based compost media mixes amended with biochar or spent lime reduced phosphate export, but not below inflow concentration
    d. Leaf-based compost media mixes amended with iron resulted in phosphate capture and poor vegetation growth
    e. Simulated road salt in runoff increased the concentration of phosphate released from leaf-based compost

CHAPTER 5: APPENDICES

5.1 LITERATURE REVIEW

The literature review in this section was originally published in Erickson, AJ, Kozarek, JL, Kramarczuk, KA, and Lewis, L. (2021). "Biofiltration Media Optimization – Phase 1 Final Report." Project Report No. 593, St. Anthony Falls Laboratory, University of Minnesota, Minneapolis, MN. January 2021. An additional literature on this topic appears in Kramarczuk, K.A. (2022). Optimizing Biofiltration Media for the Capture of Phosphate and the Support of Vegetation Growth. Master's Thesis. University of Minnesota. June 2022. *https://hdl.handle.net/11299/241550*

Urban stormwater can have extremely detrimental effects on the environment, and biofiltration is quickly emerging as a way to address these negative effects (Trowsdale et al. 2011). Urban stormwater runoff has been shown to contribute to eutrophication and cause harmful levels of phosphorus in water (Erickson et al., 2013; Jay et al. 2017; Li & Davis, 2016). Harmful levels of phosphorus can originate from fertilizers, automobile exhaust, living and decaying plants, animal remains, and detergents (USEPA, 1999) Limiting phosphorus (P) leaching, focusing on the removal of P, and a focus on media mix design has become a major theme within recent research.

Bioretention systems aid in the improvement of water quality through evapotranspiration, media filtration, adsorption, biotransformation and other natural processes (Davis et al., 2006). They mimic natural ecological systems and thus have great potential in sustaining urban environments. The efficiency of these systems depends on media mixes, infiltration rates and vegetation. Media and vegetation vary regionally and thus a blend and design specific to Minnesota is crucial for optimal bioretention system performance.

5.1.1 Phosphorus Capture and Release

A main concern regarding the design of these bioretention systems is the leaching of phosphorus and how these systems can be designed to treat large storm events or floods. Various models regarding Phosphorus (P) have begun to emerge to aid in the design and implementation of these systems.

5.1.1.1 Column Experiments

A large-scale 125-column study undertaken in Melbourne Australia by (Bratieres et al., 2008) focused on developing the optimal design to remove sediment, P and N. A major component of their design was selecting the correct plant species. The columns were dosed with semi-natural stormwater twice a week and water samples were collected from the inflow and the outflow of the columns. TP removal was >77%. A large portion of its removal was assumed to be attributed to the filtration process because most of the phosphate was in particulate form in the inflow. *Carex appressa* proved most effective in removing P and N perhaps due to its extensive root system. Although TP removal was shown to be efficient in non-vegetated columns, this study concluded that vegetation is of great importance when it comes to the efficiency of these systems, but the species must be selected carefully and organic matter should be limited to decrease the potential for leaching.

In a batch and column study by Hsieh et al. (2007) the effectiveness of bioretention systems was examined. The media involved two types of sand, three types of soil, and a mulch. The batch experiment was carried out to determine the short-term sorption capacity of P of the various media. A continuous column test involving three columns with different media compositions was completed to determine P uptake. In addition, a repetitive column test involving two columns was carried out to investigate P removal and accumulation over periods of multiple loading (80-120 days). The first column (RP1) was designed to have a media with low hydraulic conductivity over a media with high conductivity. The second column (RP2) was the opposite or a media with high conductivity over a layer of media with low conductivity. The media involved in RP1 consisted of a top mulch layer, a middle porous soil layer and a bottom sand layer. RP2 consisted of a mixed top layer (mulch, soil and sand), a middle layer or sand and a bottom layer of soil. Runoff was pumped into the columns from the top and a sample of the effluent was collected every hour for a period of 6 hours. Results from the short-term P sorption column test showed sorption capacities were higher for the three soils than either of the sands, while mulch was the lowest. The repetitive columns tests showed a TP removal efficiency of 47 to 68% for RP1 and RP2 showed almost complete removal of TP within the first 7 repetitions. However, this efficiency showed a steady decrease and by the 14th repetition removal was only 56%. The less permeable bottom layer in RP2 allowed more contact time between DP and media and thus was more efficient in P removal overall. However, it is still recommended to include a bottom fine sand layer results in the most efficient P removal and also prevents leaching and clogging of the bioretention system.

In another study that took place in Australia, the hydraulic and pollutant removal performance of fine soils within the filter media were measured in a laboratory setting (Hatt et al., 2008). Six different media types were studied including fine sand, sandy loam, 80% sandy loam, 20% Hydrocell (a synthetic, commercially available soil ameliorant); 80% sandy loam, 10% vermiculite, 10% perlite; 80% sandy loam, 10% compost, 10% mulch; and 60% sandy loam, 20% compost, 20%mulch on a charcoal drainage layer. In terms of pollutant removal efficiency P was leached across all soil-based filters. Significant accumulation of P within the top 20cm of the filter was observed therefore it was concluded that these filters were able to capture P and it may have been the native materials that leached.

5.1.1.2 Mesocosm Experiments

Some studies have focused on vegetation as a major influencer regarding the presence of both P and N within the media and the effluent. In a study by Henderson et al., (2007) six mesocosm were built with a drainage port and tap. Three different media types were tested; gravel, fine sand and sandy loam each in a vegetated and non-vegetated system. The vegetated treatments contained 5 species of plants. Mesocosms were allowed to establish for a period of 12 months, thus this was a more mature biofiltration study. Two separate studies took place, one a dosing the other a flushing. The mesocosms were dosed with approximately 240L of synthetic stormwater and 22 samples of effluent were collected. In the flushing experiment each mesocosm was dosed with 108L of synthetic stormwater and left for 7 days. They were then irrigated with tap water and effluent was collected hourly for a period of 8 hours. Results from the dosing experiment showed that vegetated mesocosms and the non-vegetated sand mesocosm removed almost all of the P from the synthetic stormwater. Results from the flushing experiment showed very little P and TP was leached from the vegetated and non-vegetated sand mesocosms. This study showed that vegetated mesocosms were much more efficient at removing N and P from stormwater than the non-vegetated mesocosms. Plants flourished the most in a sand or sandy loam mixture and did not need any addition of organic material such as compost which limits the potential of leaching from the media.

In an extensive study by (Davis et al., 2006), bioretention systems were tested for their removal of N and P specifically. Two boxes were constructed, one large (305cm long by 152cm wide with a depth of 91cm) one small (107cm long by 76cm wide with a depth of 61cm), and two PVC pipes at different depths were installed in the small box and three pipes at various depths were installed in the large. Each box was filled with sandy loam soil and had a top layer of 2.5cm mulch. Six small creeping juniper plants were installed in the small box and 12 small and 12 large creeping junipers were installed in the large. The boxes were designed to contain both a bottom port and an upper port, the bottom port remained open. The boxes were treated with synthetic stormwater at a rate of 4.1cm/hr for 6 hours. Influent and effluent samples were taken. In the field, two sites were examined: one containing sandy loam and a mulch top layer as well as grasses the other containing construction sand, leaf mulch and topsoil with some grasses, bushes and small trees. Grab samples were collected every 25-30 minutes. Results from the box studies showed that for the lower duration/ lower intensity treatment, TP removal increased with depth, 77 to 87% at the bottom of the boxes or a P reduction of 0.06 to 0.1 mg/L. In the treatment that simulated a storm the boxes received 8.1cm/L of synthetic stormwater for 12 hours which caused an increased infiltration rate in both boxes. From the bottom port effluent TP removal was around 70% and were not found to be affected by the higher hydraulic loading. The TP removal at both field sites was 65 +/- 8% at the first and 87 +/- 2% at the second and effluent concentrations were just above 0.1mg/L. This showed excellent P removal capability. It was also noted that design and management of vegetation may play an important role in nutrient removal of both N and P. Maintenance of vegetation is also crucial as any plant matter left to decay will result in the release of assimilated nutrients.

