# Evaluation of Soil Class Proxies for Hydrologic Performance of In Situ Bioinfiltration Systems

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**Abstract:** The hydrologic performance of in situ bioinfiltration systems (bioretention systems with no fill media or underdrain) is quantified and soil classes are evaluated as proxies for design requirements. A one-dimensional (1D) Richard's equation model of a bioinfiltration system is used in combination with a dataset of soil hydraulic properties to conduct a Monte Carlo analysis of the effect of soil hydraulic properties; the results are summarized both by soil textural class and by hydrologic soil group (HSG), showing that textural class is generally a poor proxy for estimating the infiltration performance of in situ bioinfiltration cells ($R^2 = 0.40$). Because infiltration measurements are required to estimate the HSG, they are a better proxy for bioinfiltration performance ($R^2 = 0.89$). It is found that soil proxies do provide certain reliable guidelines: HSG-D soils always require engineered fill media with an underdrain; whereas underdrains are not necessary for sand, loamy sand, HSG-A, and HSG-B native soils. Minimum bounds on the design capture volume are generated for these soils which may be substantially larger than the surface storage volume. **DOI: 10.1061/JSWBAY.0000813.** 



## Introduction

Bioretention systems (also known as rain gardens) and bioinfiltration systems (defined here as a subset of bioretention that are designed for infiltration and do not have an underdrain) have been applied increasingly in recent years for stormwater control. Depending on the design configuration, bioretention systems can achieve several goals of stormwater control: water quality improvement, peak flow reduction, volume control, as well as attenuating the natural hydrologic cycle imbalance via groundwater recharge and evapotranspiration (ET) (Roy-Poirier et al. 2010; Davis et al. 2012; Hunt et al. 2012). In addition to the complex interactions of these sometimes-competing design goals, bioretention design guidelines still generally lack basic quantitative hydrologic performance metrics (Schneider and McCuen 2006; Davis et al. 2009). The consequences of this lack of performance data are that there is a lack of consensus across the country about design guidelines, and that design guidelines can fail to allow for flexibility in achieving region-specific design goals.

For example, one of the primary design questions for bioretention is when to use an underdrain with fill material. Many jurisdictions consider bioretention to be a water quality filter and mandate the use of underdrains at all sites. However, bioinfiltration has been shown to be successful at infiltrating significant volumes of water for periods exceeding a decade, while significantly reducing pollutant loads from the watershed (e.g., Emerson and Traver 2008; Davis et al. 2012; Komlos and Traver 2012; Welker et al. 2013). When underdrains are not strictly mandated for all sites, there is still a lack of consensus about the minimum soil infiltration criteria to determine the necessity of an underdrain. Criteria range from a conservative 0.25 cm/h in Pennsylvania (PADEP 2006) to an order of magnitude higher in many other jurisdictions (e.g., PGDES 2007; Virginia DCR 2011). Often there may also be restrictions on textural classes allowed as bioinfiltration sites in addition to the infiltration rate limitation. For example, Maryland Department of the Environment (MDE 2000) specifies a minimum infiltration rate of 1.32 cm/h and a clay content of less than 20% and a silt/clay content of less than 40%, which means that only sand, loamy sand, and sandy loam are acceptable infiltration soils. Davis et al. (2009) cite this uncertainty about underdrain application as a primary focus of needed research.

Furthermore, the soils located at a site dictate the fill media requirements in addition to the underdrain requirements. It is almost universally assumed that a bioretention or bioinfiltration system will be filled with a highly permeable fill media, despite the relative lack of quantitative evidence for or against such a requirement. Additionally, requirements for fill media are often based on the soil textural class as a proxy, despite evidence that such proxies may not be very effective at representing the hydrologic behavior of soils (Gutmann and Small 2005; Twarakavi et al. 2010; Loosvelt et al. 2011). The textural classes are limited in estimating hydrologic behavior because the variabilities of soil grain shape, plasticity, and compaction cause a substantial variation in hydrologic performance within each textural class (e.g., Pitt et al. 2008). Although studies such as Li et al. (2009) show a correlation between fill media and increased hydrologic capture, this conclusion assumes that the fill media are of a particular permeability, however the textural assessment of the soil may not indicate the hydraulic performance very well (Tietje and Hennings 1996). Carpenter and Hallam (2010) confirm this by showing a wide gap between expected and measured performance of different fill media; Fassman-Beck et al. (2015) highlight that the particle size distribution is typically inadequate to estimate the hydraulic conductivity of bioretention media. Once again, Davis et al. (2009) highlight fill media depth, composition, and configuration as some of the

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primary areas of needed research for quantitative design and performance information.

The result of this lack of quantitative evidence is that stormwater control measures are often overdesigned, resulting in unnecessary costs of labor and materials. There is a desire to achieve more sustainable low impact development, and the use of in situ bioinfiltration systems (by in situ bioinfiltration, the authors are referring to bioinfiltration sites that use only the native soils without excavation and refill with a different soil) would decrease the environmental impact of the construction of stormwater controls (Clar 2010; Flynn and Traver 2013), in addition to significantly reducing the cost due to expensive underdrains and fill media. Whereas underdrained sites may be constructed to increase ET through an internal water storage (IWS) zone (Davis et al. 2012) and closely mimic interflow in some situations (DeBusk et al. 2011), bioinfiltration can accomplish many of the same goals along with the opportunity for increased groundwater recharge, which is important in many regions. The aim of this research is to develop quantitative data about the hydrologic performance of in situ bioinfiltration systems that will aid in the development of more efficient and effective bioretention design guidelines. The specific uncertainties in bioretention design that are being targeted in this research are the following:

- What are the native soil criteria that best serve as a proxy for underdrain requirements?
- What is the expected hydrologic capture of a typical in situ bioinfiltration design using soil class as a proxy?

