



Assessing and Improving Pollution Prevention by Swales

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Final Report

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Executive Summary

Roadside swales are drainage ditches that also treat runoff to improve water quality, including infiltration of water to reduce pollutant load. Since most roadside drainage ditches infiltrate water into the soil, filter sediments and associated pollutants out of the water, and settle solids to the bottom of the swale, there is little distinction between the two. Currently, however, there is little information that can be used to gain pollution prevention credits for these systems to meet state and watershed district requirements. The objectives of this research project are to investigate and document the infiltration potential of swales and investigate methods to improve pollution prevention by swales that may not be infiltrating much of the runoff.

In the infiltration study, a quick and simple device, the Modified Philip Dunne (MPD) infiltrometer, was utilized to measure an important infiltration parameter (saturated hydraulic conductivity, K_{sat}) at multiple locations in a number of swales. The study showed that the spatial variability in the swale infiltration rate was substantial. Thus, 20 or more measurements along the highway are recommended to get a good estimate of the mean swale infiltration rate. A runoff-flow routing model was developed that uses the K_{sat} values, soil suction, soil moisture content and swale geometry to estimate a volume of infiltration loss and runoff for a given rainfall event. This model was verified by comparing the predicted runoff volume with the actual runoff volume of a selected swale. Investigation was also performed on the impact of a given saturated hydraulic conductivity on the Natural Resources Conservation Service (NRCS) curve number (CN), which is often used to model runoff in or near a roadside swale.

This research study also developed a ditch check filtration system that can be installed in swales to provide significant treatment of dissolved heavy metals and dissolved phosphorous in stormwater runoff. First, batch studies were conducted to identify the filter media enhancement for effective removal of phosphorus and metals from water. Pilot tests were then conducted to test the pollutant removal performance of a ditch check filter with enhanced filter media. The results were utilized to develop design guidelines and recommendations, including sizing and treatment criteria for optimal performance of the full-scale design of these filters. In addition, a model was developed to predict the reduction of phosphorus and heavy metals that can be achieved with the installation of a ditch check filter.

Finally, the best available knowledge on swale maintenance was combined with information obtained from new surveys conducted to develop recommendations for swale maintenance schedules and effort. The recommendations aim toward optimizing the cost-effectiveness of roadside swales and thus provide useful information to managers and practitioners of roadways.

The research results and information obtained from this study can thus be used to design swale systems for use along linear roadway projects that will receive pollution prevention credits for infiltration. This will enable the utilization of drainage ditches to their full pollution prevention

potential, before building other more expensive stormwater treatment practices throughout Minnesota and the United States.

I. Introduction

Development and urbanization of watersheds typically increases impervious land cover (e.g. roads, parking lots, buildings) and thus leads to an increase in stormwater runoff peak flow rates, total runoff volume, and degradation of runoff water quality. The degradation in water quality occurs due to the increased pollutant concentrations in the runoff that result in increased pollutant loads to receiving waters. Currently the pollutants of greatest concern in stormwater are total suspended solids (TSS), phosphorus (P), nitrogen (N), cadmium (Cd), copper (Cu), lead (Pb), and zinc (Zn).

Historically, stormwater management focused on flood prevention through mitigation of the increase in peak flow rate by routing the runoff through a detention pond or some other system. Increased runoff volume and degradation of water quality were typically not addressed. With an increase in environmental awareness and more stringent regulations, however, these issues should be considered as part of most stormwater management plans. As a result, stormwater management has shifted to include techniques that reduce runoff volumes and improve runoff water quality in addition to reducing the peak flow rate. Such techniques are called low impact development (LID) practices and are typically designed to reduce runoff through infiltration and/or evapotranspiration as well as improve stormwater runoff quality by retaining or capturing contaminants in the runoff. LID practices include bioretention facilities, infiltration basins and trenches, constructed wetlands, sand filters, vegetated roadside swales, etc.

Vegetated roadside swales serve three purposes: removing water during rainfall-runoff events, infiltrating water into the soil, and filtering the solids and associated pollutants from the water. As with infiltration trenches and basins, grassed swales can become clogged with particles and debris in the absence of proper pretreatment and maintenance. In some cases the sediment deposits can begin to choke out the vegetated cover and create an erodible surface capable of contributing sediment and other pollutants directly downstream (Weiss et al. 2005). If requirements for infiltration and pollution remediation exist for runoff from roads, one consideration should be to consider the infiltration of drainage ditches and to re-classify them as grassed swales.

Several studies have shown that a majority of stormwater pollutants are trapped in the upper soil layers (Wigington et al. 1986, Mikkelsen et al. 1997, Dierkens and Geiger 1999). The exception being some pesticides and de-icing salts which pass directly through the soil and eventually reach the groundwater (Mikkelsen et al. 1997; Dierkes and Geiger 1999). New methods of measuring infiltration capacity have recently been developed by Asleson et al. (2009), where a rapid infiltration technique allow multiple measurements to be taken over a relatively large area. This is needed because the infiltration capacity at a given site will have substantial spatial variation.

Dissolved phosphorous and heavy metals are a substantial portion of the total load of these pollutants in runoff (Pitt, et al. 2004), which have not previously been targeted for removal. In order to target the removal of dissolved pollutant fractions from road runoff, enhanced ditch check filters can be placed in the swale. Steel and iron have been shown to be effective at removing dissolved heavy metals (Wang et al. 2003) and dissolved phosphorous (Erickson et al. 2005, 2007, 2012) from water. Sand filtration has been shown to effectively remove particles from stormwater (Winer 2000; Weiss et al. 2005) and filtration practices installed above permeable soils allowing some stormwater runoff to infiltrate and recharge groundwater. By combining these two proven technologies, a ditch check filtration system with enhanced media may be developed to provide significant treatment of particulates, dissolved heavy metals, and dissolved phosphorus.

Maintenance of swales is an integral part of swale performance for infiltration and pollution control. If managers of roads are going to consider re-classifying their drainage ditches to grassed swales, they will need to know the additional maintenance cost and effort required, so that they can make an informed decision. Therefore, maintenance of grassed swales should be examined to estimate the cost and effort required.

Swale infiltration potential, pollution prevention performance, and maintenance of swales were the focus of this research project. The following tasks summarize the objectives of this research study:

1. Perform a detailed, critical literature review of grassed swales as a pollution prevention technique.
2. Determine the effectiveness of several materials as a potential enhancement media for dissolved heavy metals and phosphorus removal.
3. Undertake field measurements of infiltration in swales, designed to obtain sufficient accuracy in swale infiltration to apply along Minnesota roads.
4. Run pilot-scale studies on the ditch check filter developed to remove dissolved heavy metals and phosphorus.
5. Apply the field measurements of infiltration in swales to represent performance of swales.
6. Develop a means of scheduling the maintenance and computing the maintenance cost and effort of grassed swales, compared to drainage ditches.
7. Develop, test and interpret models for both hydrologic analysis of swales and pollutant retention performance of ditch check filters, to be used in the application of the results by practitioners.

In order to achieve the project objectives, the infiltration capacity of five swales are characterized and a runoff-flow routing model developed and verified by actual infiltration data of a swale (tasks 3 and 5). The impact of improving saturated hydraulic conductivity on a watershed is modeled (task 7). Based on available literature, batch studies conducted to

identify potential media enhancing materials (task 2) is described. Pilot tests conducted on a ditch check filtration system with enhanced filter media to remove phosphorus and metals in water are used to develop design guidelines that can be used in a preliminary investigation (tasks 4 and 7). Finally the maintenance, cost and requirements of swales are documented (task 6).

This research will directly benefit those responsible for limiting the runoff of suspended and dissolved contaminants from watersheds via roadside swales. The documented potential of roadside drainage ditches to be re-classified as swales will enable them to be given credit for the pollution prevention that they are already achieving and provide the information necessary to develop design standards for swales along our linear road projects. In addition, the new design standards for ditch check filters with a short contact time that will capture particulates, dissolved heavy metals, and dissolved phosphorus will be applicable to full-scale design of the filters for field installation. The technologies and application models developed herein can be applied to swales or drainage ditches throughout the state of Minnesota and the United States to provide cost-effective treatment of highway runoff. Results can also be used to estimate load reductions due to infiltration in roadside drainage ditches and the installation of enhanced ditch check filters in Total Maximum Daily Load (TMDL) studies for future installations of roadside swales.

II. Literature Review: Performance of Swales for Pollution Prevention (Task 1)

Swales are shallow, flat-bottomed, open vegetated drains/channels/ditches that are designed to convey, filter and infiltrate stormwater runoff (Barrett et al. 1998a; Deletic and Fletcher 2006). Swales are classified into grassed swales, filter strips, and bioswales, and are typically employed along highways. While grassed swales are primarily designed to promote conveyance and infiltration of runoff, bioswales are especially designed to provide water quality treatment of runoff. Filter strips are generally the shoulder of the road that the runoff has to flow over to drain into a roadside swale, and are aligned perpendicular to the direction of flow. Sometimes, highway medians may essentially act as grassed swales (Barrett et al. 1998a and 1998b). [Figure 1](#) shows a roadside drainage ditch on Hwy 51 in MN.



Figure 1. Roadside drainage ditch on Hwy 51 at off ramp to County Road E.

Grassed swales have the capability to reduce runoff volume and improve water quality. Volume reduction occurs primarily through infiltration into the soil, either as the water flows over the slide slope perpendicular to the roadway into the swale or down the length of the swale parallel to the roadway. Pollutant removal can occur by sedimentation of solid particles onto the soil surface, filtration of solid particles by vegetation, or infiltration of dissolved pollutants (with stormwater) into the soil (Abida and Sabourin 2006). When solid particles settle to the soil surface or are captured by filtration on vegetation, the TSS concentration of

the runoff is reduced and overall water quality is improved as long as the solids do not become resuspended. As will be discussed later, such resuspension has been determined to be negligible.

Despite the prevalence of grassed swales that convey and treat road runoff within roadway right-of-ways, data on the performance of swales with regards to infiltration and impact on water quality is relatively sparse. Information from available literature is combined below to identify materials that have the potential to provide significant treatment of particulates, dissolved heavy metals, and dissolved phosphorus in the proposed ditch check filters for swales.

1. Infiltration Performance of Swales

The fraction of stormwater runoff that can be infiltrated by a roadside swale depends on many variables including rainfall intensity and total runoff volume, swale soil type, the maintenance history of the swale, vegetative cover in the swale, swale slope and other design variables, and other factors. Thus, reported swale performances have ranged from less than 10% to 100% volume reduction. For example, using simulated runoff, Yousef et al. (1987) found that swales infiltrated between 9% (input rate of 0.079 m/hr) and 100% of the runoff (at 0.036 m/hr) with significant variability. Due to the wide range of performance and the host of important variables, Yousef et al. (1987) stated that in order to determine the performance of individual swales, each swale should be tested separately. Due to the wide variability of infiltration rates, even within a single swale, multiple measurements should be made. One possible testing method is described later in this report.

In a review of data compiled in the International Stormwater BMP Database, Barrett (2008) found that, if the soil is permeable and the initial moisture content is low, the infiltration achieved by swales can approach 50% of the runoff volume in semiarid regions. Rushton (2001) found that the installation of swales to carry parking lot runoff in Tampa, FL resulted in 30% less runoff.