Aside from selecting vegetation and bioretention soil mixes, another area of focus is how to structure the media layers. In a study based in China by Yang et al. (2020) three different lab scale bioretention units were built. The first had a 200 mm drainage layer, a 100 mm transition layer, and a 500 mm filter layer. The second had a 200 mm drainage layer, and a 500 mm filter layer and no transition layer and the third a 100 mm transition layer, and a 500 mm filter layer, and a 50 mm thick gravel underneath the transition layer. Effluent was collected in a plastic bucket. The columns were treated with synthetic stormwater based on measurement from the nearby city Kunshan with varying rainfall durations. Using synthetic stormwater, TP removal rate was 68%. Overall, this study showed the importance of all three of these layers and in the treatment of runoff. Overall TP removal with all three layers was 86.0%, with no transition later it was 85.4%, and with no drainage layer it was 71.8%. This research suggested that a design including both a transition and drainage layer provides for better runoff control and nutrient removal.

5.1.1.3 Field Studies

Because bioretention systems are a newly emerging practice, field studies are limited. Following this laboratory study, Hatt et al. (2009) examined three different biofiltration sites in the field. At the first site, flow rate was measured and samples were collected to measure water quality for 14 storm events. At the second site, four storm events were simulated using semi synthetic stormwater and effluent samples were collected. At the third site, auto samplers collected time-weighted water quality samples. The first site showed effluent P concentrations were higher than the influent and increased with flow rate, most likely due to leaching of DP. The second site showed substantial reductions in TP and correlated with flow. The third site pollutant concentrations remained fairly constant. Overall, the three sites show significant reduction in TSS and heavy metal removal, but nutrient removal was much more variable. Use of a filter media with low organic content is recommended to prevent significant leaching of P and the influence of flow rate on effluent pollutant concentrations must be considered when designing these systems. The data from this study suggested that higher infiltration rates may lead to higher effluent concentrations of particulates and their pollutants.

In one study in North Carolina by Hunt et al., (2006), an auto sampler was used to collect the effluent from the underdrain. It was discovered that outflow concentration of nutrients was higher than the inflow concentration indicating the media was not effective in nutrient capture. TP removal rates varied from 65% removal to a 240% increase which was most likely due to the type of media in the cell and its level of P saturation. The soil from all the varying cells were analyzed using the Mehlich-3 methodology and provided a P-index which is an indicator of a soil's ability to adsorb or release P. The cells that had a lower P-index showed less TP in the outflow. The P-index measurement can help determine which media to include in sites that are more vulnerable to P pollution.

5.1.1.4 Models

In a paper by Roy-Poirier et al. (2010), the objective was to identify and characterize the bioretention P cycling processes with a focus on a P transport model. After reviewing several previous numerical models, it was determined that there remains a need for a simple numerical equation to represent the rate of particulate phosphorus dissolution and soluble phosphorus precipitation. None of the models reviewed were found to be applicable to modeling the bioretention system. The authors concluded that a new model was deemed crucial in order to predict the amount of P that would be removed from a proposed design of a bioretention system.

Li & Davis (2016) showed that the fate of phosphorus, or the concentration in the effluent, can be predicted by flow, volume and run time and varies significantly with influent P concentration. It can be used to describe both short- and long-term P removal conditions. Data from previous bioretention studies was used in order to devise the model. During both short-term and long-term studies, the effluent P concentration, C_e, is controlled by the equilibrium concentration, C_{eq}. During the event-term, variation of C_{eq} is influenced by dry duration time and the composition of the media. Longer dry time and weaker media, in terms of adsorption, will lead to larger variance of media C_{eq}. C_{eq} varies less in media with Al and Fe than in un-modified BSM. During the event and short-term studies, the concentration relationship is $C_{eq} > C_e > C_0$ for high-P media and is $C_{eq} < C_e < C_0$ for low-P media. In the long-term studies, the overall concentration relationship approaches C_{eq} (C_{eq^*}) $\approx C_e \approx C_0$. Under natural conditions, C_{eq} (C_{eq^*}) will increase or decrease slowly and approach C_0. This process can take a long time, especially if the BSM contains Al and Fe.

In a study by Jay et al. (2017) the Phosphorus Saturation Ratio (PSR) and the Phosphorus Saturation Index (PSI) were tested across a variety of BSMs. The PSI and PSR were calculated using the formula: P/(Al+Fe), where Al = Aluminum and Fe = Iron. The difference being for PSR, P, Fe, and Al represent the Mehlich-3 extractable molar concentration of each element (Maguire & Sims, 2002). Columns were constructed and fourteen different Bioretention soil medias (BSMs) were tested with four duplicates of each. The BSMs consisted of high Fe biosolids, composts from two different feedstocks (yard waste and food scrap), Water Treatment Residual (WTR), oyster shells, soil and sawdust. All the various mixtures included sand as a component of the BSM. The columns were treated with synthetic stormwater at various volumes that reflected a bioretention system designed to collect 90% of runoff. Leachate from the columns was collected during the event and 4 hours after. Highest P concentrations in the leachate came from the biosolids and yard compost. The addition of WTR to BSMs containing compost resulted in a significant reduction in P of the effluent. High Fe biosolids and sawdust also showed a significant reduction in P but overtime the decline of P in the effluent was not consistent. Both the calculated PSR and PSI were compared to the results via testing regressions. PSR proved to be the best predictor for total and dissolved P with an R^2 of 0.733 and 0.681, respectively. The PSR could potentially be used as a predictor across different regions and with different BSM ingredients.

5.1.1.5 Phosphorus Summary

As research continues regarding the design of bioretention systems, P leaching is of great concern and varies greatly based on the BSM. The development and use of P transport models could potentially aid in the design and implementation of these systems and aid in the predictability of how certain BSMs will perform in various regions. Removal of P appears to increase with the addition of certain vegetation while adding organic matter to the BSM often contributes to P leaching. In addition, infiltration rate was also shown to influence the concentration of P in the effluent.

5.1.2 Bioretention Mixes with Compost

Compost is commonly used in bioretention media mixes because it can retain moisture to support the vegetation. In addition, it has been shown to remove pollutants. However, studies have suggested that compost is not always necessary to establish plant growth or a minimal amount, if any, should be considered in the BSM design to reduce P leaching.

With a focus on the importance of saturation and nutrient leaching potential Hurley et al. (2017) showed that saturation duration did have an effect of P leaching. Compost was collected from three different locations within Vermont in addition to a thermophilic sample and vermicompost sample. These samples were compared to two engineered bioretention mixes; one containing 40% compost the other with 4%. Four different saturation times were tested: 10 minutes, 1 day, 5 days and 10 days. A modified version of the U.S. Geological survey leach test (Hageman et al., 2007) was set up to obtain measurement and samples were treated with deionized water. P levels were lowest in the engineered mixes for all saturation durations. The P concentration increased with time of saturation for all compost samples. The engineered BSM containing 40% compost showed significantly more P leaching than the BSM containing 4% and pure compost samples showed significantly higher P leaching than either of the engineered BSM. It was concluded that compost should be avoided in environments with high saturation potential or if it is necessary a low P compost should be used to limit the P leaching potential.

In the Seattle Tacoma region of Washington State, 6-month and 24-month aged compost consisting of 80% yard waste and 20% food waste, was irrigated to simulate a storm event passing through a bioretention cell (Mullane et al., 2015). A total of six columns were constructed and treated with an irrigation rate of 33.5 mm/day based on a 6-month 24-hour storm for that region. The beginning TP measurement for the 6-month compost was 2.9 ± 0.6 g/kg whereas the final was 2.5 ± 0.3 g/kg. The 24-month old compost had an initial TP measurement of 2.7 ± 0.1 g/kg and a final of 2.7 ± 0.4 g/kg. This suggested that P leaching concentrations from mature compost decrease with each individual rainstorm. With initial leaching apparent, it was suggested that bioretention systems containing compost have restricted outflow during the first several storms.