To evaluate an entire soil class, it is necessary to represent the entire range of possible soil hydraulic properties (SHP) that appear within the class. Because of the nonlinear nature of soil/water systems, Monte Carlo methods are needed to translate SHP variation into bioinfiltration performance predictions. Therefore, this research models the range of SHP within each soil class using a database of SHP, and uses these SHP models as inputs into hydrologic simulations of bioinfiltration systems. By using individual soil property models instead of class-average, the variability of SHP can be transferred into a corresponding variability in infiltration performance within each soil class (Fig. 1). It is also necessary to account for possible interactions with other bioinfiltration design parameters such as weather patterns, evapotranspiration potential, and surface storage depth. Therefore, a large number of simulations were performed allowing for a general assessment of the effect of within-class SHP variability across a range of design conditions.

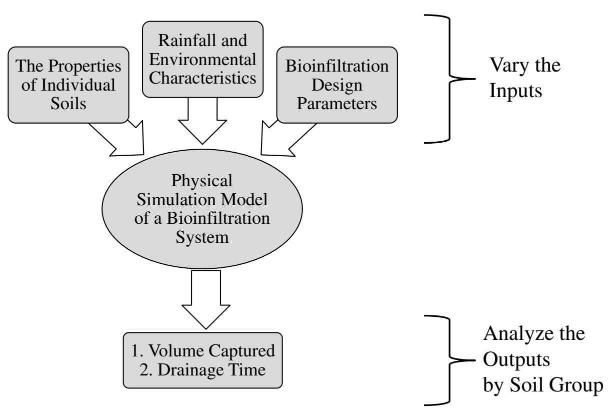


**Fig. 1.** Structure of analysis for determining the effectiveness of soil proxies

## Methods

This research is based on the assumption that the one-dimensional (1D) Richard's equation is an appropriate representation of an in situ bioinfiltration system when combined with appropriate SHP models (e.g., Aravena and Dussaillant 2009). Although a two-dimensional (2D) model would better represent situations where lateral flow is present, such as at soil interfaces or shallow groundwater tables, model complexity (such as boundary conditions and estimating transverse soil properties), and runtimes would be substantially increased over the 1D model. However in the absence of such features, the flow will be primarily vertical and a 1D representation is appropriate.

A 1D discretization of an in situ bioinfiltration system as used in this study is represented in Fig. 2, which shows the three main model zones: the surface water (pond); the modeled soil zone (native soil); and the deep GWT, which is inferred but not modeled. The surface water zone tracks the water level and may route flow to overflow if the level exceeds the maximum pond depth $D_p$. The 1D Richard's equation solver used for this analysis was Soil Water Atmosphere Plant (SWAP) (Kroes et al. 2009). SWAP models the surface water, solves the water fluxes in the soil, and computes the percolation to the deep GWT. This solver was used because of the variable ponded/unponded upper surface boundary condition necessary to simulate a bioinfiltration system, for flexibility in assigning SHP, and for fast CPU times.

In the absence of lateral flow, the 1D Richard's equation represents the exact solution to the movement of water within the soil

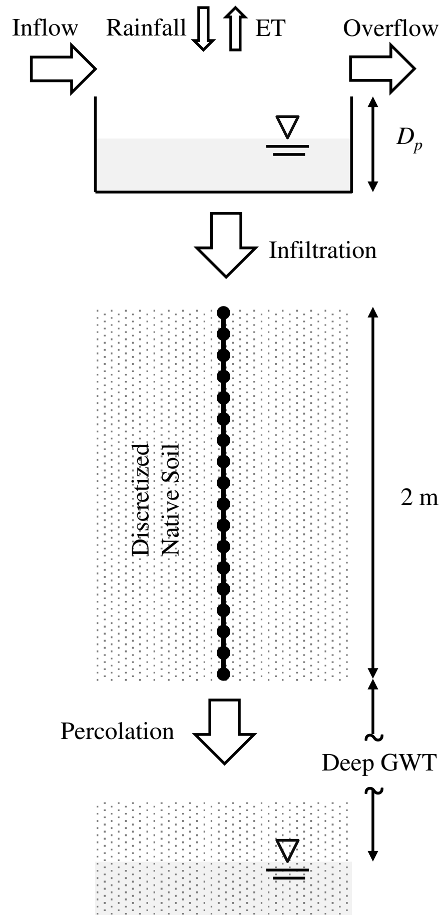


**Fig. 2.** 1D approximation of a bioinfiltration system

column, assuming homogeneity within each soil profile and accuracy of the SHP models, along with the assumption that macropores, air-entrapment, and hysteresis are either not present or are accounted for in the SHP. At the upper surface, a bioinfiltration system is both simpler and more complicated than a watershed; simpler because the surface water ponding is well defined, but more complicated because the basin will typically have a three-dimensional (3D) shape rather than the 1D plug shape represented in Fig. 2. Although it is possible to make some approximations to account for the 3D shape of the basin (e.g., Lee et al. 2013), in this research the basin shape effect was neglected to make generalizations about bioinfiltration system performance across a range of designs; therefore the 1D basin shape is a model approximation.

The biggest weakness of the 1D approach lies at the interface between a high-permeable layer and a low-permeable layer (which could be a soil with lower permeability or a groundwater table or bedrock). In such a case, a real system will certainly exhibit lateral flow. At a soil interface (such as between a high-permeable fill soil and a low-permeable native soil), this lateral flow must be neglected in a 1D model. Due to this uncertainty, only in situ (unfilled) bioinfiltration sites were considered in this analysis. At the lower boundary, the 1D model is better at representing a system with a deep GWT than a shallow GWT, where the lateral and vertical seepage fluxes are a significant source of uncertainty. Therefore, only bioinfiltration sites above GWT that are deep enough to not influence the surface infiltration were considered. The actual depth at which the GWT will begin to influence the system is dependent on the soil types and the inflow loading. A monitored site at Villanova University with sandy loam soils and an impervious area ratio of about 10:1, for example, shows no influence from the GWT that is located approximately 3 m below the site invert (Nemirovsky et al. 2014). The results of this research will be applicable to sites where the GWT depth, soil type, and design combine such that the GWT does not influence the surface infiltration.