Lancaster (2005) also measured infiltration along a roadside embankment. At one site in Pullman, Washington 36 precipitation events were monitored and in all events all runoff from the roadway had infiltrated within the first two meters from the edge of pavement. At another site in Spokane, Washington 18 precipitation events were monitored. Of these events, five were observed to have runoff at 10 ft (3.1 m) from the edge of pavement and one event had runoff 20 ft (6.2 m) from the edge of pavement.

Ahearn and Tveten (2008) investigated the performance of 41 year old, unimproved roadside embankments in the State of Washington for their ability to infiltrate and improve the quality of road runoff. Four sites were investigated and at each site monitoring stations were set up at the “edge of pavement” (EOP), 2 m from the EOP, and 4 m from the EOP. Runoff volume reductions were 71% to 89% at 2 m from EOP and 66% to 94% at 4 meters from EOP. The

smaller volume reduction at 4 m was caused by water infiltrating before the 2 m station and re-emerging as surface flow before the 4 m station at two of the monitoring locations.

Deletic (2001) developed, calibrated, and verified a model to estimate runoff generation (and sediment transport). The model, which is discussed in more detail below, was developed, in part, to estimate runoff volumes from grassed swales. A sensitivity analysis showed that the volume of runoff depends primarily on length of the grass strip and soil hydraulic conductivity.

A variable that affects the hydraulic conductivity of a swale is the compaction of the soil within the swale. Gregory et al. (2006) studied the effect of soil compaction on infiltration rates into soils and found that the infiltration capacity of soil decreased as the weight of the heavy equipment driven on the soil increased. Although the results were not determined to be statistically significant with respect to the weight of the vehicle, the study found that soil compacted by a pick-up truck, backhoe, and fully loaded dump truck had infiltration capacities of 6.8 *cm/hr*, 5.9 *cm/hr*, and 2.3 *cm/hr*, respectively. It was also found that that construction equipment or compaction treatment reduced the infiltration capacity of the soil by 70 - 99%. Gregory et al. (2006) concluded that the infiltration capacity of a swale will be greatly reduced if it is subject to heavy equipment loads.

Contrary to the effects of compaction, tree roots have been shown by Bartens et al. (2008) to have the ability to penetrate compact urban soils and increase infiltration. Bartens et al. (2008) studied flooded impoundments and found that those with trees (Black Oak or Red Maple) infiltrated water 2 - 17 times faster than those without trees. Placing a tree within a swale may not be practical, but it is possible that trees near swales could increase infiltration into the surrounding soil. Deep-rooted grasses may have a similar effect to trees.

In order to determine the infiltration capacity of five grassed swales in Canada, Abida and Sabourin (2006) performed single ring, constant head infiltrometer tests on swales that had perforated pipe underdrains. Although these swales had underdrains that collected infiltrated water and removed it from the soil, the drains were far enough below the ground surface that the authors assumed the drains did not affect infiltration rates over the course of the experiment. The investigators found that the infiltration capacity of the swales decreased exponentially with time until it reached a constant, long-term value. Initial infiltration rates varied from 3 to 13 *cm/hr* with constant rates of 1 to 3 *cm/hr* achieved after 15 to 20 minutes.

Jensen (2004) used dye to measure and visualize flow patterns of runoff infiltrating four Danish highway swales. Infiltration capacities of these swales ranged from 5.76 *cm/hr* to 6.84 *cm/hr*, which were similar to capacities of other authors (1.8 *cm/hr* to 7.2 *cm/hr*) that were reviewed by Jensen. The infiltration capacity of the Danish swales was greater than the rate water entered a sub-soil collection pipe. This resulted in a secondary but temporary water table above the collection line. It was also determined that, based on historical Danish rainfall data,

the infiltration capacity of the swales would allow for 75% of the annual road runoff to infiltrate.

Jensen (2004) also found that infiltrated water had to first pass through a homogenous layer of previously deposited sediment on the ground surface before entering the natural underlying soils. The thickness of the sediment layers was measured at three locations on three out of four swales and was estimated to deposit at a rate of 2 - 6 *mm/year* at all but two of the nine measurement locations. Two measurement locations on one swale were estimated to have deposition rates of 10 - 16 *mm/year*. This variability could be caused by differences in traffic volumes, rainfall patterns, or watershed characteristics, etc.

Jensen (2004) found the infiltration through these deposited layers was similar and uniform at each site and concluded that these layers could act as filter media. The swale soil that was placed during roadway construction was found to be homogenous at some locations and heterogeneous at other locations. The preferential flow paths that were observed in some swale media was determined to be a result of human influences such as the construction process and not of earthworms or other natural causes. This is relevant because preferential flow paths in swale media could limit contaminant capture. Thus, if water quality improvement is an objective, severe preferential flow paths should be avoided. Preferential flow paths could increase infiltration, although the potential for groundwater contamination would be increased.

A summary of reviewed swale infiltration studies is provided in [Table 1](#).

Table 1. Summary of swale infiltration studies.

Study	Infiltration Rate (cm/hr)	Volume Reduciton (%)	Comments
Yousef et al. (1987)	--	9 - 100	Simulated rainfall laboratory studies
Rushton (2001)	--	30	Swales to carry parking lot runoff
Jensen (2004)	5.8 - 6.8	75	Estimated annual road runoff reduction
Lancaster (2005)	--	100	Measured at 2 m from EOP
	--	up to 100	Measured at 3.1 & 6.2 m from EOP
Abida and Sabourin (2006)	3 - 13	--	Initial infiltration rates
	1 - 3	--	Long-term infiltration rates
Gregory et al. (2006)	6.8	--	Soil compacted by pick-up truck
	5.9	--	Soil compacted by backhoe
	2.3	--	Soil compacted by loaded dump truck
	(Compaction reduced infiltration 70-99%)		
Ahearn and Tveten (2008)	--	71 - 89	Measured at 2 m from EOP
	--	66 - 94	Measured at 4 m from EOP

Note: EOP = edge of pavement

2. Phosphorus Removal in Swales

This section reviews studies that have assessed the phosphorus removal performance of swales. Observed phosphorus removal rates in swales have ranged from negative values (i.e. the swale is a source of phosphorus) to over 60% removal. Differences can be attributed to many variables, including assessing dissolved phosphorus or total phosphorus removal, season of the year, vegetative cover, and many more. Studies that have assessed swale performance with respect to total suspended solids, nitrogen, and metal removal have also been conducted. For these results and for details on phosphorus removal studies, please see Appendix A.

Barrett (2008) reviewed the International BMP Database and found that swales provided no significant reduction of phosphorus. Barrett (2008) suggested, however, that the removal of grass clippings after mowing could remove nutrients such as phosphorus from the swale and prevent nutrient release upon decomposition.

Barrett et al. (1998b) performed experiments at different water depths on vegetated laboratory swales using synthetic stormwater created to represent typical stormwater runoff pollutant concentrations in the Austin, Texas area. Underdrains were used to collect stormwater that infiltrated into and through the swale. Upon reaching the underdrain the water had infiltrated through the top layer of Buffalo grass, 16 *cm* of topsoil, and 6 *cm* of gravel. Results for total phosphorus are shown in Table 2 (for full results please see Appendix A). The data suggests that phosphorus removal primarily occurs within the first 20 *m* of the swale length.

Table 2. Range of total phosphorus removal efficiencies (%) for a laboratory vegetated swale (from Barrett et al. 1998b).

Distance Along Swale (m)	10	20	30	40	underdrain
Removal efficiency (%)	25-49	33-46	24-67	34-45	55-65

3. Increasing Dissolved Phosphorus Removal in Swales

Stormwater can contain elevated levels of phosphorus and nitrogen Pitt (1996). Sources include fertilizer, animal waste, plant biomass, atmospheric deposition, and motor oil for phosphorus. Dissolved phosphorus and nitrogen are nutrients that have been targeted for removal from stormwater runoff. With regards to algae and plant growth in inland fresh water lakes, rivers, and other water bodies that receive stormwater, phosphorus is often the limiting nutrient. Thus, when phosphorus rich stormwater enters the water body, plant growth, which was previously limited by the lack of phosphorus, can increase rapidly. The result is lakes and ponds with an overabundance of plant matter that, upon decomposition, can deplete the dissolved oxygen in the water body. Phosphorus removal in water treatment has conventionally been accomplished through bacterial uptake and chemical precipitation. The uptake of

phosphorus by bacteria is typically slow and must occur under controlled conditions. Precipitation is usually accomplished through the addition of alum (i.e. aluminum sulfate) to the water body but adding a metal such as aluminum to a water body to remove phosphorus could be more detrimental than beneficial.

Thus, in order to cost-effectively treat large volumes of stormwater for phosphorus removal, a new, passive method or methods that remove dissolved forms of these nutrients would be optimum. The following sections review methods that have been investigated.

3a. Materials for Improving Phosphorus Removal

Dissolved phosphorus removal can occur through adsorption to iron (Fe) and aluminum (Al) hydroxides or oxides under acidic conditions. It has been reported that optimal removal via this mechanism occurs from pH = 5.6 to pH = 7.7 whereas precipitation of phosphate with calcium (Ca) is optimal at pH values of 6 to 8.5 (O'Neill and Davis 2010).

Iron has demonstrated the ability to remove dissolved phosphorus from aqueous solutions. In the work by Erickson et al. (2007) steel wool was used to remove dissolved phosphorus from synthetic stormwater in both laboratory batch and column studies. Based on this work, scrap iron shavings were used as an enhancing agent in a 0.27 *acre* stormwater sand filter in Maplewood, Minnesota (Erickson et al. 2010). As a waste product, the iron shavings were selected over steel wool due to its smaller cost. The filter is currently being monitored and is removing over 80% of the dissolved phosphorus from the runoff it filters. The iron shavings appear to have most, if not all, of the qualities needed for an optimum enhancing agent that can remove dissolved nutrients. It's relatively inexpensive, safe to handle, does not significantly alter the pH, and, based on the work by Erickson et al. (2007, 2010), appears to have a large capacity such that the life of the iron will outlast the life of the filter.

Lucas and Greenway (2011) utilized water treatment residuals (WTRs) in bioretention mesocosms. At hydraulic loads from 24.5 to 29.3 *m/yr* at a flow-weighted average between 2.8 and 3:2 mg/L PO₄-P, input mass load removals ranging between 95 to 99% were observed over 80-week period in these WTR amended mesocosms.

O'Neill and Davis (2012a, 2012b) investigated the ability of aluminum-based drinking water treatment residual (that contained aluminum and iron), triple-shredded hardwood bark mulch, and washed quartz sand to remove dissolved phosphorus from water. Phosphorus adsorption isotherms were developed from batch studies and column experiments were performed. The laboratory column experiments were run continuously for 57 days. The columns contained 12 *cm* of media and had a water loading rate of 15 *cm/hr* (1.3 *mL/min*) for the first 28 days. Loading rates were increased to 31 *cm/hr* (days 29 through 49) and finally to 61 *cm/hr* (days 49 through 57) to force column breakthrough. At a loading rate of 15 *cm/hr*, which is typical for stormwater bioretention facilities, quartz sand with 4% water treatment residual reduced the dissolved phosphorus concentration from 0.12 mg/L to, on average, less than 0.02 mg/L for the

entire 28 day span (750 bed volumes). Although the effluent concentration increased as the loading rate increased, breakthrough did not occur over the course of the 57 day experiment (2500 bed volumes). As expected, greater phosphorus removal was observed as the amount of WTR increased.