In a bioretention study by Shrestha et al. (2020) one cell was built without compost or vegetation, the second was planted with just one plant species and no compost and the third consisted of a low P compost and vegetation. All three of the cells showed TP and SRP reductions. Average influent TP concentrations were higher compared to effluent concentrations. Cell 1 had a 95.6% TP reduction from three events; cell 2 had a 94.2% TP reduction from seven events, and cell 3 had a 92.6% TP reduction from six events. Effluent average SRP in Cell 1 showed a 94.8% reduction, cell 2 had a 96.1% SRP reduction from eight outflow events, and cell 3 had a 94.1% SRP reduction from six outflow events. Slightly higher effluent concentrations from the cell containing low P compost suggest there was some leaching. The use of compost appeared to have short term effects aiding plants in establishment but may not be a necessary media to add to these systems because they can attribute to P leaching long term. Vegetation did not appear to have any effect on the removal of pollutants but may be more beneficial once the plants are more established.

Another study by Liu et al. (2014) examined three different media blends specifically for N and P removal. Those blends included Terrasolve, Biofilter and a Virginia Tech (VT) mix. The VT and Terrasolve mix included WTR and Biofilter and VT mix included yard waste compost. Additionally, the Terrasolve included a mixture of coir and peat. Columns with vegetation were not as efficient in nutrient removal. The study also had a focus on hydraulic retention time. An increased retention time resulted in greater P removal across all media. Terrasolve proved to be the most efficient in removing P followed by the VT mix and finally the Biofilter. It was the addition of WTR that aided in P removal as was discovered in other studies (Brown et al. 2010; Lucas & Greenway, 2011)

Logsdon & Sauer (2016) compared a mixture of cow manure and yard clippings, a fine loamy mixture and a 50% sand, 26% silt, and 24% clay media within columns. These mixtures were additionally compared to three treatments that had soil and two that did not. The treatments containing soil had a mixture of 1/3 compost,1/3 sand and 1/3 soil. The treatments without compost contained 20% compost and 80% sand. Columns with soil showed significantly lower levels of TP than columns without soil and P that was leached from compost was not sorbed by the sand. It was concluded that compost derived from manure should not be used in these systems and that other forms of compost should be added in small amounts as plant growth may result from organic material already present within the media.

Compost has shown some effectiveness in the uptake of heavy metals such as zinc. In a column and batch study conducted in Australia it was effective in reducing the amount of zinc in the effluent until the compost reached its metal sorption capacity (Al-Mashaqbeh & McLaughlan, 2012). Results indicated that when compost particles were greater than 1.18mm limited uptake of Zinc occurred suggesting that particle size is something to consider in the design of these systems when heavy metal uptake is to be addressed. In another column experiment by Lim et al. (2015) compost showed a removal efficiency of more than 90% on the heavy metals Cu, Zn, Pb and Cd.

In a batch and column study that involved three different types of compost all were shown to be effective in capturing cadmium and zinc (Paus et al., 2014a). Compost samples were collected from two different locations in Minnesota and one in Texas. Ten continuous flow columns were constructed and treated with synthetic stormwater and flow rate was monitored daily. Cu was effectively removed from all columns and total Cu uptake increased with the percentage of compost in the media. It was determined that pollutant breakthrough is not of concern regarding compost because dissolved metals were shown to be removed through sorption. The columns that contained 30% and 50% compost showed a substantial release in both P and dissolved P. P leaching potential was addressed by carrying out batch experiments involving 0.1-1.0g of compost and 230mL deionized water. The batch experiments showed P leaching; one of the samples collected from Minnesota showed 203 ± 24 mg P per kg compost. Thus, the significant P leaching was of concern.

5.1.2.1 Compost Summary

Overall, the use of compost needs further study because while it has shown potential in removal of heavy metals such as copper (Chahala et al., 2016; Silvertooth & Nason, 2014, not discussed above). Research has shown it significantly leaches phosphorus (Hurley et al., 2017, Paus et al., 2014b). It is recommended that if compost is included in the media mix then the design should include two layers. One layer should address toxic metal and pollution retention and another with sand containing Fe to address Phosphorus removal (Paus et al., 2014c). In addition, it is not yet deemed essential in the growth and prosperity of vegetation.

5.1.3 Bioretention Media Amendments

Certain studies have focused on using amendments and enhancements to aid the bioretention media mixture in its ability to capture phosphorus from source stormwater or any P leached from organic materials such as compost.

5.1.3.1 Water Treatment Residuals

Water treatment residuals (WTR) have been a popular cost-effective amendment and have shown to be efficient in capturing phosphorus. An aluminum heavy WTR was used in both a batch and column study by (O'Neill & Davis, 2012a, 2012b). Two large gravity controlled vegetated columns were built. The base bioretention soil media (BSM) was developed in their earlier batch and mini column study and consisted of 77% sand, 14% silt and 8% clay. One column consisted of a mixture of 69% BSM, 5% WTR, 22% additional sand, and 3% hardwood bark mulch and the control without the WTR was 74% BSM, 22% additional sand, and 3% hardwood bark mulch. Synthetic storm water was used as based on previous studies (US EPA 1983, Bratieres et al., 2008) and columns were treated with 182 mL min^{-1} for a continuous 6-hr period. Overall, the column with WTR had an average adsorption of 3.18mg P kg^{-1}. The column without WTR showed an export of P at 2.38mg P kg^{-1}. Therefore, just 5% WTR in the media is capable of removing sufficient P from stormwater.

In an amendment study by Li et al. (2018) 12 columns were built and modified with different fillers. The BSM was a mix of 30% soil, 65% sand, and 5% wood chips. Twelve different columns were built consisting of filler layers containing; soil, 30% planting soil 70% sand, 30% planting soil, 65% sand and 5% wood chips, BSM + 10% WTR, BSM + 10% green zeolite, BSM + 10% medical stone, BSM + 10% fly ash, BSM + 5 % vermiculite, BSM + 5% peat soil, BSM + 5% coconut chaff, BMS + 5% medical stone: peat soil 1:1 and BSM + 5% green zeolite: peat soil 1:1. The analysis focused on TP and SRP and results showed that BSM + 10% WTR had the best median TP removal of 96.80% and the best average TP removal of 97.13%. Overall WTR was deemed the most efficient in P removal. In another mini column study (Zhang et al., 2018) using the same BSM, static isothermal adsorption experiments were carried out for P using a single filler as well as modified fillers. The columns contained the following; soil, BSM, BSM + 10% maifanite, BSM + 10% maifanite, BSM + 10% WTR, BSM + 10% zeolite. The results from the single filler static isothermal adsorption showed that saturation ranked as follows; WTR > fly ash > zeolite > maifanite > soil. The modified filler with 10% WTR reached an adsorption capacity of

94.29mg/kg which was roughly 3.5 - 4.5 times more than the BSM and the other fillers and was the recommended bioretention amendment.

Shrestha et al., 2019 examined the use of spent lime as a BSM amendment for nutrient uptake in a two-part study involving a field-based mesocosm experiment and a laboratory column study. In the field study, eight different soil medias were used and replicated four times in a raised bed design. Different levels of manure were used in the plots as opposed to synthetic storm water to dose the plots with varying levels P and N. This was a comparative study between plot that had the spent lime and those that did not, soil treatments were randomly assigned. The effluent from each mesocosm was collected on a weekly basis. The volume of compost greatly increased the amount of P leached and spent lime showed significantly less P in the effluent. In the lab, eleven PVC columns were constructed with various media mixes of compost, sand, spent lime and coir. Fourteen 20-second rainfall simulations were conducted using tap water which did not contain any P and the effluent was collected. The laboratory study showed the same as the mesocosms: that the volume of compost in the column increased P leaching and decreased with spent lime and a mixture or spent lime and coir. In conclusion, spent lime performed well in the field and in the lab and is suggested as a cost-effective amendment for BSM design.

In Maryland (Liu & Davis, 2014) were able to study an already existing bioretention cell with an underdrain for a period of 22 months. 5% WTR media was mixed with the top 40cm of the media already in the site and any removed vegetation was replanted. Discrete sampling was used for both inputs and outputs and 12 samples were collected per event. For TP, the peak decreased from 0.66 mg/L in influent to 0.12 mg/L in effluent and for PP 0.61 mg/L in influent to 0.06 mg/L in effluent. Concentrations of SRP in the effluent were essentially constant and ranged from less than 0.01 to 0.09 mg/L. In addition to measuring P concentrations, the flow rate was monitored and WTR showed no effect on filtration rate. This study was significant in that WTR demonstrated the ability to reduce stormwater P loads in a bioretention system that was amended after its installation proving it to be a good enhancement in stormwater treatment.