## Soil Hydraulic Properties

The SHP models are generated from the U.S. Department of Agriculture's UNSODA dataset of soil property measurements (Leij et al. 1996; Nemes et al. 2001). To obtain the greatest number of soils from this dataset and to ensure that the soil properties are conservatively representative of field-scale properties (i.e., not skewed by high-permeability macropore-dominated samples), the methods from Lee et al. (2015) are used to filter the data and estimate parameters for the modified van Genuchten–Mualem SHP model [Eq. (1); Ippisch et al. 2006]. The modified van Genuchten–Mualem model is given by

$$S_e = \frac{\theta - \theta_r}{\theta_s - \theta_r} = \begin{cases} \frac{1}{S_c} \cdot [1 + (\alpha\text{VGM}|h|)^n]^{-m} & h < h_e \\ 1 & h \geq h_e \end{cases} \quad (1a)$$

$$K_r = \begin{cases} S_e^{\tau} \cdot \left\{ \frac{1 - [1 - (S_e S_c)^{1/m}]^m}{1 - (1 - S_c^{1/m})^m} \right\}^2 & S_e < 1 \\ 1 & S_e \geq 1 \end{cases} \quad (1b)$$

where $S_e$ = effective degree of saturation [dimensionless, 0 to 1]; $\theta$ = water content [L/L, $\theta_r$ to $\theta_s$]; $\theta_s$ and $\theta_r$ = saturated and residual water contents respectively, $S_c = [1 + (\alpha\text{VGM}|h_e|)^n]^{-m}$ and $h_e$ [L, less than zero indicating suction] is a model parameter related to the air-entry pressure; $\alpha$VGM = model fit parameter [1/L]; $h$ = pressure head [L, negative in suction]; $n$ = model fit parameter [dimensionless, greater than 1], $m = 1 - 1/n$; $K_r$ = relative hydraulic conductivity [dimensionless, 0 to 1]; and $\tau$ = model fit parameter [dimensionless].

In summary, the soil data are filtered by saturated hydraulic conductivity ($K_s$) based on the data of Rawls et al. (1998) to remove data with very high $K_s$ that are dominated by macropore flow. The removal of macropore-dominated samples is a procedure that makes the results of this research conservative in the absence of knowledge about the macropore structure at a bioretention site; if macropores are present at a site, it could only increase the infiltration capacity, however knowledge of macropore structure is generally not known a priori. Additionally, the removal of macropore-dominated soils allows the conductivity curve to be normalized to $K_s$ resulting in SHP models that have appropriate transitions in variably saturated flow [Eq. (2)]. The remaining data are parameterized with van Genuchten (1980) soil water characteristic curve fit parameters ($\theta_s$, $\theta_r$, $\alpha$VGM, and $n$), and then the hydraulic conductivity curve is normalized to $K_s$ and the air-entry parameter $h_e$ is set to −0.1 cm. The $h_e = -0.1$ cm parameter greatly increases the accuracy of the van Genuchten–Mualem (Mualem 1976) model for many fine-textured soils, while behaving like the van Genuchten–Mualem for coarse-textured soils. Full details of the SHP model development and behavior are described in Lee at al. (2015)

$$K = K_s K_r \quad (2)$$

The result of this filtering process was a dataset of 296 soils. Unfortunately, there are not enough soils in each of the 12 USDA textural classes to use them all for this analysis, so the textural classes were combined into the eight *textural groups* presented in Table 1. This is an acceptable simplification because the 12 textural classes have been shown to not optimally describe hydraulic properties anyway (Twarakavi et al. 2010). The hydrologic soil group (HSG) is determined for each soil in the database by comparing the measured saturated hydraulic conductivity to the limits provided by the National Resources Conservation Service (NRCS 2009): $K_s \geq 86.6$ cm/d is HSG-A; else $K_s \geq 34.7$ cm/d is HSG-B; else $K_s \geq 3.7$ cm/d is HSG-C; else ($K_s < 3.7$) the soil is HSG-D. This is the NRCS recommended method for determining the HSG of soils that have at least a 1 m distance between the soil surface and either the GWT or an impermeable layer. All of the bioinfiltration simulations in this study satisfy that requirement.

**Table 1.** Soil Textural Groups

| Textural group | Textural classes within group | Total number of soils in group |
|---|---|---|
| Clayey | Clay | 25 |
| | Silty clay | |
| | Sandy clay | |
| Sandy clay loam | Sandy clay loam | 16 |
| Moderately clayey | Clay loam | 20 |
| | Silty clay loam | |
| Silty | Silt | 64 |
| | Silt loam | |
| Loam | Loam | 24 |
| Sandy loam | Sandy loam | 52 |
| Loamy sand | Loamy sand | 29 |
| Sand | Sand | 66 |

### *Simulation Parameters*

#### Inflow Volume

The bioinfiltration simulation is configured to estimate the performance during a typical *design storm*. Although the design volume is highly variable depending on the location and design goals, it is assumed that the design storm volume will always exceed the basin depth (the volumes in this study are normalized to the bioinfiltration area); otherwise, the solution would be trivial (100% capture in each simulation). In addition, the simulated volume needs to not be so high that it would inundate a highly permeable system, which would be an unrealistic result. And finally, the simulated volume should be the same for all of the performed simulations; otherwise comparisons of the hydrologic capture would be muddled by multicollinearity between soil group and inflow volume. With those considerations in mind, the inflow volume simulated in this study was set to the bioinfiltration pond depth plus 60 cm. This value was selected to be on the high end of the "bioretention abstraction volume" proposed by Davis et al. (2012), which is an approximation of soil storage that might be considered in bioretention design.

#### Initial Conditions

For an infiltration simulation the initial soil moisture condition (SMC) plays an important role in the infiltration rates and thus the hydrologic capacity. Unfortunately, the SMC is a highly variable state and difficult to define without being overly conservative or general. Qualitatively, the most appropriate initial SMC to use for a design simulation is an *expected* or *typical* initial SMC for the site, which would vary depending on the site design, soils, and regional hydrology. It might be beneficial to err on the side of conservative (i.e., higher water content and lower infiltration capacity), however, being too conservative ultimately leads to nearly saturated soils and an underestimation of the typical performance. A continuous simulation with a good soil moisture accounting model would be effective at estimating the expected site performance, however that technique was not used here because of the desire to have simple parameters, small runtimes, and general hydrology (rather than region-specific).