A benchmark for phosphorus adsorption to provide necessary stormwater treatment was presented and discussed in O'Neill and Davis (2012a, 2012b). As developed in the paper, the benchmark was calculated to be 34 mg P/kg of media at a phosphorus water concentration of 0.12 mg P/L. The WTR media met this benchmark when WTR content was at least 4-5% by weight of the media mixture. This paper also presented and examined the oxalate ratio (OR), which is defined as:

$$OR = \frac{Al_{ox} + Fe_{ox}}{P_{ox}} \quad (1)$$

where Al_{ox} = oxalate-extractable Al, Fe_{ox} = oxalate-extractable Fe, and P_{ox} = oxalate-extractable phosphorus. Results indicated that at the phosphorus concentration examined in this study (0.12 mg P/L), there was a linear relationship between the OR and phosphorus adsorption. Thus, the authors state that the OR may be a useful predictive tool to estimate the ability of a medium to adsorb phosphorus for stormwater treatment.

Other materials that have been investigated for phosphorus removal potential include slag, red mud, other iron based components, zirconium, coal fly ash, crab shells, lithium, magnesium or manganese-layered double hydroxides (Chouyyok et al. 2010; Zhang et al. 2008), aluminum oxide, calcareous sand, limestone, blast oxygen furnace dust (a by-product of the steel making industry), iron oxides, calcite, high calcium marble, clay, diatomaceous earth, and vermiculite (Erickson 2005). While these materials may be effective, steel wool and/or iron shavings have been found to be effective at capturing dissolved phosphorus while meeting other practical requirements such as being relatively inexpensive, easy to place, not clogging the filter, and having a long life. The materials previously mentioned typically have limitations associated with them such as changing the pH of the water to unacceptable values, clogging the filter, dissolving and passing through the filter, etc.

Ma et al. (2011) developed a lightweight ($< 400 \text{ kg/m}^3$) engineered media from perlite and activated aluminum oxide to adsorb Ortho-phosphorus and to filter particulate matter. This engineered media, along with perlite, zeolite, and granular activated carbon (GAC) were studied to develop adsorption isotherms with respect to dissolved phosphorus removal. The engineered media had the highest adsorption capacity (7.82 mg/g), which was almost seven times that of the GAC (1.16 mg/g) and higher than any of the other materials tested. Under adsorption breakthrough testing, the engineered media treated 838 empty bed volumes before dissolved phosphorus removal efficiency dropped below 50%. GAC treated 12 empty bed volumes before dropping below 50% removal. Over the life of the engineered media and GAC, dissolved phosphorus removal rates can be expected to be 33 - 35% for both materials but the

engineered media was found to have a life expectancy of over twice that of the GAC (>2000 empty bed volumes to 1000 empty bed volumes for GAC).

Okochi and McMartin (2011) investigated the potential of electric arc furnace (EAF) slag as a possible sorbent to remove dissolved phosphorus from stormwater. The slag was obtained from a steel recycling facility. Six different synthetic stormwater recipes were used (Table 3) to test the effect of other contaminants in the stormwater on the effectiveness of the EAF slag. All solutions, although varying in metal concentrations, had a constant phosphorus concentration of 5 mg P/L.

Table 3. Chemical composition of the six stormwater solutions tested by Okochi and McMartin (2011).

	Description	Composition
1	Metal-free stormwater	$\text{Cd}^{2+}, \text{Cu}^{2+}, \text{Pb}^{2+}, \text{Zn}^{2+} \approx 0$
2	Cd-dominant stormwater	$\text{Cd}^{2+}: 1 \text{ mg/L}, \text{Cu}^{2+}, \text{Pb}^{2+}, \text{Zn}^{2+} \approx 0$
3	Cu-dominant stormwater	$\text{Cu}^{2+}: 1 \text{ mg/L}, \text{Cd}^{2+}, \text{Pb}^{2+}, \text{Zn}^{2+} \approx 0$
4	Pb-dominant stormwater	$\text{Pb}^{2+}: 1 \text{ mg/L}, \text{Cd}^{2+}, \text{Cu}^{2+}, \text{Zn}^{2+} \approx 0$
5	Zn-dominant stormwater	$\text{Zn}^{2+}: 1 \text{ mg/L}, \text{Cd}^{2+}, \text{Cu}^{2+}, \text{Pb}^{2+} \approx 0$
6	Multi-metal stormwater	$\text{Cd}^{2+}, \text{Cu}^{2+}, \text{Pb}^{2+}, \text{Zn}^{2+} = 1 \text{ mg/L}$

In summary, the presence of cadmium, lead, and zinc had little effect on the uptake of phosphorus by the EAF slag while copper was a significant inhibitor of phosphorus uptake. The effect of copper could be due to a stronger attraction to the adsorbent but the effect was not noticed in the multi-metal stormwater solution containing copper and other metals. Phosphorus uptake was greatest in the metal-free and multi-metal stormwater solutions. Phosphorus adsorption onto the EAF slag ranged from 0.82 mg P/g in the copper dominated synthetic stormwater solution to 1.18 mg P/g in the metal-free stormwater solution.

Wium-Anderson et al. (2012) tested crushed limestone, shell-sand, zeolite, and two granular olivines, which is a magnesium iron silicate ($(\text{Mg,Fe})_2\text{SiO}_4$), for their ability to remove dissolved phosphorus and metals from water. The limestone contained 96.8% calcium carbonate, 1% magnesium carbonate, and trace amounts of aluminum oxide. The zeolite was Clinoptilolite ($(\text{Ca}, \text{K}_2, \text{Na}_2, \text{Mg})_4\text{Al}_8\text{Si}_4\text{O}_{26}$). The shell-sand used was a carbonatic material derived from fossils of shelled organisms such as snails, crabs, and mussels. The material consisted of 33% calcium, 1% magnesium, and traces of arsenic, cadmium, and lead. Initial contaminant concentrations in the synthetic stormwater used are shown in Table 4. Results in the form of percent removal after 1 minute and after 10 minutes are shown in Table 5.

Table 4. Initial synthetic stormwater concentrations used with sorbents (Wium-Anderson et al. 2012).

	Limestone (µg/L)	Shell-sand (µg/L)	Zeolite (µg/L)	Olivine I (µg/L)	Olivine II (µg/L)
As	200	200	200	200	200
Cd	40 000	40 000	4 000	40 000	40 000
Cr	40 000	40 000	8 000	40 000	8 000
Cu	40 000	40 000	2 000	40 000	8 000
Ni	200	2 000	4 000	8 000	4 000
P	8 000	8 000	200	40 000	4 000
Pb	40 000	40 000	40 000	40 000	40 000
Zn	40 000	8 000	2 000	40 000	4 000

Table 5. Median initial removal efficiencies in percent of the initial concentration. 25% and 75% percentiles are shown in brackets (Wium-Anderson et al. 2012).

Removal after 1 minute								
	Arsenic (% removal)	Cadmium (% removal)	Chromium (% removal)	Copper (% removal)	Nickel (% removal)	Lead (% removal)	Zinc (% removal)	Phosphorus (% removal)
Olivine I	23.3 (19.9/24.2)	96.0 (89.7/96.7)	94.2 (74.0/95.2)	96.7 (80.6/97.0)	74.1 (71.5/78.0)	89.9 (61.7/91.9)	94.6 (87.6/96.3)	49.1 (43.8/54.2)
Olivine II	15.5 (13.5/18.5)	97.2 (95.9/97.6)	96.4 (77.1/98.6)	95.9 (74.8/96.3)	95.0 (94.4/95.6)	86.3 (84.8/87.4)	98.5 (98.1/99.1)	74.7 (72.0/77.8)
Limestone	0.1 (-0.5/0.5)	79.8 (77.1/83.0)	73.1 (58.2/89.2)	74.3 (64.9/88.7)	33.5 (18.8/35.6)	72.4 (72.3/93.0)	87.2 (82.9/88.4)	6.8 (2.4/11.0)
Shell-sand	0.5 (-1.7/2.8)	93.2 (83.2/96.6)	99.6 (91.1/99.7)	87.5 (83.3/95.5)	66.2 (59.7/78.0)	88.0 (74.3/95.3)	92.9 (87.3/95.8)	12.4 (2.0/31.1)
Zeolite	0.3 (-1.2/1.3)	59.6 (53.1/79.1)	94.1 (92.2/96.9)	84.3 (61.5/91.0)	68.5 (50.1/83.5)	81.9 (78.8/85.8)	83.1 (81.2/89.5)	-2.2 (-3.0/-0.2)
Removal after 10 minutes								
Olivine I	72.4 (71.4/75.9)	99.3 (99.3/99.8)	92.4 (77.6/94.8)	97.8 (95.3/98.1)	98.8 (98.3/99.5)	92.4 (81.5/94.7)	95.6 (95.0/98.0)	94.5 (94.3/96.2)
Olivine II	53.6 (49.8/59.1)	99.2 (98.4/99.5)	93.5 (43.9/94.8)	96.6 (83.2/97.4)	99.7 (99.4/99.8)	91.9 (89.2/92.5)	99.8 (99.6/99.9)	97.7 (95.8/98.8)
Limestone	9.6 (5.2/15.0)	99.1 (98.4/99.2)	82.4 (68.6/93.9)	86.0 (74.1/94.1)	79.8 (73.7/84.4)	95.8 (82.7/98.4)	96.4 (95.6/96.8)	62.5 (31.8/78.7)
Shell-sand	17.4 (5.3/35.8)	98.8 (98.6/99.1)	99.6 (99.2/99.8)	88.8 (81.5/93.8)	95.4 (94.5/96.6)	89.6 (85.6/91.5)	95.8 (91.8/97.8)	75.0 (37.7/87.8)
Zeolite	14.7 (10.2/19.1)	87.8 (83.4/96.5)	99.2 (88.2/99.6)	92.5 (80.8/95.1)	90.7 (73.8/95.3)	86.3 (82.0/87.3)	95.3 (90.2/97.3)	30.8 (25.4/36.1)

The granular olivine materials had the highest removal efficiencies, the most rapid sorption kinetics, and the highest cost. They also raised the pH to almost 10, which may not be acceptable in many applications, and leached chromium into solution. The shell-sand had the lowest unit cost but relatively high sorption capacity and caused the pH to stabilize around 8. The zeolite and limestone had relatively low sorption capacity. The authors concluded that none of the sorbents tested were ideal for all contaminants, however, at least some of the materials and/or combinations thereof, could be effective in removing dissolved contaminants from water.

Wendling et al. (2013) performed column experiments to investigate the ability of water treatment residuals (WTR), coal fly ash, and granular activated carbon (GAC) from biomass combustion for its ability to retain nutrients such as nitrogen and phosphorus in water treatment systems. Water treatment residuals were CaO and CaCO₃ based materials used to remove color and odor from groundwater. The CaCO₃ also included garnet, an iron rich material. The coal fly ash was obtained from a coal-fired power plant and the granular activated carbon was obtained from a eucalyptus tree processing plant. Influent containing dissolved phosphorus

concentration of 0.52 mg/L was pumped through the columns in an upflow mode so that the residence time was 12 hours. Removal rates for phosphorus and other contaminants studied are shown in [Table 6](#).

Table 6. Cumulative removal rates found by Wendling et al. (2013).