5.1.3.2 Biochar

Biochar has been studied as a BSM addition to bioretention systems because it has been shown to adsorb heavy metals, and nutrients (Cao et al. 2009). It has also been shown to enhance plant biomass (Kasak et al.2018). A biochar specific study by Iqbal et al. (2015) examined its effectiveness when mixed with compost in preventing the leaching of N, P and organic carbon. 6-month aged compost and biochar from forest slash were obtained locally. The biochar was also mixed with an 80% yard and 20% food-based compost obtained from the same facility. Nine columns total were built containing: 100% biochar, 100% compost, 100% sand, 100% co-composted biochar, 75% compost/ 25% biochar, 75% compost / 25% co-composted biochar, 30% compost / 70% sand, 30% compost/ 70% layered sand. Each treatment was replicated three times and the columns containing compost and other media were thoroughly mixed aside from the one containing the layered sand. Deionized water was used to irrigate the columns at a flow rate to mimic 6-month 24-hr storms. Adding biochar to the media did not show a reduction in P leaching. There was no difference in the amount of P leached between the compost-layered sand and compost-biochar compared to the pure compost. However, more P was leached from the compost – co-composted biochar mix overall and less leached from the compost-sand blend. This study demonstrated that biochar did not have any positive effect on P leachates and should be used in these systems to address certain metals rather than nutrients.

In a study involving 5 columns, differing filter media and an iron-coated biochar in a saturation zone was examined for its removal potential of N and P (Xiong et al. 2019). Each column had the same media composition: a submerged layer, mulch layer, filter media layer and gravel drainage layer. The filter layer was different for each column; 88% concrete sand and 12% soil (T1), 95% T1 and 4% rice husk biochar and 96% T1 and 4% iron-coated biochar. The two remaining columns had biochar added to the upper layers of the media to explore denitrification. The columns were treated with synthetic runoff at a rate of 3.47 mm/h for 6 h. Overall, there was no significant difference in N removal within the columns. The columns that contained biochar showed lower removal rates for TP which may have been a result of the biochar itself leaching P. The iron-coated biochar showed higher removal of TP and thus the team concluded that adding iron-coated biochar to the upper layers of the media in bioretention cells may enhance P removal. (Xiong et al. 2019). Adding rice husk biochar is not recommended if P leaching is of concern.

In other studies, biochar has been shown to be effective in treating certain contaminants in urban stormwater runoff (Reddy et al. 2014). In a column study with a focus on biochar derived from waste wood pellets, the columns were designed to have a layer of biochar in between layers of pea gravel with the same thickness to allow for uniform flow conditions through the biochar. The columns were treated with synthetic stormwater and contaminant concentrations were compared between the influent and effluent. Flushing the columns with synthetic stormwater with a phosphate concentration of 0.82mg/L resulted in P concentrations in the effluent of 0.4 to 0.52mg/L, with a removal efficiency of 47%. For N, there was a removal efficiency of over 85% and cadmium, chromium, copper, lead, nickel, and zinc concentrations were decreased by 18, 19, 65, 75, 17, and 24%, respectively. Contrary to other studies, overall, the biochar showed a significant reduction in P when the influent and effluent concentration were compared and showed to be an effective medium for nutrient removal and certain contaminants.

5.1.3.3 Fly Ash, Iron, Red Mud, and Other Amendments

Three different types of BSM were tested in a study by Yu et al., 2015 including sludge pyrolysis and two types of soils. P adsorption kinetics and P adsorption isotherms were both measured using batch experiments. Al, Fe, and Ca were found to be the main components in all three media, and the Al and Ca were higher in the sludge pyrolysis residue. Adsorption kinetics of P proved to be faster with the sludge pyrolysis and results from the adsorption isothermal experiment showed that the sludge pyrolysis residue is an effective adsorbent to remove P from water. Overall, this amendment showed promise in working as a filter media with bioretention systems.

Other common enhancements to the media include fly ash, iron and red mud and sludge pyrolysis residue. In Oklahoma, four bioretention cells were constructed with the main filter media being a blend of sand and fly ash (Kandel et al., 2017). These cells were analyzed by collecting soil samples and three of the cells' influent and effluent were compared. Various techniques were used to measure TP, SRP and Mehlich phosphorus. Examination of the soil samples revealed that TP concentration increased over time within the topsoil and filter media of all four cells, however it was not statistically significant. When influent and effluent P concentrations of water samples were compared, TP showed a reduction of 64% to 75% and TP mass showed a reduction of 76% to 93% at the three sites. There was a lot of variability in P concentrations below the cell top layer reflecting the need for a better mixing method if fly ash is to be used as an amendment in the future.

A vegetated mesocosm study focused on the use of WTR, red mud and kraznosem soil and the potential of each to remove P (Lucas & Greenway, 2011). There were seven media mixes total: 10% kraznosem (7% kraznosem soil and 93% turf sand), 20% kraznosem (14% kraznosem soil and 86% turf soil) , 40% kraznosem (30% kraznosem soil and 70% turf soil), 6% red mud (75% turf soil 20% top soil and 5% red mud), 10% red mud (71% turf soil 20% top soil and 9% red mud), 30% WTR (80% turf sand and 20% WTR), and a seventh with a mix of 15% WTR and 40% kraznosem (71% turf sand, 20% kraznosem soil and 9% WTR) . The columns were analyzed for two separate 80-week periods and influents and effluents were compared during different loading regimes. The red mud columns were shown to retain more P than the kraznosem soil columns. The columns containing WTR had an effluent concentration that was below 0.10 mg/L for 90% of the runs when the columns were treated with wastewater. They performed equally well when treated with stormwater retaining up to 99% of P over simulated three decades worth of stormwater. The 10% and 20% kraznosem treatments became ineffective after the second and third stormwater dosings. Red mud showed too much P leaching potential and is not a recommended amendment. The 40% kraznosem mix showed an increase in P uptake until the final dosing. This study also showed the effectiveness of vegetation in aiding P uptake and deemed vegetation essential for the longevity of these systems.

In a review by Penn et al. (2017) over 40 studies were examined for cumulative P removal as a function of cumulative P loading. In addition, retention time, P inflow concentration and the type of P sorption material were also analyzed. In wastewater treatment shale, soil and sand were shown to be the least effective due to their low P sorption capacity which was calculated at a cumulative 21% overall. Fe-based P sorption material was more efficient than Ca in systems with shorter retention times and lower P inflow. For material involving Ca, retention times must be maximized to increase efficiency. Flow rate and retention time are majorly influenced by the P sorption material and its hydraulic conductivity. Further research into P sorption material re-use rather than replacement will make these systems more cost effective and increase their overall use in stormwater treatment.

5.1.3.4 Amendments Summary

In the studies reviewed biochar showed conflicting results when added to the BSM to remove P and further research may be necessary to determine its efficiency. Fly ash and sludge pyrolysis showed promise in TP removal but fly ash may require a better mixing method when added to the BSM. Red mud and kraznosem soil were not effective at removing P unless WTR was also in the mix. Overall, WTR seems the most efficient amendment for these systems but further research is necessary to determine how this could change with flow conditions (O’Neill & Davis, 2012a, b). In addition, further research is necessary to determine if any amendments could potentially cause blockage and decrease the productivity of the bioretention system (Li et al., 2018).

5.1.4 Minnesota Specific Designs

Previous studies have provided significant insight into improving the design, performance and maintenance of biofiltration practices in cold climates and what factors, such as frost, may influence their performance. However, Minnesota is lacking in resources and tools that aid in the design of a system that reduces the leaching of phosphorus.