To estimate the expected initial SMC, this method introduces an initial primer event, followed by a period of drainage and evapotranspiration (ET), followed by the primary event for hydrologic assessment. Prior to the initial primer event, the soil is initialized to equilibrium with a deep GWT: 8 and 10 m were used but the results were not sensitive to this difference. Because the primer event was chosen conservatively to represent a storm that exceeds the design conditions, the soil moisture state immediately following the primer event is generally inundated and thus the conditions at the primary event are relatively insensitive to the initial conditions selected for the primer event; this result was confirmed through exploratory studies.

Because rainfall hydrology varies significantly across the country, it is necessary to simulate a range of rainfall patterns to evaluate the effect of this initial SMC on bioinfiltration site performance. Specifically, the necessary parameters are the typical duration (thus, intensity) of an event and the typical event recurrence interval (expected time between events). Rainfall volumes are not needed because they are subsumed into the inflow volume which is predetermined. The primary source of data for the rainfall statistics is Driscoll et al. (1989), who provide the mean and coefficient of variation for regional event duration ($d$) and event recurrence interval ($\delta$) for events with at least 0.25 cm of rainfall separated by an interevent time of 6 h for 15 regions across the continental United States. Although somewhat dated, it is expected that rainfall patterns have not changed too significantly over the past quarter-century. To reduce the number of simulation parameters and to evaluate their effects independently, $d$ and $\delta$ are varied over three levels each rather than simulating all 15 regions. The levels analyzed are $d = 7$, 10, or 14 h and $\delta = 5$, 7, or 12 days, which correspond approximately to the 5, 50, and 95% percentiles for the 15 regions with the exception of $\delta = 12$ days, which is only the 75% percentile (the 95% is 24 days which seems unconservative for design purposes). Because the Driscoll et al. (1989) statistics are for all events of at least 0.25 cm, they might not be exactly equal to design-storm-only statistics; however, analyzing the parameter levels rather than regional averages allows for flexibility where this is a design issue.

To fully specify the storm events and simulation initial condition, the storm event hyetograph and the ET characteristics need to be specified. Although rainfall design hyetographs vary by region, for this analysis the hyetograph was approximated using a symmetric triangular shape, which has higher peak intensity than a constant-rate rainfall. The effect of variations in hyetograph shape were not explored in this analysis because the primary research goal is to evaluate the effect of SHP variation; although the hyetograph shape will affect site performance, it is being assumed that hyetograph variation would not substantially alter the conclusions pertaining to SHP variation. To simplify integration into SWAP, the triangular hyetograph shape is discretized into 1-hr constant-intensity increments.

SWAP offers many options for ET modeling. Because these simulations are general in nature, it is desired only to use a simple ET model. Therefore, a constant potential ET ($ET_0$) rate is supplied at different levels of $ET_0 = 1$, 4, or 7 mm/d. These values are based off of typical values of reference $ET_0$ from Allen et al. (1998), and span some of the expected values seen for bioretention systems in literature (e.g., Wadzuk et al. 2014). The actual ET rate is computed by SWAP based on water availability at the surface and in the root zone. While not expected to be a significant influence on hydrologic performance, the root depth was varied between 30 and 60 cm to assess whether it plays any role in bioinfiltration design performance.

#### Upper Boundary Condition and Surface Water Model

The upper surface boundary condition is either a flux boundary or fixed-head depending on whether the basin is unponded or ponded, respectively. The flux boundary may be governed by inflow or evaporation, and may switch to a fixed-head condition if the inflow flux exceeds the infiltration capacity of the upper surface. This boundary condition switching is a difficult numerical implementation that SWAP handles well.

The surface water balance is governed by an equation that relates the runoff rate to the ponded depth, allowing for a critical depth below which no runoff occurs (the pond depth). This equation may in general be correlated to an overflow weir, but for these simulations the parameters are set to maximize the overflow rate once the basin depth is exceeded. This approximation conservatively estimates the amount of capture of the bioinfiltration system; if the overflow is more restricted, it would allow for additional capture due to infiltration. As simulated, the overflow occurs almost instantly and the pond depth does not exceed the overflow depth by any substantial amount.

#### Lower Boundary Condition

As previously discussed, there is no accurate way to model a GWT boundary in 1D. It is not a fixed-head (zero pressure) boundary because it has been shown that mounding can occur beneath a bioinfiltration basin (e.g., Nemirovsky et al. 2014), indicating that the pressure may exceed zero at the lower boundary. Because the GWT itself cannot be modeled accurately, the lower boundary condition for these simulations is a unit-gradient lower boundary condition,

**Table 2.** Simulation Input Parameters and Their Levels

| Parameter | Levels |
|---|---|
| Native soil group | Textural groups or HSG |
| Inflow volume | Pond depth +60 cm |
| Pond depth (cm) | $D_p = 15$, 30, and 45 |
| Rainfall intensity (h) | $d = 7$, 10, and 14 |
| Rainfall frequency (days) | $\delta = 5$, 7, and 12 |
| Average potential ET (mm/d) | $ET_0 = 1$, 4, and 7 |
| Root depth (cm) | 30 and 60 |

representing a GWT that is deep enough to not influence the simulation model (McCord 1991). This condition sets the seepage rate equal to the hydraulic conductivity of the lowest discrete element, which is located 2 m below the pond invert for these simulations (the GWT is assumed to be deeper than 2 m).

**Numerical Solution Parameters**

The SWAP default convergence criteria were used for these simulations. The minimum timestep was set to the SWAP minimum of $10^{-7}$ days, and the maximum allowable timestep was set to 0.01 days. The maximum number of iterations and backtrack cycles allowed were set to the SWAP maximums (100 and 10, respectively). Following the advice of van Dam and Feddes (2000), a node spacing of 1 cm was applied for the upper 10 cm, otherwise a node spacing of 2.5 cm was used. Exploratory studies confirmed the appropriateness of the node spacing.