Column contents	Effluent pH range	Effluent volume (L)	Cumulative nutrient and DOC removal (-) or enrichment (+)													
			NO _x -N		NH ₃ -N		DON		TN		PO ₄ -P		TP		DOC	
			mg/kg	%	mg/kg	%	mg/kg	%	mg/kg	%	mg/kg	%	mg/kg	%	g/kg	%
Influent water	6.9-7.4	-	0.44 mg/L		0.32 mg/L		1.3 mg/L		2.1 mg/L		0.52 mg/L		0.69 mg/L		59.2 g/L	
Native sand	7.0-8.2	43.3	+0.2 ^a	+0.7	+4	+17	+8	+10	+12	+9	-2	-6	-3	-8	+0.2	+6
GAC	7.7-8.5	42.9	-86	-11	-579^b	-76	-876	-38	-1,540	-40	+278	+35	+91	+8	-42.19	-59
CFA	7.7-8.8	26.7	-134	-86	-136	-84	-39	-12	-309	-49	-14	-12	-42	-27	-4.23	-27
CaCO ₃ -based WTR	7.2-8.1	14.7	-124	-97	-72	-61	+97	+58	-99	-24	-25	-37	-40	-46	+2.49	+29
CaO-based WTR	8.9-13.0	43.3	-167	-92	-128	-73	-12	-2	-308	-35	-184	-98	-254	-98	-9.51	-38

^a +xx = net increase as compared to inlet water.

^b Cumulative reductions of >50% as compared to inlet water in bold.

Based on previous work, the most promising enhancing agents for the capture of phosphorus appear to be steel wool, iron shavings, water treatment residual, or other iron based materials. They have already been shown to have relatively high capture rates, be inexpensive, and have a long life.

4. Summary

Swales can achieve significant stormwater runoff volume reduction. The extent of volume reduction is dependent on many variables including rainfall characteristics, antecedent soil moisture, vegetation characteristics in the swale, swale slope, etc. Reported volume reductions range from less than 10% to 100% with many values above 50%. Highly important variables affecting volume reduction performance appear to be the length of the swale and the saturated hydraulic conductivity of the soil.

Phosphorus removal by swales has ranged from large negative values to over 60%. Differences can be attributed to many variables, including assessing dissolved phosphorus or total phosphorus removal, season of the year, vegetative cover, and more. It has been suggested that removing grass clippings from swales could increase their overall performance with respect to phosphorus removal.

Ditch check filters with enhanced filter media can be installed in the swales to increase the potential of swales to remove dissolved phosphorus from stormwater runoff. Of the materials reviewed for phosphorus removal, ferrous based materials, alumina, and water treatment residual have potential to provide effective phosphorus removal.

For additional information on swale performance, including removal of total suspended solids and other dissolved contaminants such as nitrogen and metals, and materials for enhanced metals removal, see Appendix A.

III. Infiltration Performance of Swales (Task 3)

Infiltration measurements taken in the field along with other information can be used to quantify the infiltration performance of an existing swale. This section of the report will briefly describe the K_{sat} measurements and apply them through a model to compute the quantity of runoff that is infiltrated and that which remains in the swale as surface flow.

1. Methods

One objective of this project was to characterize the infiltration capacity of different grassed swales of different soil types located in Minnesota. Primarily fifteen swales were selected and based upon the soil type five swales were finally selected for this study. Infiltration measurements were taken in these five swales. Among these five swales, three were chosen to study the effect of season, soil moisture content and distance from the outflow pipe downstream (due to sedimentation) on saturated hydraulic conductivity. At each highway three swales/drainage ditches were chosen and at each swale infiltration measurements were taken for three different initial moisture contents.

Another objective of this project is to apply the field measurements of infiltration in swales to represent performance of swales by developing an infiltration model. A swale at Madison, WI was selected to verify the infiltration model by comparing the predicted outflow rate data with monitored outflow rate data. In addition an analysis was performed on this swale to estimate the number of measurements required to obtain a representative geometric mean K_{sat} of that swale and to compare the infiltration at the side slope versus infiltration at the center of the swale.

1a. Field Measurements of Infiltration in Swales /Drainage Ditches

Initially fifteen swales were chosen, at the locations provided in [Figure 2](#), as representative of those in Minnesota.

- 1) TH 10 and Main St
- 2) TH 47 north of I-694
- 3) TH 65 north of I-694
- 4) TH 13 – Prior Lake
- 5) TH 51 – north of Bethel College south of 694
- 6) TH 36/61 area
- 7) TH 77 south of TH 62
- 8) TH 212 – New ditches on new roadway or farther out past Chaska
- 9) TH 61 – south of Cottage Grove
- 10) TH 7 – western Henn County
- 11) TH 5 – east of I-494
- 12) TH 95 – north of Stillwater, south of Taylors Falls

- 13) I-35W near TH 10
- 14) I-35W in Burnsville
- 15) I-35E north of I-694

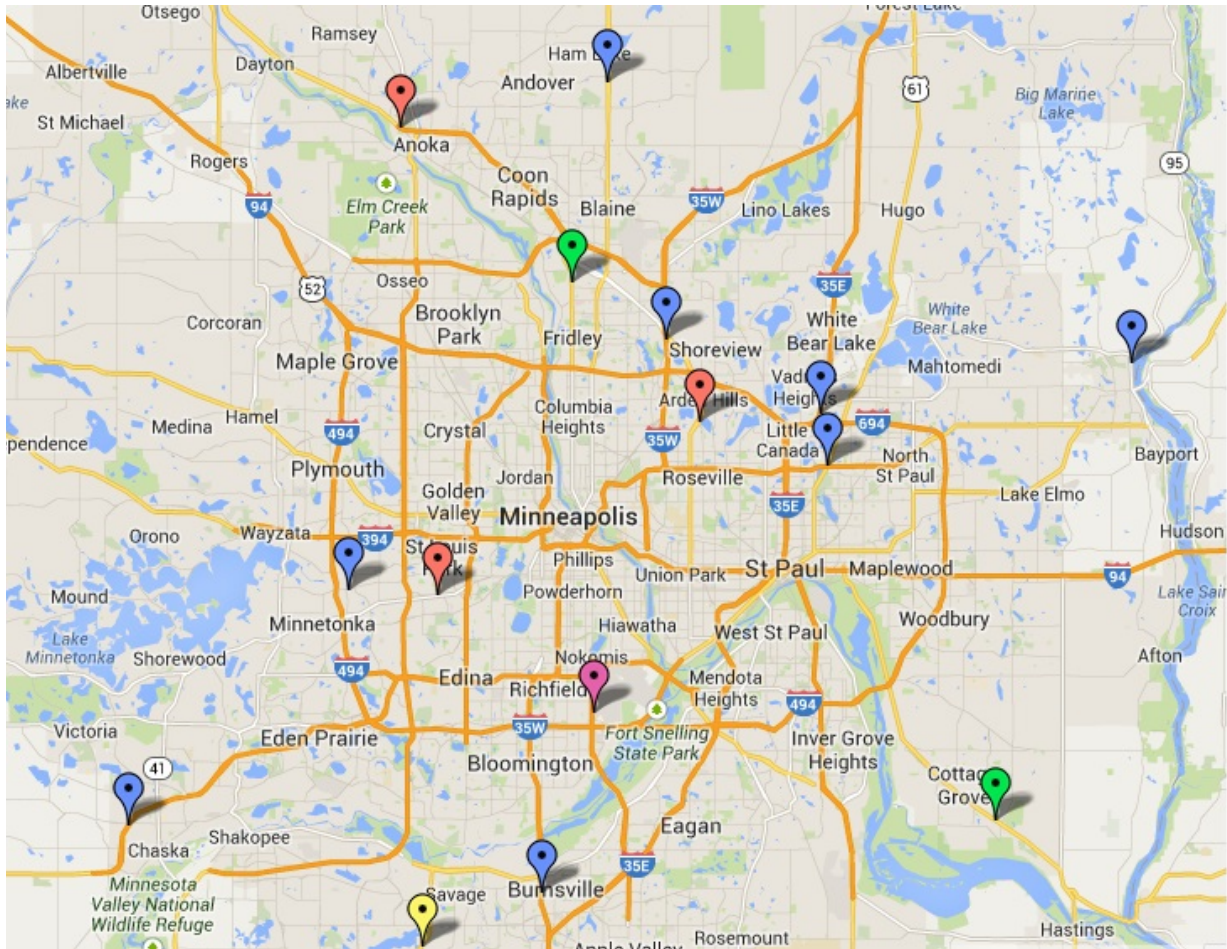


Figure 2. Locations of preliminary selected swales.

From each of the swales three or four soil cores were collected using a soil corer. Soil cores were approximately 24 inches deep. From each soil core soil samples were collected from different depth where a change in soil color and texture was observed visually. These soil samples were brought into the lab for further analysis. The purpose of collecting soil cores is to perform textural analysis on the soil sample and identify the soil type of that swale. On each soil sample wet sieving analysis and hydrometer analysis were performed. This combination is a standard method to determine % clay, % silt and % sand in a soil sample. Using these percentages in a textural triangle, the soil type was identified and listed on pg. 33.

For each type of soil one swale has been selected for measurement of K_{sat} and the number of swales was narrowed down to five. Infiltration tests were performed at each of these five swales using a Modified Philip Dunne (MPD) Infiltrometer (Figure 3). This is a new method of measuring infiltration capacity which has recently been used by Asleson et al. (2009) and Olsen et al. (2013). It is a rapid infiltration technique that allows multiple measurements to be taken over space. This is needed because the infiltration capacity at a given site will have substantial spatial variation. The MPD Infiltrometer was used to calculate the saturated hydraulic conductivity (K_{sat}) of the soil at that location. At each swale site, seventeen to twenty measurements were taken within a 20~60 ft (7~21 m) long stretch on the highway (Figure 4) based on the statistical analysis of the data summarized in the Results section.



Figure 3. Picture of a Modified Philip Dunne (MPD) infiltrometer.



Figure 4. Collecting infiltration measurement at Hwy 51, Arden Hills, MN.

The K_{sat} results were inspected visually and believed to be log normally distributed. An analysis of variance (ANOVA) test was performed to determine the differences among sample geometric means of the logarithm of K_{sat} for different swales or for the same swale but with different soil moisture content for the cases where the number of observation of each swale was the same. For the cases where the number of observation for each swale was not same, a Tukey-Kramer test (Kleinbaum et al. 2007) was performed to determine differences among sample means of K_{sat} .

In addition to the measurements taken at the sites shown in Figure 2, a total of 108 infiltration measurements were taken in a swale located near Hwy 51 in Madison, WI. The purpose was to compare predictions of flow from K_{sat} measurements with flow monitoring being undertaken by the U.S. Geological Survey in the swale. The swale was divided into 20 cross sections, 40 to 78 ft apart and at each cross section 3 to 7 measurements were taken 4 ft apart. ANOVA and regression analysis was performed to determine the differences among sample means of K_{sat} of each cross section. An ANOVA test was also performed to determine the differences among sample means of K_{sat} of each row, and an analysis of the uncertainty associated with a given number of measurements was performed.

1b. Application of Infiltration Field Measurements to Runoff

A stormwater runoff model was developed to calculate the stormwater infiltration efficiency of a swale by using field infiltration measurements. This model is applicable for different design

storms, or for observed storm events. The stormwater runoff model as applied to a swale focuses on the calculation of the water balance for rain falling directly on the swale and for stormwater introduced into the swale from the adjacent roadway and concentrated inflows by culverts or drainage pipes. The water balance is conducted by calculating infiltration loss of direct rainfall and introduced stormwater, and routing the excess volume to the outlet of the swale.