There have been studies that have focused specifically on bioretention performance and cold weather climates. One such study that was specific to Minnesota showed that bioretention systems continued to infiltrate at varying levels throughout the winter (LeFevre et al., 2009). The study had a duration of three years and four existing bioretention cells were selected for analysis within the greater Twin Cities area. Hydrologic performance, infiltration and frost type analysis were performed at all the sites. A well-draining soil was noted as an essential design characteristic to maintain good infiltration rates. If the soil quality is poor, an underdrain is necessary to maintain function. It was also discovered that the type of frost that forms has a stronger influence on infiltration than the presence of frost or its depth.

In a cold temperature specific column study in Sweden by (Blecken et al., 2010), mesocosms were kept in three separate temperature-controlled rooms at an average of 2, 7 and 20°C. The BSM consisted of a top layer of sand and fine gravel and a bottom layer of medium to fine sand. Semi-artificial stormwater (which included natural sediment from a stormwater gully pot and laboratory grade chemicals added to tap water) was used and the concentration of target pollutants was measured. Each column was dosed with 15mL of stormwater twice a week for 12 weeks. Inflow was compared to outflow and TP average removal was 91.4 ± 6.6% and removal was not influenced by temperature but improved with run time. The percentage of dissolved P in the outflow was higher and increased with temperature. The average DP after two sampling events was 12.7%, 15.5% and 17.8% at 2, 7 and 20 °C, respectively. TSS was not significantly affected by temperature, N removal was poor, and N leaching was shown to increase with temperature which may have been attributed to the vegetation selected.

In Finland, a study by Valtanen et al. (2017) took place in a large scale lysimeter facility where bioretention systems were underground in a bunker and the tops were exposed to open air. Eight lysimeters were built containing an organic soil layer, sand filter layer, transition layer of fine gravel, drainage layer of coarse gravel and saturated layer of coarse gravel. Each system was irrigated 6 times during the experiment, one in autumn, three in spring and two in summer. No irrigations were performed in the winter because no runoff is generated in freezing temperatures. Based on stormwater measurements from a nearby town, Zn, Cu, Al, P and N were studied. Inflow and outflow measurements were compared, and all systems showed close to 100% P retention throughout each season. On the contrary, N was not well retained during the first irrigations, but retention increased over time. This large-scale study showed that biofiltration systems perform in cold climates, but there is a need for a longer-term study to determine their efficacy.

An examination of low impact development designs in cold climates included two bioretention systems at the University of New Hampshire Stormwater Center field facility (Roseen et al., 2009). A total of 27 rainwater events were examined for two winters and two summers. Frost penetration did not influence the overall hydraulic performance of the systems, and it was determined that frozen media may still have significant permeability. Influent concentrations of various contaminants were compared to their effluent concentrations and TP removal in the bioretention systems did not show a significant decline in performance in winter months.

The effect of freeze thaw cycles on bioretention media was examined in a study by Ding et al. (2019). Four soil samples were collected and three ended up being used, from an existing bioretention site in Ontario, Canada. The cell had mulch on the top and was amended with a media enriched with Al and Fe oxides before installation. To evaluate the effects of the freeze thaw cycles (FTC), six replicate injection experiments were performed. The injection solution of 25 mg/L each of $PO_4$3− (8.33 mg/L of PO_4^{3-}-P), NO_3^- (5.65 mg/L of NO_3^--N), and bromide (Br^-) was prepared in 0.01 M calcium chloride ($CaCl_2$) to mimic contaminated surface runoff. Concentrations of N and P were higher than average stormwater to test the systems under extreme conditions. Overall, more than 98% of TDP was removed from the columns during all of the FTCs. The effluent P concentrations fell below 0.15 mg/L and ≤2% of the added stormwater phosphate was present in the effluent. Further research is needed to study more variables regarding FTC but overall, this study found that when designed properly cells will perform well in cold climates.

In a critical literature review by Kratky et al., 2017 it was noted that various studies suggest that removal of organics, heavy metals and nutrients is temperature dependent yet also may be dependent on BSM. Another obstacle with cold climates is the freeze thaw cycle that may influence plant root growth and cause some clogging and may also influence the system's permeability. In addition, plants must be selected for winter hardiness, nitrogen degradation and must be salt tolerant. More research is needed to examine the relationship between coarse media and cold weather hydraulic performance, amendments that will enhance contaminant removal and longevity in these climates.

5.1.5 Literature Review Summary

As the bioretention practice becomes more accepted as an effective way to treat stormwater runoff, more research is required to determine the cost effectiveness of design including the potential cost of maintenance to maintain their optimal performance and longevity in pollutant and nutrient removal. Certain studies have shown that the top layer must be removed every two years to prevent clogging of the filter (Hatt et al., 2009). Another study has shown that vegetation removal and maintenance is an effective way to increase nutrient removal as well as prevent leaching (Davis et al. 2006). As research continues, and the dangers of P leaching are addressed, it is evident that the effectiveness of these systems is region specific in terms of BSM and vegetation selection and therefore further developing an optimal design for Minnesota is crucial when it comes to addressing local stormwater treatment.

5.2 EDUCATION AND TECHNOLOGY TRANSFER PLAN

The Education and Technology Transfer Plan consists of three primary mechanisms: outdoor signage, presentations (oral and poster), and practitioner training. The current and expected outputs from these mechanisms are described in the sections below.

5.2.1 Outdoor Signage

The outdoor mesocosm experiments were performed in the decommissioned spillway adjacent to the Outdoor StreamLab at St. Anthony Falls Laboratory. This area is immediately adjacent to, and visible from, Water Power Park which is typically open to the public during the spring, summer, and fall. This project developed signage for display in Water Power Park to give visitors insights into urban runoff, pollution, treatment (such as biofiltration), and environmental research including the mesocosm experiments. This reaches a diverse public audience that would not typically expect to learn about urban water resources topics while visiting Water Power Park and may engage local residents that are completely unaware of the complex stormwater management system in their ultra-urban neighborhood. The signage developed for this project is shown in Figure 30.

Figure 30: Signage developed for display at Water Power Park, Minneapolis, MN.

5.2.2 Presentations

Several presentations have been given to individuals and practitioners in Minnesota and beyond, and are listed below:

- February 1, 2019. "Biofiltration Media Optimization." Oral Presentation. Technical Advisory Team meeting. In-person and Virtual.
- April 17, 2019. Invited to present "Biofiltration Media Optimization." Oral Presentation. Minnesota Composting Council Operator Training. Chaska, MN.
- July 18, 2019. Invited to present "Biofiltration Media Optimization." Oral Presentation. Minnesota Stormwater Research Council Annual Meeting. Minneapolis, MN.
- February 27, 2020. Moderated Panel Discussion for "Bioretention and Treatment Media: A Phosphorus Story." Panel Discussion Moderator. Minnesota Stormwater Seminar Series. Minneapolis, MN.
- May 6, 2020. Invited to present "Biofiltration Media Optimization." Oral Presentation. MPCA Updates on engineered (bioretention) media. Virtual.
- May 7, 2020. Moderated Panel Discussion for "Bioretention Soil Media, Vegetation, and Maintenance: Lessons learned from Green Stormwater Infrastructure research in Vermont." Panel Discussion Moderator. Minnesota Stormwater Seminar Series. Virtual
- October 20, 2020. "Biofiltration Media Optimization: Phase I Results and Phase II Preview." Poster Presentation. Minnesota Water Resources Conference 2020. Virtual.
- February 8, 2021. "Biofiltration Media Optimization: Phase I Review and Phase II Preview." Oral Presentation. Technical Advisory Team meeting. Virtual.
- May 26, 2021. Invited to present "Biofiltration (Engineered) Media Optimization: Phase I Review & Engineering Media w/ Compost." Oral Presentation. MPCA Engineered/Bioretention Media Webinar. Virtual.
- July 15, 2021. Invited to present "Removing CECs from Stormwater with Biofiltration." Oral presentation. 2021 LCCMR Proposal Process. Virtual.
- September 16, 2021. Invited to present "Biofiltration Media Optimization: Phase I Results, Phase II preliminary results, and Future Work." Minnesota Stormwater Research Spotlight Series. Virtual
- October 20, 2021. Invited to present "St. Anthony Falls Laboratory; A tour for delegates from Finland and the MN DEED Office." Oral Tour. Finnish Delegation and MN Department of Employment and Economic Development. Minneapolis, MN.
- October 21, 2021. Moderated Panel Discussion for "Spatial and Temporal Scaling of Biofilter Media Performance Data and Updates to WinSLAMM." Panel Discussion Moderator. Minnesota Stormwater Seminar Series. Virtual
- November 11, 2021. Invited to present "St. Anthony Falls Laboratory; A tour for the MN Chamber of Commerce." Oral Tour. Minnesota Chamber of Commerce. Minneapolis, MN.
- February 3, 2022. Invited to present "New biofiltration media to remove chemicals of emerging concern (CECs) from stormwater." Oral Presentation (virtual). 2022 Minnesota Stormwater Research Council Proposal Process. Virtual.
- February 4, 2022. Invited to present "Can Biofilters Remove Phosphate and Grow Plants?" University of Vermont Plant and Soil Science Department Seminar. Virtual.
- February 9, 2022. Invited to present "Can Biofiltration Media Capture Phosphate and Grow Plants?" The North American Stormwater and Erosion Control Association (NASECA) of Wisconsin 19th Annual Conference & Trade Show. Madison, WI.