### *Design of Monte Carlo Analysis*

To evaluate the effect of variable SHP on bioinfiltration hydrologic capture, a Monte Carlo analysis was performed using the computational model of a bioinfiltration system. The assumption of this analysis was that the SHP models obtained from the UNSODA database represent the full distribution of SHP that would be expected in bioinfiltration systems. If so, an individual SHP model can be randomly selected (with replacement) from the soil group being analyzed for each individual simulation; and once many such simulations are conducted, the overall performance of each soil group can be estimated and compared. However, in such a model the interaction between the soil properties and other parameters may be important. Therefore an even greater number of simulations was required to capture all of the effects on the bioinfiltration performance. Table 2 shows all of the simulation input parameters and their levels.

To analyze all parameter combinations once, 1,296 simulations would be needed. However, to properly represent the within-class variation of SHP, a large number of replicate analyses were needed for each parameter combination to fully represent the distribution of SHP within each soil group. A total of 93,312 simulations were run resulting in 72 replicates for each of the parameter combinations. That number was chosen in consideration for some other interactions which were explored but are not reported here. The CPU time of most simulations was less than 10 s using a standard desktop computer.

## Results and Discussion

The primary output variable from the simulations is the capture volume, $V$, equal to the inflow volume minus the overflow volume. Of particular interest is whether bioinfiltration systems can reliably capture more than the surface storage volume; that is, whether the volume infiltrated during the event is significantly different from zero. Therefore, the capture volume $V$ was modeled as the sum of the pond depth $D_p$ and the *additional capture volume*, denoted $V^+$ [Eq. (3)]. In general, a site with $V^+$ much greater than zero will not need an underdrain, and a site with $V^+$ less than zero will certainly need an underdrain ($V^+$ may be less than zero due to the initial conditions of these simulations: the primer event may not have completely drained resulting in a capture volume less than the full surface storage). Near $V^+ = 0$, the necessity of an underdrain will depend on the allowable ponding time so this relationship will be explored. Because $D_p$ is an independent parameter, only $V^+$ and not $V$ was modeled as a function of the simulation parameters. Fig. 3 shows some of the simulation results, highlighting a strong influence of $K_s$ on $V^+$, and also showing the difference between HSG and textural groups as predictors for $V^+$; the HSG clearly follow the trend whereas the textural groups are muddled together

$$V = D_p + V^+ \tag{3}$$

To analyze the simulation results, two data models were developed to estimate the value of $V^+$ for different soil proxies, one for the textural groups (TG) and another for the HSG (NRCS 2009), shown as two possibilities in Eq. (4):

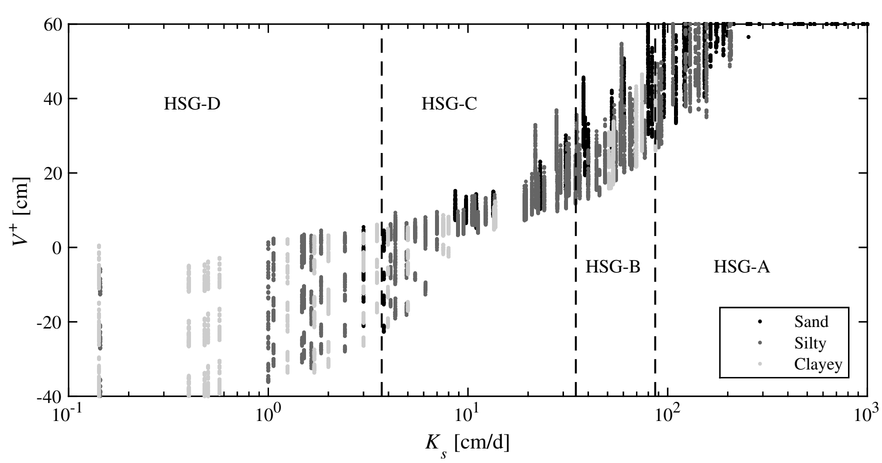


**Fig. 3.** Simulation results by textural group (only selected groups shown for clarity), highlighting the dependence of $V^+$ on $K_s$ and HSG

$$V^+ = \alpha_d + \alpha_\delta + \alpha_{ET} + \alpha_{Dp} + \begin{cases} \beta_{TG} + \varepsilon_{TG} \\ \text{or} \\ \beta_{HSG} + \varepsilon_{HSG} \end{cases} \quad (4)$$

The models for $V^+$ follow the form of an analysis of variance (ANOVA) model: the parameters $\alpha_i$ and $\beta_i$ are the mean effects from each independent variable, and the model residual is denoted $\varepsilon_i$ (the root-depth term is excluded from the model because it has negligible effect on the result). However, the ANOVA model assumptions of independent, identically distributed normal residuals are not met; therefore the ANOVA F-test cannot be used. These assumptions do not affect the computation of the mean effects or the model $R^2$, which may still be computed using the ANOVA technique. The $R^2$ for these models quantifies how precisely the mean effects ($\alpha$ and $\beta$) account for the observed variation in hydrologic performance (specifically, $V^+$). An $R^2$ close to 1 would indicate that the mean effects are reliable proxies for estimating the hydrologic performance, and a smaller $R^2$ would indicate that the within-group variation is very large compared to the mean effects, indicating that the mean effects are poor proxies.

The $\alpha$ and $\beta$ terms are denoted using a different letter to distinguish between terms that will be renormalized to be greater than or equal to zero ($\alpha$) and terms that may be positive or negative ($\beta$). This was done so that the environmental parameters may be neglected if desired (set equal to zero). The $\beta$ term may be less than zero, however, depending on the soil group; for example, HSG-D soils may capture less volume per event than the pond depth due to lack of drainage, resulting in a negative $V^+$. The model residual $\varepsilon$ is also renormalized so that the results may be applied conservatively. That is, rather than the residual having a mean of zero as is typical, the residual is renormalized so that it equals zero at the 5% percentile. Therefore, only 5% of soils within a group will have $\varepsilon$ less than zero; this was done so that $\varepsilon$ can be neglected in general and it can be assumed that the within-class SHP variability will result in a positive benefit to $V^+$, making the computed model parameters a conservative proxy.