The routing of the flow in the swale is accomplished by dividing the swale into cross-sections separated by a longitudinal distance B_j for each time increment (Figure 5) and then computing a water surface profile (Figure 6), taking into account the surface area of infiltration and the water balance for each cross section. The water surface profile in the swale is developed using the dynamic equations of gradually varied flow (Figure 7). For a known water depth, equations 2 through 5 were applied between the two ends of each cross section to calculate Δx , the longitudinal distance required to achieve a specified value of depth, Y .

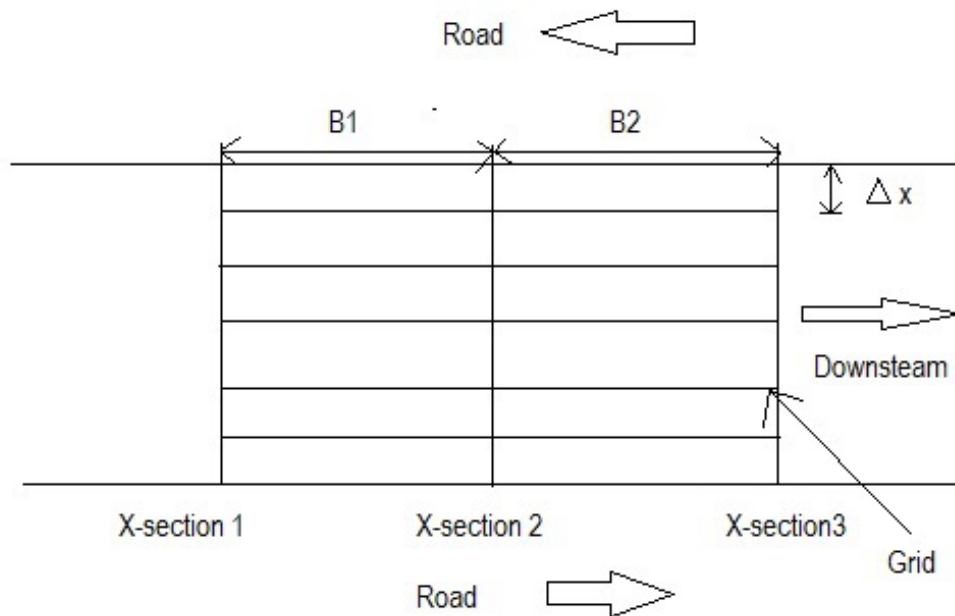


Figure 5. Plan view of a typical swale.

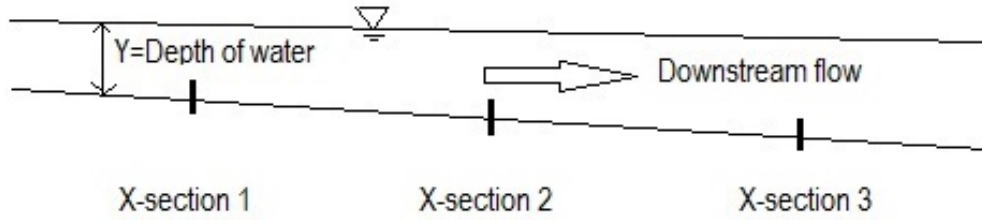


Figure 6. Longitudinal view of a typical swale.

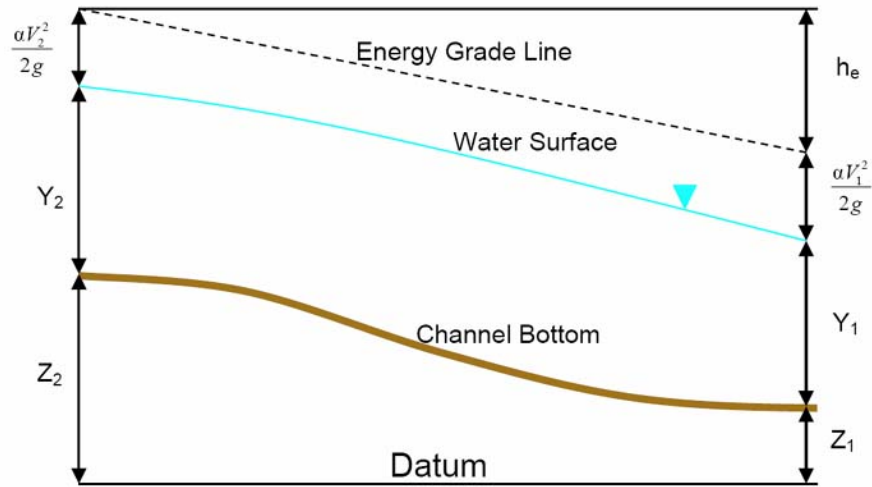


Figure 7. Gradually varied flow energy balance.

$$E_1 = Y_1 + \alpha \frac{V_1^2}{2g} \quad (2)$$

$$E_2 = Y_2 + \alpha \frac{V_2^2}{2g} \quad (3)$$

$$\Delta x = \frac{E_2 - E_1}{S_0 - S_f} = \frac{\Delta E}{S_0 - S_f} \quad (4)$$

$$S_f = \left(\frac{nQ_{in_j}}{AR^{2/3}} \right)^2 \quad (5)$$

where, E = Specific energy,

Y = Depth of water in the swale (ft),

a = Energy co-efficient,

V = Velocity of water across the cross-section (ft/s),

S_0 = Average bottom slope of the swale,
 S_f = Frictional slope computed from Manning's equation,
 n = Manning's coefficient,
 Q_{in_j} = Inflow rate to section j (ft^3/s),
 A = Cross-sectional area across the swale (ft^2),
 R = Hydraulic radius (ft).

For a given flow rate from upstream (Q_{in_j}), the water surface profile for each cross section is quantified. The surface area for infiltration is calculated from the wetted perimeter and width of the cross section, B_j . Then the outflow rate is calculated by the water balance equation, which is applied at each cross section. The water balance equation is given by:

$$Q_{out_j}\Delta t = (Q_{in_j} + Q_{side_j} + Q_{conc_j})\Delta t - A_j * F_j/12\left(\frac{in}{ft}\right) \quad (6)$$

where, Q_{out_j} = downstream outflow rate for section j (ft^3/s),

Q_{in_j} = inflow rate to section j (ft^3/s),

Q_{side_j} = lateral inflow to section j (ft^3/s),

Q_{conc_j} = concentrated inflow from culverts or pipes to section j (ft^3/s),

A_j = Area of the infiltrating surface for section j (ft^2),

$F_j(t)$ = infiltration depth (in) into the swale bottom of section j during the time interval.

The outflow discharge at the downstream end of the swale is the discharge from the swale. The depth of the water in the swale used in equations 2 through 5 is a fitted depth to obtain the monitored outflow rate.

The excess flow from the swale side slope Q_{side_j} is calculated by determining infiltration into the side slope of direct rainfall combined with the stormwater flow from the adjacent road surface. The Green Ampt equation (Mays 2005) is employed to compute Q_{side_j} . The equation for cumulative infiltration is:

$$F(t) = K_{sat}t + \psi\Delta\theta\ln\left(1 + \frac{F(t)}{\psi\Delta\theta}\right) \quad (7)$$

where, $F(t)$ = cumulative infiltration (in),

K_{sat} = saturated hydraulic conductivity of soil (in/hr),

Ψ = wetting front suction (in),

$\Delta\theta$ = change in soil moisture content during the storm, or $(\theta_s - \theta_i)$,

θ_s = saturated moisture content (fraction), and
 θ_i = initial moisture content (fraction).

By using [equation 7](#), the infiltration loss into the soil as well as the volume of runoff that does not infiltrate in the swale can be quantified. The unknown parameters in [equation 7](#) are K_{sat} , Ψ and $\Delta\theta$. The moisture content needs to be determined prior to the rainfall event. K_{sat} and Ψ can be determined using either field measurements or estimation methods such as pedo-transfer function procedures (Schaap et al. 2001). For this study, a field measurement method, the Modified Philip Dunne (MPD) Infiltrometer, was employed. This device facilitates taking multiple measurements simultaneously, which allows the capture of spatial variability of K_{sat} and Ψ (Asleson et. al, 2009; Ahmed, et al. 2014). In the field the swale is first divided into grids and infiltration measurements are taken at each cell to estimate K_{sat} and Ψ .

The model is developed so that it can receive surface runoff from one or both sides of the swale. The swale is divided into multiple cross sections in the longitudinal direction ([Figure 15](#), page 48). Then, each cross section is divided into multiple cells along the swale side slope down to the base of the swale. These cells are employed to calculate the progressive downslope loss of stormwater introduced at the edge of the road surface. Rain falling directly on each cell is also accounted for in the calculation of infiltration. The calculation procedure is therefore the following: For a given rainfall intensity, the amount of infiltration of direct rainfall and input road surface storm flow of the cell closest to the road is calculated for each swale cross section and the excess volume is passed along the side slope of the swale on to the next cell downslope. The rainfall and stormwater that does not infiltrate along the cross section side slope is excess flow and reaches the center of the swale. This excess flow becomes the input, $Q_{side,j}$, for cell j in [equation 6](#). The sum of outflow volume ($V_{out} = \sum Q_{out,j}\Delta t$) and the volume of total rainfall (V_{rain}) is calculated for each rainfall event. [Figure 8](#) shows a flow chart of the steps involved in the runoff-routing model.

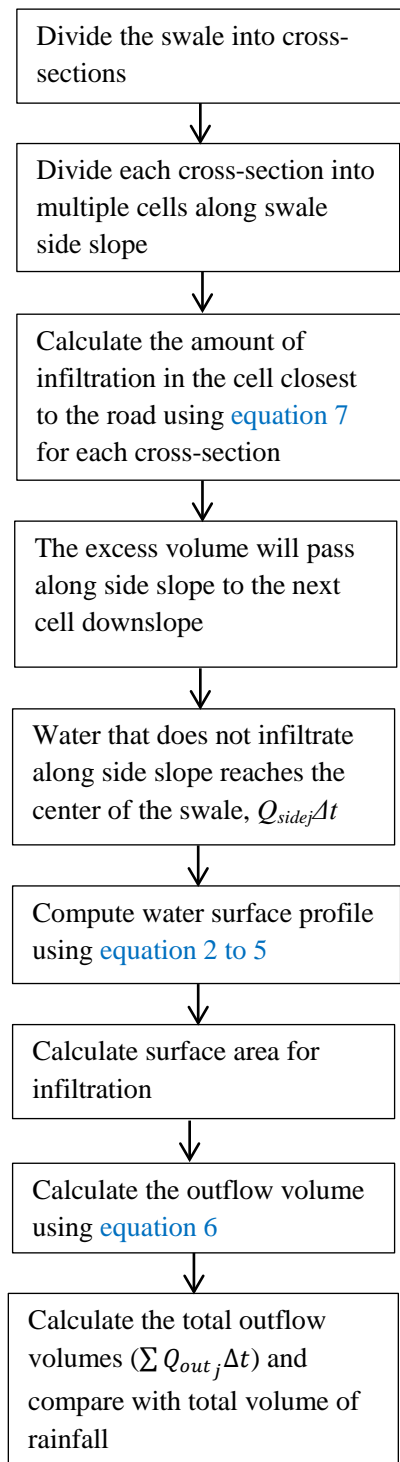


Figure 8. Flow chart of the steps involved in runoff-routing model.