- February 18, 2022. Invited to present "Biofiltration Media: What to Choose and Why." Oral presentation with Mike Trojan. 2022 International Erosion Control Association 50th Annual Conference. Minneapolis, MN.
- February 18, 2022. Invited to lead "St. Anthony Falls Laboratory; A tour for participants of 2022 IECA Annual Conference and Trade Show." Oral Tour. 2022 International Erosion Control Association 50th Annual Conference. Minneapolis, MN.
- March 9, 2022. Invited to present "Biofiltration Media: What to Choose and Why." Oral Presentation. 23th Annual Fox-Wolf Watershed Alliance Conference. Green Bay, WI.
- April 19, 2022. Invited to lead "Urban Stormwater Runoff Management." Oral presentation with Poornima Natarajan. St. Anthony Falls Laboratory; A tour for UMN CSE Dean Andrew G. Alleyne. Minneapolis, MN.
- May 4, 2022. Invited to lead "St. Anthony Falls Laboratory; A tour for Eric Watkins Research Group." Oral tour. St. Anthony Falls Laboratory. Minneapolis, MN.
- May 12, 2022. Invited to present "Can Biofilters Remove Phosphate and Grow Plants?" Oral presentation. 2022 Ohio Stormwater Conference. Sandusky, OH.
- May 19, 2022. Invited to lead "St. Anthony Falls Laboratory; A tour for MN GreenCorps Members." Oral tour. St. Anthony Falls Laboratory. Minneapolis, MN.
- June 8, 2022. Invited to lead "St. Anthony Falls Laboratory; A tour for MN Watershed Partners." Oral tour. St. Anthony Falls Laboratory. Minneapolis, MN.
- July 21, 2022. Invited to present "Biofiltration Media Optimization." Oral Presentation. Minnesota Stormwater Research Council Annual Meeting. Minneapolis, MN.
- August 9, 2022. Invited to present "Removing CECs from Stormwater with Biofiltration." Oral presentation. 2022 LCCMR Proposal Process. Virtual.

The poster that was presented at the 2020 Minnesota Water Resources Conference is shown in Figure 31.

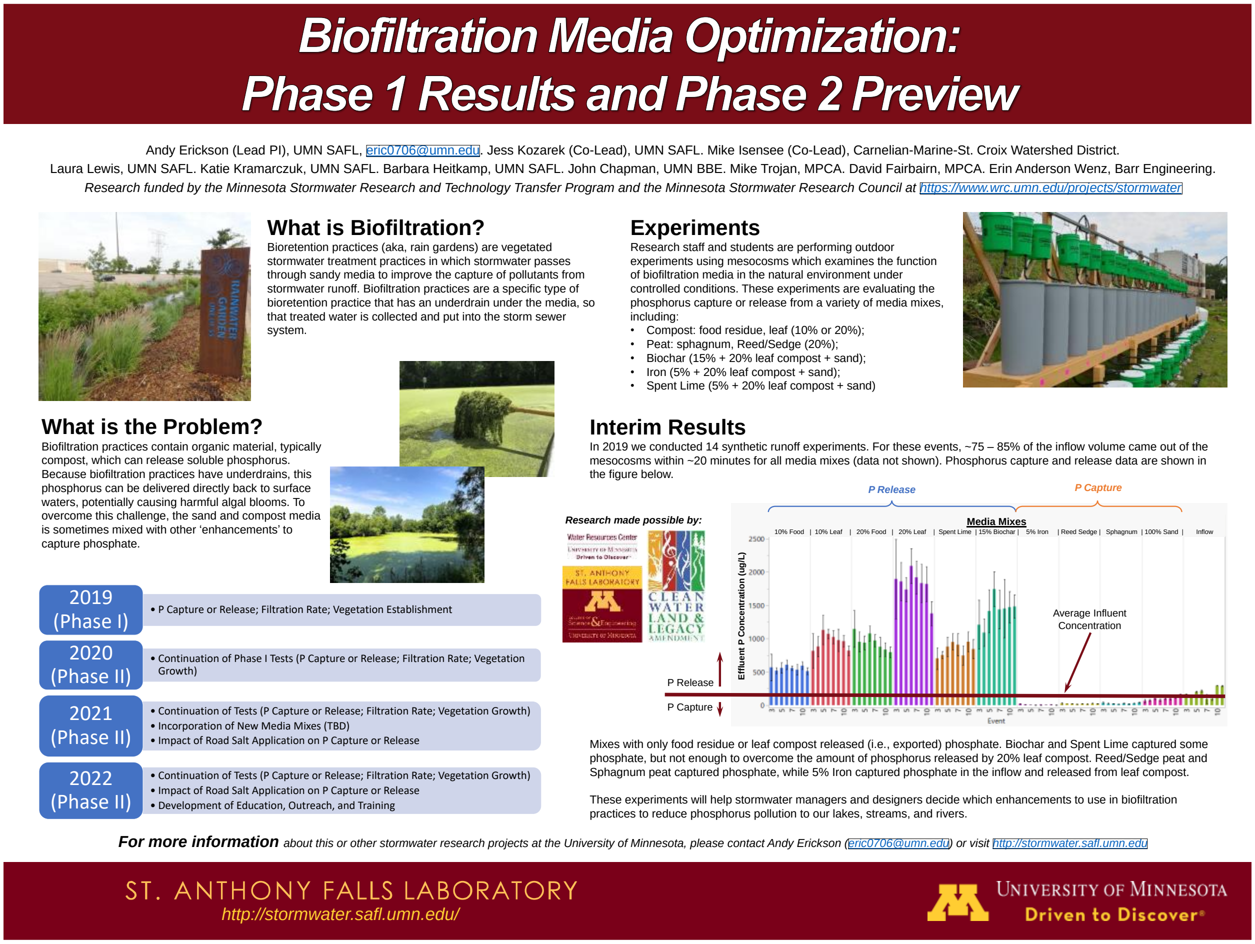

Figure 31: Poster presented at the 2020 Minnesota Water Resources Conference.

5.2.3 Practitioner Training

Planning for incorporating the results of this research into professional training is underway. A draft Canvas course has been developed by University of Minnesota Erosion and Stormwater Management Certification Program (https://www.erosion.umn.edu/) and will be revised and updated with the information from this final report. Additional planning and discussions are underway with the Minnesota Pollution Control Agency to develop a webinar on biofiltration media, similar to those hosted by the MPCA (https://stormwater.pca.state.mn.us/index.php?title=Stormwater_Manual_webinars). The audience for this course is stormwater practitioners and professionals.

The project team intends to give additional webinars on the results of this research in the future. A webinar like this would include data and recommendations from this report. The audience for this training would include stormwater practitioners and citizen scientists. In addition, pertinent results and recommendations may be included in the Minnesota Stormwater Manual (e.g., https://stormwater.pca.state.mn.us/index.php/Design_criteria_for_bioretention) pending approval by MPCA staff.