It was observed that the $\alpha$ values have negligible difference between the textural group and HSG models (no differences greater than 1 mm), so only the average values of the two models are presented (Table 3). There is a mean effect of $D_p$ despite the fact that $V^+$ is already being added to $D_p$; this makes sense as deeper ponds take longer to drain and will generally have less infiltration capacity at the next event than shallow ponds. The rainfall intensity parameter $\alpha_d$ shows that longer events will have greater time for infiltration and thus, a higher $V^+$. Rainfall frequency ($\alpha_\delta$) also affects the infiltration capture because less frequent events will have more time to dry out, and thus, a greater infiltration capacity. And finally, the potential ET plays a minor role in that higher $ET_0$ may free up space for more infiltration. While ET plays a minor role in this design-storm capture, it should not be confused with the role of ET in an overall water budget for a bioinfiltration site: ET is more meaningful in the context of an annual water budget (Wadzuk et al. 2014).

**Table 3.** Average $\alpha$ Parameter Estimates for the Mean Effects Model Eq. (4)

| Parameter | Level | Estimate (cm) |
|---|---|---|
| $\alpha_{Dp}$ (cm) | 45 | 0.0 |
| | 30 | 2.6 |
| | 15 | 4.2 |
| $\alpha_d$ (h) | 7 | 0.0 |
| | 10 | 2.6 |
| | 14 | 5.3 |
| $\alpha_\delta$ (d) | 5 | 0.0 |
| | 7 | 1.1 |
| | 12 | 3.0 |
| $\alpha_{ET}$ (mm/d) | 1 | 0.0 |
| | 4 | 0.7 |
| | 7 | 1.3 |

**Table 4.** $\beta$ and $\varepsilon$ Parameter Estimates for the Mean Effects Model Eq. (4) for the TG and HSG

| Parameter | Level | Estimate (cm) TG | HSG |
|---|---|---|---|
| $\beta_{TG}$ | Sand | 6 | — |
| | Loamy sand | 3 | — |
| | Sandy loam | −16 | — |
| | Loam | −24 | — |
| | Silty | −26 | — |
| | Sandy clay loam | −45 | — |
| | Clayey | −41 | — |
| | Moderately clayey | −30 | — |
| $\beta_{HSG}$ | HSG-A | — | 30 |
| | HSG-B | — | 3 |
| | HSG-C | — | −16 |
| | HSG-D | — | −41 |
| $\varepsilon$ | 1% | −10 | −6 |
| | 5% | 0 | 0 |
| | 50% | 39 | 19 |
| | 95% | 71 | 31 |
| | 99% | 80 | 37 |

The estimates of $\beta$ and $\varepsilon$ for the textural groups and HSG models are presented in Table 4. Overall, the textural group model has an $R^2 = 0.40$ with a root-mean-square residual equal to 21.4 cm while the HSG model has an $R^2 = 0.89$ with a root-mean-square residual equal to 9.0 cm. From those overall numbers, it can be seen that the HSG is a much better proxy for explaining the hydrologic performance of bioinfiltration systems than are the textural groups. Because the residuals are much larger in the textural group model, the group means are necessarily more conservative than the HSG model to maintain the conservative renormalization of the $\alpha$ and $\varepsilon$ parameter values. The main conclusion for bioinfiltration sizing is that HSG-A soils have a substantial infiltration capture—30 cm or more—which should be accounted for in the site design so that the sites are not oversized. Additionally, HSG-B soils and the sand and loamy sand textural classes have a positive minimum infiltration capture ranging between 3 cm and almost 20 cm depending on the $\alpha$-terms. The actual performance can be anywhere between the minimum values computed by the model and up to $\varepsilon_{95\%}$ additional, which may be another 30–70 cm depending on the model. For the aforementioned groups with $\beta > 0$, an underdrain will not be needed in most conditions, especially where environmental conditions increase the capture potential ($\alpha > 0$). Fig. 4 presents the within-group variation of simulation results, after adjusting the data by subtracting off the $\alpha$ mean-effects. Therefore, Fig. 4 represents performance expectations that are conservative among the simulated $\alpha$ parameters. In Fig. 4, the black dots represent the middle 90% of data, the vertical bars are the middle 80%, and the box represents the middle 50% of data, with the median at the vertical bar in the box.

For the many soil groups with $\beta$ less than zero, it should not automatically be assumed that underdrains are needed. First, consider the sum of $\beta$ and $\varepsilon$. For example, $\beta_{HSG\text{-}C} = -16$ cm, but $\varepsilon_{HSG,50\%} = 19$ cm. Adding these two numbers shows that more than half of HSG-C soils have a $V^+$ greater than zero indicating that

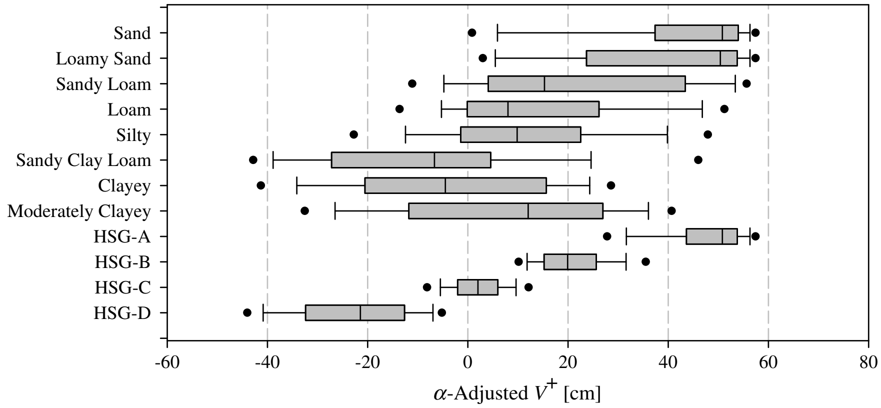


**Fig. 4.** The $V^+$ distributions for each soil group adjusted such that all $\alpha$-terms are subtracted off; showing the middle 90% of data (black dots), 80% of data (lines), 50% of data (box), and median (vertical bar in box)