2. Results and Discussion

2a. Selecting Swales to Perform Infiltration Tests

As discussed in the methods section (1a), after collecting a soil sample from the fifteen swales, soil textural analysis was performed on the soil samples. The lists of soil type for different swales are given in Table 7. In Table 7, if the same highway is addressed in two rows it indicates that the swale located in that highway contains both types of soil (i.e. Hwy 35E, Hwy 35W near TH 10).

Table 7. Soil type of different swales located in Minnesota. Letters in parenthesis are the HSG classification that would result from the mean values of K_{sat} in Rawls et al. (1983).

Swale locations	Soil type
Hwy 10, Hwy 35E, Hwy 35W near TH 10	Sand (A)
Hwy 5, Hwy 47, Hwy 65, Hwy 96, Hwy 97, Hwy 77, Hwy 7, Hwy 35W Burnsville, Hwy 35E, Hwy 35W near TH 10	Loamy sand (B) and Sandy loam (C)
Hwy 51, Hwy 36	Loam (C) and Sandy loam (C)
Hwy 212	Silt loam (C) and Loam (C)
Hwy 13	Loam(C), Sandy clay loam (D) and Silt (C)

For each type of soil one swale has been selected for infiltration measurement and the number of swales was narrowed down to five on which infiltration measurements were taken in Fall of 2011. Table 8 shows the soil type, number of measurements and the initial soil moisture content of these five swales.

Table 8. Soil types of five swales selected for infiltration measurement.

Swale location	Soil type	# of measurement	Soil moisture content(%)
Hwy 77	Loamy sand	17	18
Hwy 47	Loamy sand/ Sandy loam	20	32
Hwy 51	Loam/ Sandy loam	20	26
Hwy 212	Silt loam/ Loam	20	29
Hwy 13	Loam/ Sandy clay loam/ Silt	19	24

From these five swales three were chosen where repeated infiltration measurements were taken in following spring. The purpose of taking the measurements in Spring 2012 is to analyze the effect of season on the geometric mean saturated hydraulic conductivity (K_{sat}) of the swale. Table 9 shows the number of measurements and soil moisture content in three swales for Fall 2011 and Spring 2012.

Table 9. Number of infiltration measurements in Fall 2011 and Spring 2012.

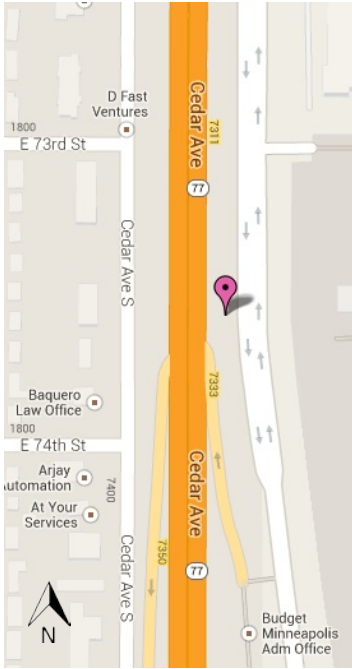
Location	Measurements in Fall 2011		Measurements in Spring 2012	
	# of measurements	Soil moisture content (%)	# of measurements	Soil moisture content (%)
Hwy 47	20	32	20	28
Hwy 51	20	26	20	15
Hwy 212	20	29	20	20

At each of these three highways three swales/drainage ditches were chosen and at each swale infiltration measurements were taken for three different initial moisture contents. The purpose is to analyze the effect of season, soil moisture content and distance from the outflow pipe downstream (due to sedimentation) on geometric mean K_{sat} . Table 10 shows the location and number of measurements on these swales.

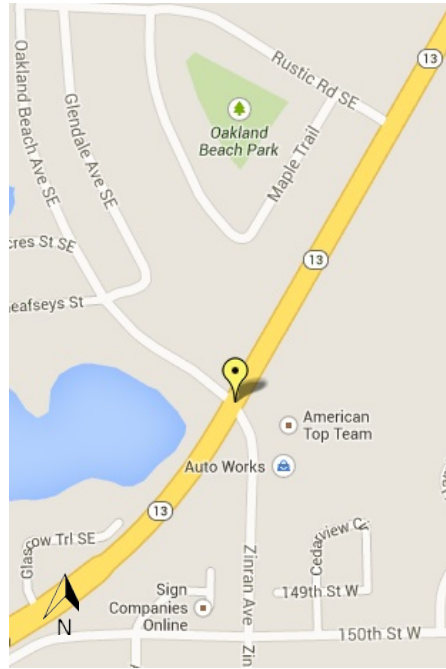
Table 10. Location and measurements at Minnesota swales.

Location	Date	Number of measurements	Moisture content (%)
Hwy 47 (north)	6/7/12	21	22
	6/27/12	21	23
Hwy 47 (center)	4/2/12	21	31
	6/1/12	21	37
	6/18/12	19	43
Hwy 47 (south)	8/2/12	21	23
	8/23/12	21	10
Hwy 51 (north)	6/28/12	18	27
	7/13/12	18	30
	8/10/12	18	32
Hwy 51 (center)	5/30/12	21	33
	6/11/12	21	30
	not recorded	18	15
Hwy 51 (south)	6/5/12	21	29
	6/12/12	21	38
Hwy 212 (east)	8/9/12	21	24
	8/24/12	21	26
	8/27/12	21	25
Hwy 212 (center)	5/19/12	21	18
	6/21/12	12	39
	7/10/12	21	19
Hwy 212 (west)	7/12/12	21	12
	8/20/12	21	29

The map of location of swales in Minnesota where infiltration measurements were taken are shown in [Figure 9](#).



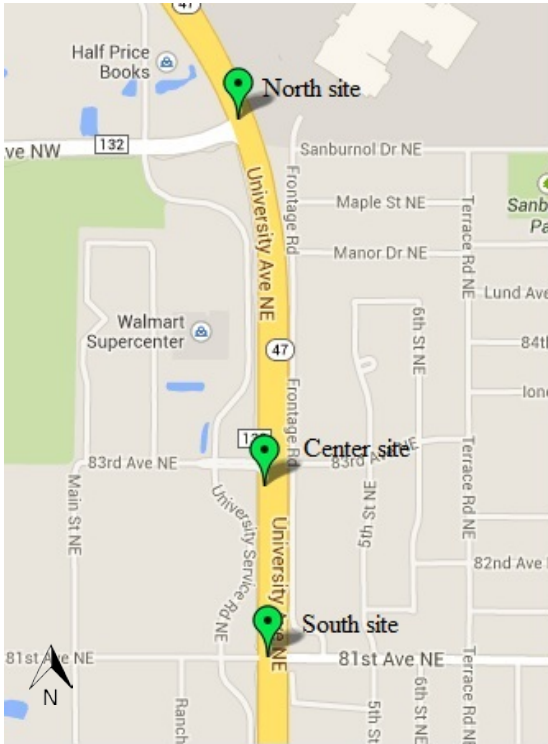
(a)



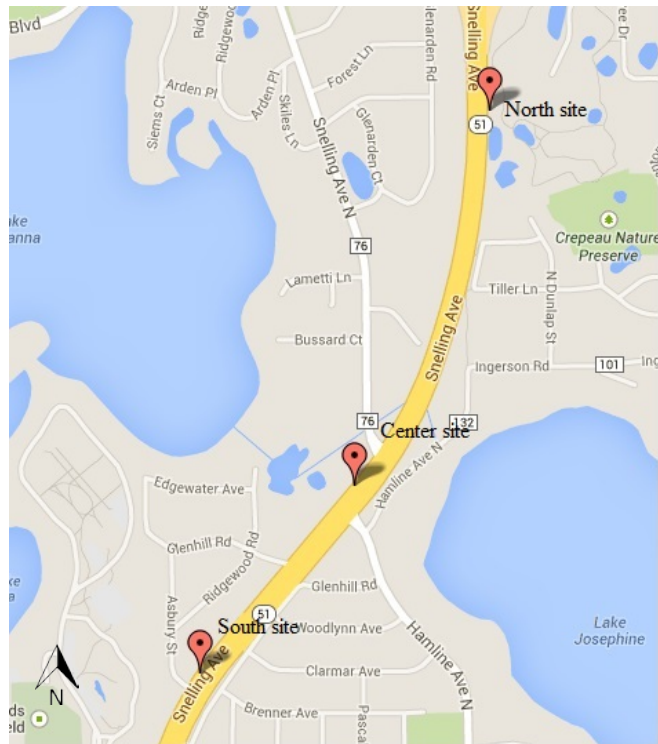
(b)



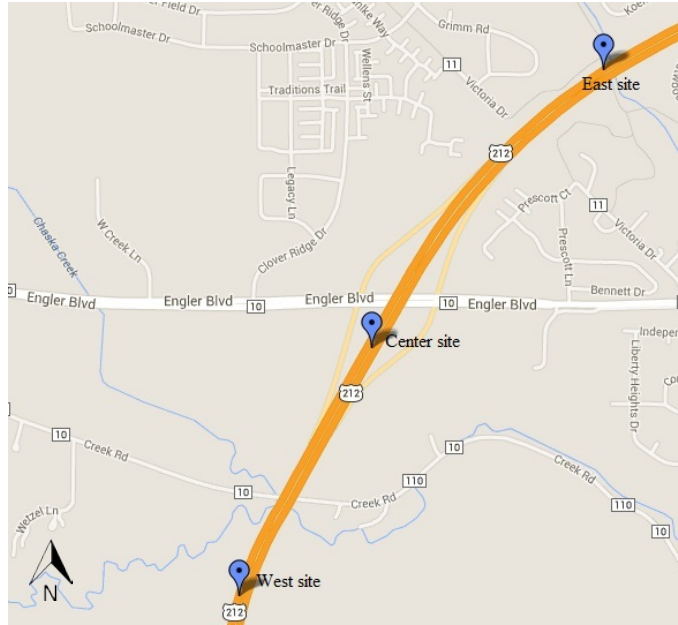
(c)



(d)



(e)



(f)

Figure 9. Location of infiltration test sites at (a) Hwy 77 (Cedar Ave. and E 74th St., North of Hwy 494, Bloomington, MN), (b) Hwy 13 (Hwy 13 and Oakland beach Ave. SE, Savage, MN), (c) Hwy 51 Madison, WI (S Stoughton Rd and Pflaum Rd, Madison, WI), (d) Hwy 47 (University Ave. NE and 81st Ave. NE, Fridley, MN), (e) Hwy 51 (Snelling Ave. and Hamline Ave. N, Arden Hills, MN) and (f) Hwy 212 (Hwy 212 and Engler Blvd, Chaska, MN).

2b. Statistical Analysis of Infiltration Measurements

The K_{sat} values of swales were observed to be log-normally distributed. As an example, the K_{sat} values of the Madison swale are plotted in a histogram, showing the frequencies of occurrence in given intervals for the values of K_{sat} (Figure 10). To obtain a distribution that better fits a normal distribution, a logarithmic transformation was performed for all infiltration measurements. The histogram of $\log_{10}(K_{sat})$ (Figure 11) indicates that the data is closer to a normal distribution in log space (a log-normal distribution). The statistical analyses on the infiltration measurements were therefore performed on the log-transformed distribution for all swales. The mean of a normal distribution in log space is the geometric mean, which will be reported for all K_{sat} measurements herein.