Finally, the results and recommendations from this research will continue to be presented at local and national conferences and via webinars to practitioners and interested parties. A summary newsletter article will also be developed and disseminated via UPDATES, the University of Minnesota stormwater research e-newsletter distributed to over 2000 email subscribers. The results will also be submitted for inclusion in other e-news lists and distribution channels such as the MPCA and Water resources Center email lists.

5.3 REFERENCES

Al-Mashaqbeh, O., & McLaughlan, R. G. (2012). Non-equilibrium zinc uptake onto compost particles from synthetic stormwater. *Bioresource Technology*, *123*, 242–248. https://doi.org/10.1016/j.biortech.2012.07.034

ASTM (2017). Standard Test Method for Determination of Erosion Control Product (ECP) Ability to Encourage Seed Germination and Plant Growth Under Bench-Scale Conditions. ASTM International. West Conshohocken, PA. https://www.astm.org/d7322_d7322m-17.html

Blecken, G. T., Zinger, Y., Deletic, A., Fletcher, T. D., Hedström, A., & Viklander, M. (2010). Laboratory study on stormwater biofiltration: Nutrient and sediment removal in cold temperatures. *Journal of Hydrology*, *394*(3–4), 507–514. https://doi.org/10.1016/j.jhydrol.2010.10.010

Bratieres, K., Fletcher, T. D., Deletic, A., & Zinger, Y. (2008). Nutrient and sediment removal by stormwater biofilters: A large-scale design optimisation study. *Water Research*, *42*(14), 3930–3940. https://doi.org/10.1016/j.watres.2008.06.009

Brinton, W. (2019). Soil CO2 Respiration: Official Solvita Instructions (CO2-Burst), SOP 2019 Rev. 1 (DCR Models 701.2+). Solvita & Woods End Laboratories. Accessed at https://solvita.com/publications/ on 2020/11/02.

Brinton, W. and E. Evans. (2006). Monitoring Carbon-Dioxide and Ammonia from Composts by Use of Solvita® Digital Colour Reader. Solvita & Woods End Laboratories. Proceedings of ORBIT 2006, Weimar, Germany. Accessed at https://solvita.com/publications/ on 2020/11/17.

Brown, R. A., Iii, W. F. H., & ASCE, M. (2010). *Impacts of Construction Activity on Bioretention Performance*. https://doi.org/10.1061/(ASCE)HE.1943-5584.0000165

Cao, X., L. Ma, B. Gao, W. Harris. (2009). Dairy-manure derived biochar effectively sorbs lead and atrazine Environ. Sci. Technol., 43 (2009), pp. 3285-3291. https://doi.org/10.1021/es803092k

Chahal, M. K., Shi, Z., & Flury, M. (2016). Nutrient leaching and copper speciation in compost-amended bioretention systems. *Science of the Total Environment*, *556*, 302–309. https://doi.org/10.1016/j.scitotenv.2016.02.125

Davis, A. P., Shokouhian, M., Sharma, H., & Minami, C. (2006). Water Quality Improvement through Bioretention Media: Nitrogen and Phosphorus Removal. *Water Environment Research*, *78*(3), 284–293. https://doi.org/10.2175/106143005x94376

Ding, B., Rezanezhad, F., Gharedaghloo, B., van Cappellen, P., & Passeport, E. (2019). Bioretention cells under cold climate conditions: Effects of freezing and thawing on water infiltration, soil structure, and nutrient removal. *Science of the Total Environment*, *649*, 749–759. https://doi.org/10.1016/j.scitotenv.2018.08.366

Erickson, A. J., Weiss, P. T., Gulliver, J. S., Erickson, A. J., Weiss, P. T., & Gulliver, J. S. (2013). Impacts and Composition of Urban Stormwater. In *Optimizing Stormwater Treatment Practices* (pp. 11–22). Springer New York. https://doi.org/10.1007/978-1-4614-4624-8_2

Erickson, A.J., J.S. Gulliver and P.T. Weiss. (2007). "Enhanced Sand Filtration for Storm Water Phosphorus Removal." Journal of Environmental Engineering, 133(5), 485–497. http://dx.doi.org/10.1061/(ASCE)0733-9372(2007)133:5(485)

Erickson, A.J., J.S. Gulliver and P.T. Weiss. (2012). "Capturing Phosphates with Iron Enhanced Sand Filtration." Water Research, 46(9), 3032–3042. http://dx.doi.org/10.1016/j.watres.2012.03.009

Erickson, AJ, Kozarek, JL, Kramarczuk, KA, and Lewis, L. (2021). "Biofiltration Media Optimization – Phase 1 Final Report." Project Report No. 593, St. Anthony Falls Laboratory, University of Minnesota, Minneapolis, MN. January 2021.

Hageman, P. L. (2007). "U.S. Geological Survey field leach test for assessing water reactivity and leaching potential of mine wastes, soils, and other geologic and environmental materials." U.S. Geological Survey, Reston, VA

Hatt, B. E., Fletcher, T. D., & Deletic, A. (2008). Hydraulic and pollutant removal performance of fine media stormwater filtration systems. *Environmental Science and Technology*, *42*(7), 2535–2541. https://doi.org/10.1021/es071264p

Hatt, B. E., Fletcher, T. D., & Deletic, A. (2009). Hydrologic and pollutant removal performance of stormwater biofiltration systems at the field scale. *Journal of Hydrology*, *365*(3–4), 310–321. https://doi.org/10.1016/j.jhydrol.2008.12.001

Henderson, C., Greenway, M., & Phillips, I. (2007). Removal of dissolved nitrogen, phosphorous and carbon from stormwater by biofiltration mesocosms. *Water Science and Technology*, *55*(4), 183–191. https://doi.org/10.2166/wst.2007.108

Hsieh, C.-H., Davis, A. P., & Needelman, B. A. (2007). *Bioretention Column Studies of Phosphorus Removal from Urban Stormwater Runoff*. https://doi.org/10.2175/106143006X111745

Hunt, W. F., Jarrett, A. R., Smith, J. T., & Sharkey, L. J. (2006). Evaluating Bioretention Hydrology and Nutrient Removal at Three Field Sites in North Carolina. Journal of Irrigation and Drainage Engineering, 132(6), 600–608. https://doi.org/10.1061/(ASCE)0733-9437(2006)132:6(600)

Hurley, S., Shrestha, P., & Cording, A. (2017). Nutrient Leaching from Compost: Implications for Bioretention and Other Green Stormwater Infrastructure. *Journal of Sustainable Water in the Built Environment*, *3*(3), 04017006. https://doi.org/10.1061/JSWBAY.0000821

Iqbal, M. Garcia-Perez, M. Flury. (2015). Effect of biochar on leaching of organic carbon, nitrogen, and phosphorus from compost in bioretention systems. Sci. Total Environ., 521–522 (2015), pp. 37-45. https://doi.org/10.1016/j.scitotenv.2015.03.060

Jay, J.G., S.L. Brown, K. Kurtz, F. Grothkopp. (2017). Predictors of phosphorus leaching from bioretention soil media. J. Environ. Qual., 46 (2017), pp. 1098-1105 https://doi.org/10.2134/jeq2017.06.0232

Kandel, S., Vogel, J., Penn, C., & Brown, G. (2017). Phosphorus Retention by Fly Ash Amended Filter Media in Aged Bioretention Cells. *Water*, *9*(10), 746. https://doi.org/10.3390/w9100746

Kasak, K., J. Truu, I. Ostonen, J. Sarjas, K. Ookaup, P. Paiste, M. Kõiv-Vainik, Ü. Mander, M. Truu. (2018) Biochar enhances plant growth and nutrient removal in horizontal subsurface flow treatment wetland. Sci. Total Environ., 639 (2018), pp. 67-74, 10.1016/j.scitotenv.2018.05.146

Kramarczuk, K.A. (2022). Optimizing Biofiltration Media for the Capture of Phosphate and the Support of Vegetation Growth. Master's Thesis. University of Minnesota. June 2022. https://hdl.handle.net/11299/241550

Kratky, H., Li, Z., Chen, Y. et al. A critical literature review of bioretention research for stormwater management in cold climate and future research recommendations. Front. Environ. Sci. Eng. 11, 16 (2017). https://doi.org/10.1007/s11783-017-0982-y