**Table 5.** Percentage of All Simulations with Different Surface Drawdown Times

| Surface drawdown time | Minimum $V^+$ (cm) | | | |
|---|---|---|---|---|
| | 0 | 2.5 | 5 | 10 |
| <2 (days) | 80% | 82% | 88% | 96% |
| <3 (days) | 87% | 90% | 94% | 99% |
| <4 (days) | 91% | 94% | 97% | 99% |

an underdrain may not be needed; so using the negative $\beta$ as a conclusive proxy in general is not recommended. An exception would be HSG-D soils, where $\beta_{\text{HSG-D}} = -41$ cm and $\varepsilon_{\text{HSG},99\%} = 37$ cm, showing that nearly all HSG-D soils can be expected to have an overall $V^+$ less than zero, indicating the need for an underdrain in all cases. Due to the lack of precision of the textural group model, the same conclusion cannot be reached in general for any of the textural groups. These conclusions may also be observed in Fig. 4.

To verify that $V^+ > 0$ indicates an acceptable design without underdrains, the ponding time must also be considered. For example, with $\delta = 12$ days, a pond might dewater in 10 days and still have a $V^+$ slightly above zero, but this design would need an underdrain and fill media due to unacceptably long ponding time. Table 5 shows the percentages of all simulations that dewater within a few days, given a minimum observed $V^+$. Considering that these simulations represent a fairly conservative design storm, drawdown times in excess of two days may often be acceptable, as the average drawdown time for the site will be less. Based on Table 5, a $V^+$ above 5 cm indicates reasonable confidence for a design that does not require an underdrain, as approximately 95% of simulations with $V^+ > 5$ cm dewater in less than three days after the back-to-back design storms. Soil groups that always exceed this are sand and HSG-A; loamy sand and HSG-B are very close and will generally exceed $V^+ = 5$ cm when any of the $\alpha$-terms are added to the model; for example, loamy sand and HSG-B will have $V^+ > 5$ cm for any of the evaluated weather conditions if the basin depth is limited to 30 cm, which adds $\alpha = 2.6$ to the minimum $V^+$.

## Application

Because the $V^+$ model was constructed to be a minimum expected capture from a soil group for a typical design storm, these proxies can be used as sizing guidelines for bioinfiltration systems designed over nonrestrictive GWT. For example, consider a bioinfiltration system design in the Pacific Northwest, with a loamy sand ($\beta_{\text{TG,loamy sand}} = 3$ cm) soil. It might be assumed that the rainfall intensity will be low, the frequency will be high, and the potential ET will be low, corresponding to $\alpha_d = 5.3$ cm, $\alpha_\delta = 0$ cm, and $\alpha_{\text{ET}} = 0$ cm. If a shallow (15 cm) pond is desired, then $\alpha_{Dp} = 4.2$ cm and the expected capture can be estimated as

$$V = D_p + \alpha_d + \alpha_\delta + \alpha_{\text{ET}} + \alpha_{Dp} + \beta_{\text{TG,loamy sand}} \tag{5}$$

$$\therefore V = 15\text{ cm} + 5.3\text{ cm} + 0 + 0 + 4.2\text{ cm} + 3\text{ cm} = 27.5\text{ cm}$$

From the result in Eq. (5), it can be seen that the 15 cm in situ bioinfiltration cell can be sized such that it captures 27.5 cm averaged over the ponded area; quite a significant increase over 15 cm. The SHP variation within the loamy sand class would only serve to increase the capture for 95% of soils within the class. If it were known that in addition to being a loamy sand, the soil were HSG-A, then instead of using $\beta_{\text{TG,loamys and}} = 3$ cm, the model would use $\beta_{\text{HSG-A}} = 30$ cm, and the design capture volume would be approximately 55 cm in the 15 cm basin. Such large infiltrated volumes should certainly be accounted for in bioinfiltration system designs.

## Limitations

There are a few important model approximations that limit the interpretation of results. Affecting the results substantially is the assumption that there is no air entrapment so the soil may always become completely saturated. It is known that infiltrating water may not truly saturate the soil due to the presence of entrapped air that cannot exit the system. In this case, the effective hydraulic conductivity may be significantly less than the laboratory-measured saturated hydraulic conductivity. However in the model, the soil often becomes completely saturated and so the infiltration and seepage rates match the measured saturated hydraulic conductivity. This could be a significant source of unconservatism in the model. To account for this effect a factor of safety could be applied to positive values of $V^+$. For example, if it is estimated that only 50% of $K_s$ is reached, then a positive $V^+$ might be reduced by 50% to approximate this limitation. Additional limitations, which have already been discussed are (1) the soil properties are homogeneous; (2) the GWT does not influence the infiltration; (3) the basin has a

1D shape; and (4) the SHP from the UNSODA database represent the population of bioinfiltration soils.

In addition to the model limitations, it is being assumed that there is no clogging from deposition of fines near the surface that would limit the infiltration; site maintenance may be required to achieve this assumption in some situations (Brown and Hunt 2012). However, at least one bioinfiltration site has been reported to operate without loss of infiltration capacity for over a decade without maintenance (Gilbert Jenkens et al. 2010). Clogging due to fines deposition is an issue that affects both in situ and filled bioretention sites, however. Also, for optimum performance of bioinfiltration sites, construction best practices should be followed so that postconstruction performance is on par with preconstruction expectations (Brown and Hunt 2010). Particular attention should be paid to soil compaction at the construction site.

## Conclusion

A dataset of individual soil hydraulic properties was used to evaluate the variability of soil textural groups and HSG using simulations of an in situ bioinfiltration system performed with SWAP, a 1D Richard's equation model. For each simulation, the expected capture for a design storm was predicted by priming the soil with a rainfall event and then allowing a drainage and evapotranspiration period before the design event. This allowed for realistic estimates of the initial soil moisture condition without being overly conservative. The interaction between the soil properties, the environmental characteristics, and the pond depth were evaluated by conducting a large number of simulations.