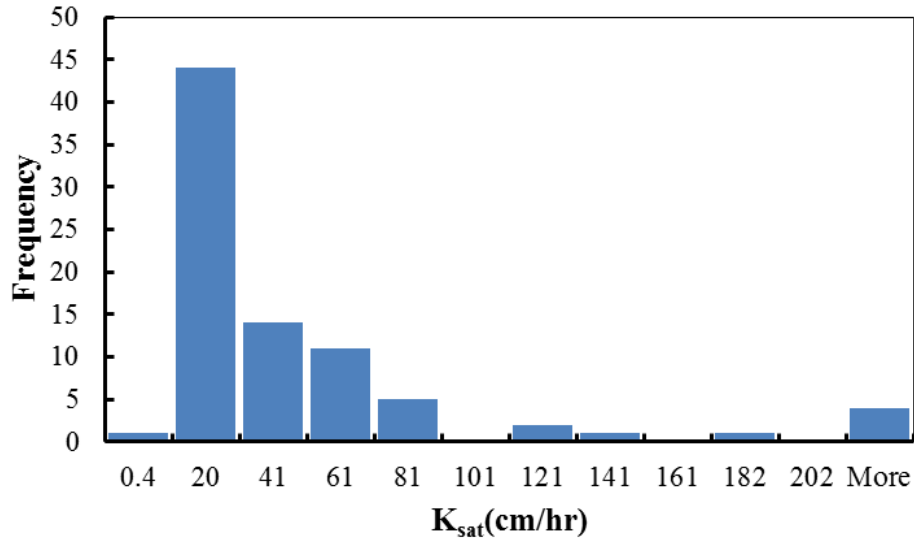


Figure 10. Histogram of actual K_{sat} values of Madison swale.

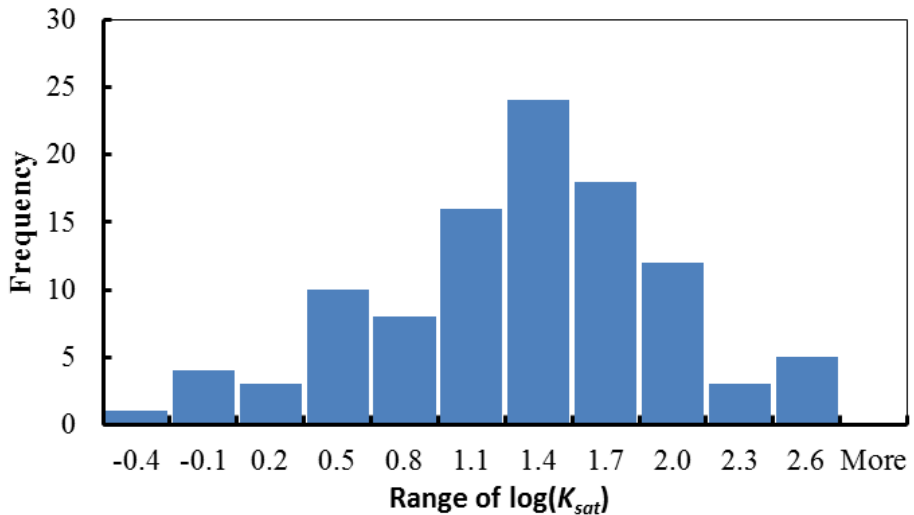


Figure 11. Histogram of log transformed K_{sat} values of Madison swale.

The impact of soil type, season, soil moisture content and distance from outflow pipe downstream were observed for the swales. A summary of the statistical analysis is provided in the following sections.

i. Estimating the Number and Location of Measurements

An analysis was performed on the 83 infiltration measurements taken upstream of the swale in Madison, WI on how many measurements need to be made to reduce uncertainty and keep the number of measurements to a reasonable value. Infiltration measurements at 6, 11, 17, 19, 39,

44 and 83 locations were selected randomly, and the geometric mean along with the 95% upper and lower confidence intervals were calculated for each case. Figure 12 shows the relation between the number of measurements, the geometric mean of K_{sat} and the uncertainty of the geometric mean. From the figure we can conclude that

- A large uncertainty of greater than a factor of three is associated with 11 or less measurements,
- Between 11 and 19 measurements, the geometric mean of K_{sat} varies between 30 and 45 cm/hr , with a rapidly decreasing uncertainty to a factor of 1.83 (multiply and divide the geometric mean by 1.83 to obtain the 95% confidence interval),
- Between 19 and 39 measurements the geometric mean of K_{sat} decreases from 30 to 25 cm/hr with a 95% confidence interval that decreased to a factor of 1.6, and
- Between 39 and 83 measurements the geometric mean of K_{sat} varies between 19 and 25 cm/hr , with a 95% confidence interval that reaches a factor of 1.4.

The uncertainty of the geometric mean does not decrease as rapidly beyond 20 measurements. This is roughly at the “knee” of the upper confidence interval curve. Our experience with the Madison swale and the other swales in this report is that approximately 20 infiltration measurements is an optimum balance between effort and accuracy of K_{sat} measurements.

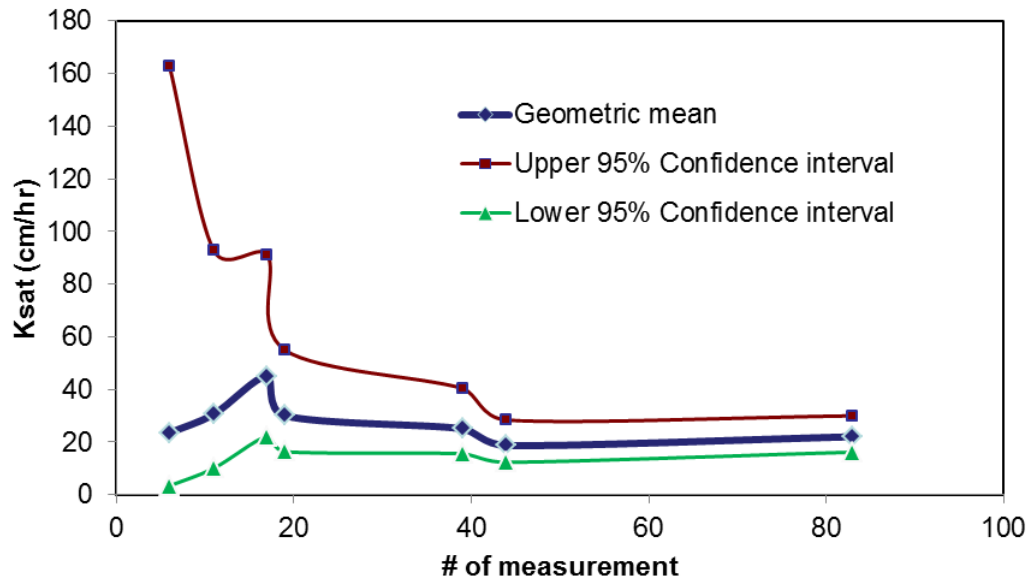


Figure 12. Relationship between number of measurements and K_{sat} for the Madison swale.

ii. Infiltration at Different Locations in the Swale

The Madison swale was divided into twenty cross sections and each section was divided into three to seven infiltration measurement locations. The objective is to identify statistically significant correlations that may allow or disallow patterns in the value of K_{sat} within the swale. Two questions will be asked: 1) is there a correlation of K_{sat} with longitudinal distance in the swale and 2) is there a correlation with distance from the edge of the swale.

For this purpose the Tukey-Kramer test was performed between the K_{sat} values of twenty cross sections and the K_{sat} of three cross sections (two cross sections at two ends and one at the center) within 90 to 99% confidence interval, and no significant difference was found between the geometric mean K_{sat} . Regression analysis was performed among the geometric mean K_{sat} of each cross section and no significant difference was observed among geometric mean K_{sat} values. The slope of the geometric mean K_{sat} vs. distance was close to zero, which indicates that there is little or no relation between distance and geometric mean K_{sat} .

Finally, a Tukey-Kramer test was performed between K_{sat} values of the side slopes and K_{sat} values of center of the swales located in Madison, WI, and in Minnesota. Except for one swale located near Hwy 212, MN, a significant difference between geometric mean K_{sat} values of the side slopes and the center of the swale was observed for swales located in Minnesota and Madison, WI, at a 95% confidence interval. The geometric mean of K_{sat} from the side slopes and the center of the swales, shown in Table 11, indicate that there is no strong trend. At Hwy 212 and Hwy 47, the side slopes had a higher K_{sat} than the center of the swale, and at Hwy 51 in Minnesota and Madison, the side slopes had a lower K_{sat} than the center of the swale. Thus, while the side slopes are different than the center of the swale, there is no indication of which will have a higher K_{sat} . There is, therefore, no observed evidence that sedimentation in the center of the swale causes a reduction in infiltration. One reason could be that the growth of plant roots in the center of the swale creates macropores that alleviates sedimentation-induced reduction of infiltration rates. It is noteworthy that except Hwy 212 all other swales are located at the side of the highway and Hwy 212 is located at the median of the highway.

Table 11. Comparison between geometric mean K_{sat} at the center and side slope of the swales.

Location	Center of the swale		Side slope of the swale	
	# of measurements	Geometric mean (in/hr)	# of measurements	Geometric mean (in/hr)
Hwy 212, Chaska, MN	78	0.3	90	0.5
Hwy 47, Fridley, MN	63	1.1	87	2.6
Hwy 51, Arden Hill, MN	78	1.9	81	0.9
Hwy 51, Madison, WI	20	9.5	84	5.6

Thus, we can conclude that care should be taken to separately measure infiltration rates at the side slope and center of the swale but there is no longitudinal interval that needs to be maintained between measurements. Our recommendation is to space the measurements cross-sections out over the region of interest, to make sure that side slopes are identified separately from the center of the swale, and to have at least 20 measurements of K_{sat} over this region.

iii. Effect of Soil Type

Measurements were taken at five swales with different soil types, given in Table 12. The geometric mean value of K_{sat} varied from 0.3 to 1.56 *in/hr*, a fairly small range given the variation in soil types. Note that the hydrologic soil group (given in parenthesis) tends to be higher than those listed in the second column of Table 12. These swales are up to 50 years old, and it is possible that the macropores created by the grass in the swale provides them with a higher K_{sat} than would normally be expected. Since the number of measurements was not the same, a Tukey-Kramer method was used to determine differences in geometric mean K_{sat} value for different soil types. No significant difference was observed among geometric mean K_{sat} values within the 50% to 95% confidence interval. The reason for this might be the high coefficient of variation (COV), which is the ratio between standard deviation and arithmetic mean of log transformed K_{sat} data, between 1.09 and 14.4. A high COV value represents higher spatial variability of data, which is substantial in this case.