Lefevre, N. J., Davidson, J. D., & Oberts, G. L. (2009). *Bioretention of Simulated Snowmelt: Cold Climate Performance and Design Criteria*. http://www.dakotacountyswcd.org/pdfs/Bioretention%20of%20Simulated%20Snowmelt%202009.pdf

Li, J., & Davis, A. P. (2016). A unified look at phosphorus treatment using bioretention. In *Water Research* (Vol. 90, pp. 141–155). Elsevier Ltd. https://doi.org/10.1016/j.watres.2015.12.015

Li, J., Li, L., Dong, W., & Li, H. (2018). Purification effects of amended bioretention columns on phosphorus in urban rainfall runoff. *Water Science and Technology*, *78*(9), 1937–1945. https://doi.org/10.2166/wst.2018.464

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. *Journal of Environmental Management*, *147*, 24–33. https://doi.org/10.1016/j.jenvman.2014.04.042

Liu, J., & Davis, A. P. (2014). Phosphorus speciation and treatment using enhanced phosphorus removal bioretention. *Environmental Science and Technology*, *48*(1), 607–614. https://doi.org/10.1021/es404022b

Liu, J., Sample, D., Bell, C., & Guan, Y. (2014). Review and Research Needs of Bioretention Used for the Treatment of Urban Stormwater. *Water*, *6*(4), 1069–1099. https://doi.org/10.3390/w6041069

Logsdon, S. D., & Sauer, P. A. (2016). Nutrient leaching when compost is part of plant growth media. *Compost Science and Utilization*, *24*(4), 238–245. https://doi.org/10.1080/1065657X.2016.1147398

Lucas, W.C., & Greenway, M. (2011). Phosphorus Retention by Bioretention Mesocosms Using Media Formulated for Phosphorus Sorption: Response to Accelerated Loads. *Journal of Irrigation and Drainage Engineering*, *137*(3), 144–153. https://doi.org/10.1061/(ASCE)IR.1943-4774.0000243

Maguire, R. O., & Sims, J. T. (2002). Soil Testing to Predict Phosphorus Leaching. Journal of Environmental Quality, 31(5), 1601–1609. https://doi.org/10.2134/jeq2002.1601

Minnesota Pollution Control Agency (MPCA). (2022). "Minnesota Stormwater Manual: Design Criteria for Bioretention." https://stormwater.pca.state.mn.us/index.php/Design_criteria_for_bioretention. Accessed December 28, 2022; page last edited on December 1, 2022.

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. *Science of the Total Environment*, *537*, 294–303. https://doi.org/10.1016/j.scitotenv.2015.07.157

O'Neill, S. W., and Davis, A. P. (2012a). "Water treatment residual as a bioretention amendment for phosphorus. I: Evaluation studies." J. Environ. Eng., 10.1061/(ASCE)EE.1943-7870.0000409, 318–327.

O'Neill, S. W., and Davis, A. P. (2012b). "Water treatment residual as a bioretention amendment for phosphorus. II: Long-term column studies." J. Environ. Eng., 10.1061/(ASCE)EE.1943-7870.0000436, 328–336.

Paus, K. H., Morgan, J., Gulliver, J. S., & Hozalski, R. M. (2014a). Effects of bioretention media compost volume fraction on toxic metals removal, hydraulic conductivity, and phosphorous release. *Journal of Environmental Engineering (United States)*, *140*(10), 04014033. https://doi.org/10.1061/(ASCE)EE.1943-7870.0000846

Paus, K.H., J. Morgan, J. S. Gulliver, T. Leiknes, and R. M. Hozalski. (2014b). "Effects of Temperature and NaCl on Toxic Metal Retention in Bioretention Media." Journal of Environmental Engineering, 140(10), 04014034. http://dx.doi.org/10.1061/(ASCE)EE.1943-7870.0000847

Paus, K.H., J. Morgan, J.S. Gulliver, T. Leiknes and R.M. Hozalski. (2014c). "Assessment of the Hydraulic and Toxic Removal Capacities of Bioretention Cells after 2 to 8 Years of Service." Water, Soil and Air Pollution, 225 (1803). http://dx.doi.org/10.1007/s11270-013-1803-y

Penn, C., Chagas, I., Klimeski, A., & Lyngsie, G. (2017). A Review of Phosphorus Removal Structures: How to Assess and Compare Their Performance. *Water*, *9*(8), 583. https://doi.org/10.3390/w9080583

Reddy, K.R., T. Xie, S. Dastgheibi. (2014). Evaluation of biochar as a potential filter media for the removal of mixed pollutants from urban water runoff. J. Environ. Eng., 140 (2014), p. 04014043. https://doi.org/10.1061/(ASCE)EE.1943-7870.0000872

Roseen, R. M., Ballestero, T. P., Houle, J. J., Avellaneda, P., Briggs, J., Fowler, G., & Wildey, R. (2009). Seasonal Performance Variations for Storm-Water Management Systems in Cold Climate Conditions.

Journal of Environmental Engineering, *135*(3), 128–137. https://doi.org/10.1061/(ASCE)0733-9372(2009)135:3(128)

Roy-Poirier, A., Champagne, P., & Filion, Y. (2010). Review of bioretention system research and design: past, present, and future. Journal of Environmental Engineering, 136(9), 878–889. https://doi.org/10.1061/(ASCE)EE.1943-7870.0000227

Shrestha, P., Faulkner, J. W., Kokkinos, J., & Hurley, S. E. (2020). *Influence of low-phosphorus compost and vegetation in bioretention for nutrient and sediment control in runoff from a dairy farm production area*. https://doi.org/10.1016/j.ecoleng.2020.105821

Shrestha, P.; Salzl, M.T.; Jimenez, I.J.; Pradhan, N.; Hay, M.; Wallace, H.R.; Abrahamson, J.N.; Small, G.E. Efficacy of Spent Lime as a Soil Amendment for Nutrient Retention in Bioretention Green Stormwater Infrastructure. Water 2019, 11, 1575. https://doi.org/10.3390/w11081575

Silvertooth, J. R., & Nason, J. A. (2014). *AN ABSTRACT OF THE THESIS OF Title: Evaluation of Copper Removal from Stormwater Runoff Using Compost and Apatite II™*. Oregon State University.

Trowsdale, S.A. and R. Simcock. (2011). Urban stormwater treatment using bioretention. J. Hydrol., 397 (2011), pp. 167-174. https://doi.org/10.1016/j.jhydrol.2010.11.023

US EPA. (1999). *Preliminary Data Summary of Urban Storm Water Best Management Practices*. EPA-821-R-99-012. U.S. Environmental Protection Agency, Washington, D.C.

USEPA (U.S. Environmental Protection Agency). (1983). "Results of the Nationwide Urban Runoff Program." Final Rep., Washington, DC.

Valtanen, M., Sillanpää, N., & Setälä, H. (2017). A large-scale lysimeter study of stormwater biofiltration under cold climatic conditions. *Ecological Engineering*, *100*, 89–98. https://doi.org/10.1016/j.ecoleng.2016.12.018

Xiong, J., S. Ren, Y. He, X.C. Wang, X. Bai, J. Wang, M. Dzakpasu. (2019). Bioretention cell incorporating Fe-biochar and saturated zones for enhanced stormwater runoff treatment. Chemosphere, 237 (2019), p. 124424. https://doi.org/10.1016/j.chemosphere.2019.124424

Yang, F., Fu, D., Liu, S., Zevenbergen, C., & Singh, R. P. (2020). Hydrologic and pollutant removal performance of media layers in bioretention. *Water (Switzerland)*, *12*(3), 921. https://doi.org/10.3390/w12030921

Yu, G., Zhang, B., Lu, X. et al. (2015). Efficient removal of phosphorus in bioretention system by sludge pyrolysis residue. Arab J Geosci 8, 3491–3499 (2015). https://doi.org/10.1007/s12517-014-1462-3

Zhang, B.; Li, J.; Li, Y.; Li, H. Adsorption Characteristics of Several Bioretention-Modified Fillers for Phosphorus. Water 2018, 10, 831. https://doi.org/10.3390/w10070831