The simulation results were modeled showing how the soil textural classes and the HSG perform as proxies for in situ bioinfiltration performance. Although there are certainly differences between the mean performance of the textural groups, the difference in means is not large compared to the within-group variation ($R^2 = 0.40$) making soil textural class a relatively poor proxy for bioinfiltration site performance. Better at predicting the hydrologic performance are the HSG, with a predictive $R^2 = 0.89$. Intuitively this makes sense, because various levels of compaction and soil grain shape can heavily affect the hydraulic conductivity of soils within the same class; the HSG has accounted for this by directly using the measured saturated hydraulic conductivity rather than inferring the conductivity through the grain-size distribution. By comparing these two models and the $\alpha$ and $\beta$ parameter estimates, it can be seen that soil properties are by far the largest influence on bioinfiltration system performance of the three main input groups seen in Fig. 1. Next important are the environmental characteristics, and finally the pond depth; although the importance of pond depth is much increased when estimating the total volume capture and drawdown time rather than the infiltration design capture ($V^+$). This highlights a key result of this study: that if the soil hydraulic behavior is not well known, then changing the design characteristics will not salvage a bioinfiltration site. The contrapositive form of this conclusion is: if the soil is known to infiltrate, then the bioinfiltration system will work, regardless of the soil textural classification.

Models for predicting the additional capture volume of the site ($V^+$) were presented using both the textural groups and HSG as a proxy. These models may be used to help size bioinfiltration systems and evaluate the need for an underdrain, determined if $V^+$ is small or negative, although the usefulness of the models is limited to certain applications due to the high in-class variability of soil hydraulic properties. Although the models lack precision due to the large within-class variation of SHP, some useful conclusions were still drawn. In particular, for sizing,

- HSG-A native soils can be expected to capture at least an additional 30 cm of water in excess of the mean pond depth, without the addition of any fill media;
- HSG-B, sand, and loamy sand native soils will have minimum capture of a few centimeters in addition to the mean pond depth, but potentially much higher minimum capture if the $\alpha$ terms are included Eq. (4), without the addition of fill media;

and for underdrains,

- HSG-A and sand native soils do not need underdrains or fill media for all site designs explored in this research;
- HSG-B and loamy sand native soils can always be designed such that neither an underdrain nor fill media are needed ($D_p <= 30$ cm will satisfy this condition); and
- HSG-D native soils will always require an underdrain with engineered fill media to avoid continuous saturation.

Although the models may be used in some other applications, often the transition from needing an underdrain to not needing one will land somewhere in the distribution of soil properties within the class. For example, the authors do not conclude that clayey textures always need fill media and underdrains. This transition is visible in HSG-C soils in Fig. 3, near $K_s = 10$ cm/d, but if $K_s$ is not known then it is not possible to determine whether a soil in these transition classes requires an underdrain. Based on these results, it is recommended that in situ soils be tested for grain size distribution and plasticity and also for infiltration rate and/or saturated hydraulic conductivity with a method such as the modified Philip-Dunne infiltrometer (Asleson et al. 2009) prior to the site design, using good practices with multiple samples (Weiss and Gulliver 2015); care should be taken to ensure that soil tests accurately reflect the effective properties of the bioinfiltration site. The soil tests combined with the results of this study may be used to determine the necessity of an underdrain and fill media, and also the potential for sizing the site to capture the infiltrated volume in addition to the surface storage volume. Better sizing and utilization of in situ soils for infiltration can lead to significantly reduced costs and more sustainable construction methods for stormwater green infrastructure.

These models were developed to be conservative estimates of the hydrologic capture of in situ bioinfiltration sites; the 1D model is conservative for this application as long as there are no restrictions to drainage. The extreme 5% of soil properties will violate the model estimates, but this is often an acceptable risk for engineering designs. The primary limitation and unconservatism of these analyses is that the model assumes that saturated conditions are reached while infiltrating water at the site; in reality, air entrapment and hysteresis in the wetting and drying of the site may cause this to be violated, and then the saturated hydraulic conductivity ($K_s$) may not be achieved. Certain infiltration measurement techniques may account for this last effect, however, which reinforces the main conclusion of this study: knowing the actual, effective saturated hydraulic conductivity and/or infiltration rate at the bioinfiltration site is the paramount design consideration, far exceeding textural assessment or other design considerations.

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## Notation

*The following symbols are used in this paper:*

$D_p$ = depth of ponding prior to overflow, also the surface storage volume (L);
$d$ = duration of a storm event (T);
$ET_0$ = potential evaoptranspiration (L/T);
$h$ = pressure head, negative in suction (L);
$h_e$ = air-entry parameter in modified van Genuchten −Mualem model (L);
$K$ = hydraulic conductivity (L/T);
$K_0$ = hydraulic conductivity curve normalization constant (-);
$K_s$ = saturated hydraulic conductivity (L/T);
$m$ = van Genuchten−Mualem model parameter (typically equal to $1 - 1/n$) (−);
$n$ = van Genuchten−Mualem model parameter (−);
$S_c$ = effective degree of saturation when $h = h_e$ (−);
$S_e$ = effective degree of saturation (−);
$V$ = volume captured by a bioinfiltration cell (L);
$V^+$ = additional volume captured (in addition to the surface storage volume, $D_p$) (L);
$\alpha_P$ = hydrologic model fit parameter, where $P$ = parameter name (L);
$\alpha$VGM = van Genuchten−Mualem model parameter ($1/L$);
$\beta_i$ = hydrologic model fit parameter for soil class, either $i$ = TG or $i$ = HSG (L);
$\delta$ = time between storm events (T);
$\varepsilon_i$ = hydrologic model residual, for $i$ = TG or $i$ = HSG (L);
$\theta$ = soil moisture content (L/L);
$\theta_r$ = residual soil moisture content (van Genuchten−Mualem model parameter) (L/L);
$\theta_s$ = saturated soil moisture content (van Genuchten −Mualem model parameter) (L/L); and
$\tau$ = van Genuchten−Mualem model parameter (−).

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