Table 12. Summary of infiltration measurements in swales of different types of soil. HSG is the hydrologic soil group. (Conversion: 1 *in/hr* = 2.54 *cm/hr*)

Location	Soil type	Number of measurements	Geometric mean of K_{sat} , (<i>in/hr</i>) and (HSG)	Coefficient of variation (COV)* of K_{sat}
Hwy 47	Loamy sand & Sandy loam	20	0.8 (B)	3.55
Hwy 51	Loam & Sandy loam	20	0.8 (B)	3.34
Hwy 212	Silt loam & Loam	20	0.3 (C)	14.38
Hwy 13	Loam, Sandy clay loam & Silt loam	19	1.56 (A)	5.60
Hwy 77	Loamy sand	17	0.82 (B)	1.09

*COV = standard deviation of log transformed data/mean of log transformed data

iv. Effect of Season

Measurements taken at three swales (Hwy 212, Hwy 47 and Hwy 51) of different soil types for two seasons are given in Table 13. The geometric mean of K_{sat} was higher in Spring 2012 for Hwy 51 and lower for Hwys 47 and 212. The differences, however, were not great compared to the COV of each data set. Since the number of measurements is the same for each treatment, two factor ANOVA tests were performed on season. No significant difference was observed between the geometric mean K_{sat} values of fall and spring with 50 to 95% confidence intervals, partially due to the high COV (1.92~14.38). No significant difference in K_{sat} can be observed between seasons.

Table 13. Summary of infiltration measurements in Fall 2011 and Spring 2012.
(Conversion: 1 in/hr = 2.54 cm/hr)

Location	Fall 2011			Spring 2012		
	Number of measurements	Geometric mean of K_{sat} , (in/hr)	COV* of K_{sat}	Number of measurements	Geometric mean of K_{sat} , (in/hr)	COV* of K_{sat}
Hwy 47	20	0.8	3.55	20	0.61	4.94
Hwy 51	20	0.8	3.34	20	1.13	1.92
Hwy 212	20	0.3	14.38	20	0.18	2.66

*COV = standard deviation of log transformed data/mean of log transformed data

v. Effect of Moisture Content

Initial moisture content should not affect the measurement of K_{sat} . A K_{sat} measurement program was set up to test this assumption. At each swale infiltration measurements were repeated in order to result in different initial moisture contents. Seven swales at three different highways were chosen, as given in Table 14. The results of this investigation are mixed. At Hwy 51 (south), K_{sat} went up with an increase in initial moisture content. At Hwy 47(south) and Hwy 212 (west), K_{sat} went down with an increase in initial moisture content. At Hwy 47 (center), Hwy 51 (north), Hwy 52 (center) and Hwy 212 (center), K_{sat} went both up and down with an increase in initial moisture content. It appears that initial moisture content does not substantially affect K_{sat} in any given manner, as was expected for the Green-Ampt analysis technique.

The Tukey-Kramer test for significant differences confirms the above result. The test resulted in no significant difference among geometric mean K_{sat} value in the same swale for different moisture content within 90 to 99% confidence interval, with the exception of one swale (Hwy 47, center). In this swale the geometric mean K_{sat} for moisture content of 0.37 was statistically

significantly different than the geometric mean K_{sat} for moisture content of 0.43. These statistical tests confirm that saturated hydraulic conductivity is a soil property that, with the Green-Ampt analysis, is relatively unchanged with the change of moisture content.

Table 14. Summary of infiltration measurements with different moisture content.
(Conversion: 1 *in/hr* = 2.54 *cm/hr*)

Location	Initial moisture content	Number of measurements	Geometric mean of K_{sat} , (<i>in/hr</i>)	COV* of K_{sat}
Hwy 47 (center)	31	21	0.67	4.16
	37	21	1.54	1.49
	43	19	0.34	15.21
Hwy 47 (south)	23	21	4.59	0.58
	10	21	6.8	0.24
Hwy 51 (north)	27	18	2.6	0.78
	30	18	2.28	0.87
	32	18	0.84	2.58
Hwy51 (center)	33	21	0.7	3.36
	30	21	1.78	0.8
	15	18	1.12	1.87
Hwy 51 (south)	29	21	0.8	2.85
	38	21	1.6	1.23
Hwy 212 (center)	18	21	0.18	2.39
	39	12	0.11	0.93
	19	21	0.42	32.41
Hwy 212 (west)	12	21	2.61	1.06
	29	21	0.81	2.80

*COV = *standard deviation of log transformed data/mean of log transformed data*

vi. Effect of Distance from Downstream Outflow Pipe:

At each of three highways three swales were selected where 41 - 61 infiltration measurements were taken at different distances from the outflow pipe. The purpose was to test whether sediment that would be eroded in the swale during a large storm could settle near the outlet pipe. The geometric mean value of each swale was calculated and a Tukey-Kramer test was performed to see if there are significant differences among geometric mean K_{sat} for three

different swales of the same highway for 90, 95 and 99% confidence intervals. From this test it was observed that in some cases there is a significant difference among the geometric mean K_{sat} of the same highway and in some cases there is no significant difference. For further investigation, regression analysis was performed on these data. For all three highways it was found that there is significant difference in at least one of the geometric mean K_{sat} among three swales. The regression equations for all three swales are as follows:

$$\text{Hwy 47: } K_{sat} = 10^{1.45-0.98x} \quad (8)$$

$$\text{Hwy 212: } K_{sat} = 10^{1.12-1.16x} \quad (9)$$

$$\text{Hwy 51: } K_{sat} = 10^{-0.97+0.85x} \quad (10)$$

where, x is the distance from upstream (*miles*), and K_{sat} is the saturated hydraulic conductivity (*in/hr*). Over the length of each swale in the downstream direction (approximately a quarter mile), the geometric mean K_{sat} of Hwy 47 decreased by a factor of 1.8, the geometric mean K_{sat} of Hwy 212 decreased by a factor of 2.0 and the geometric mean K_{sat} of Hwy 51 increased by a factor of 1.6.

We can conclude that in some cases geometric mean K_{sat} will increase and in some cases geometric mean K_{sat} will decrease downstream. In general, one might expect to have lower K_{sat} value downstream because of sedimentation. However, vegetation density, erosion, presence of macropores, soil bulk density and soil compaction can have an effect on the K_{sat} values which might increase or decrease the geometric mean K_{sat} value downstream. As a result, there was no consistent trend to have K_{sat} decrease with distance in the downstream direction, and thus no evidence of sedimentation reducing K_{sat} in the sedimentation zones. It is possible that the grass growth and associated macropores kept the soil permeable to water, such that the effects of sedimentation on infiltration rates would be reduced.

2c. Comparison Between Monitored and Predicted Flow Rate of a Selected Swale

The stormwater runoff and routing model was verified by comparing the predicted runoff-rainfall ratio V_{out}/V_{rain} with the monitored runoff-rainfall ratio V_{out}/V_{rain} of one selected

swale, where V_{out} is the volume of water that leaves the study reach during a storm and V_{rain} is the volume of rainfall in the catchment during the storm. The swale selected for this study is located near Hwy 51 at Madison, WI, north of Hwy 12/18. The precipitation, inflow and outflow were monitored by the U.S. Geological Survey, and were available for this swale. The swale is divided into two reaches, which will be termed the upstream swale and downstream swale.

The upstream swale receives surface runoff from the roadway on the east side. On the west side the swale receives water through a pipe as a concentrated flow that discharges street

runoff. In the downstream swale a barrier is installed on the east side, which prevents the downstream swale from receiving any runoff from roadway. There are three monitoring flumes: one to monitor concentrated flow on the west side, one to monitor discharge from the upstream swale and one to monitor discharge from the downstream swale. A schematic diagram of the swale is shown in Figure 13. For this study, 108 infiltration measurements were taken at this site (Figure 14).

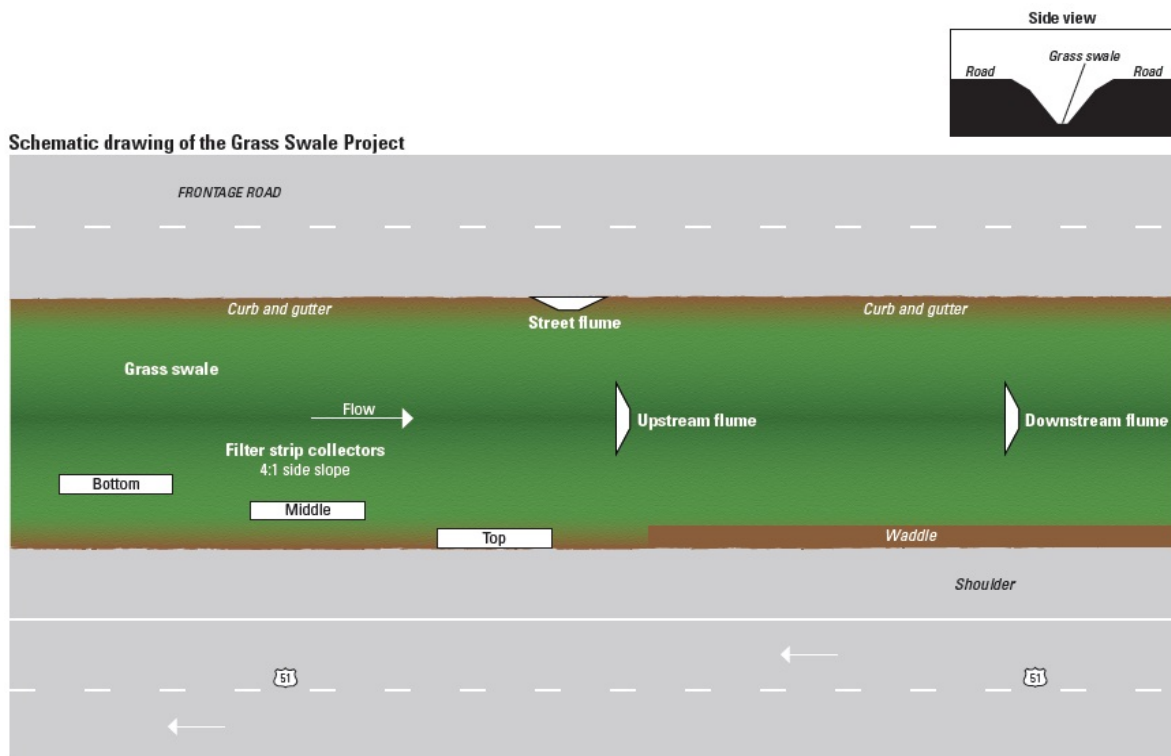


Figure 13. Schematic diagram of the swale located in Madison, WI. The waddle is a barrier that prevented flow off of Hwy 51 into the downstream swale.



Figure 14. Taking infiltration measurements using the MPD Infiltrometer in the swale located near Hwy 51, Madison, WI.

The water balance model of [equation 2](#) through [7](#) was tested using several rainfall events. The infiltration model component of the water balance was parameterized using the K_{sat} and ψ data collected in the field on 7/16/12 and on 7/17/12.

[Figure 15](#) shows the plan view of the swale which includes the grids for infiltration measurements. At each cross section, a grid was assigned to that area so that each cell is 78 ft in the longitudinal direction by 4 ft in width, keeping the infiltration measurement location at the center of each cell. It will be assumed that the K_{sat} value of each cell will be the same as the K_{sat} value measured at the center of the cell. Looking downstream, the east side of the swale receives surface runoff from Hwy 51. The west side of the swale does not receive water because of the presence of a curb near the highway, except for an inlet near the downstream weir that brings concentrated water off of the road. Starting from the right side of the swale, the amount of infiltration and runoff of the cell closest to the highway was calculated and then the remaining runoff was passed on to the next cell downhill.

After summing up total infiltration and runoff for the upstream swale, it was found that the entire amount of surface runoff, for all the events that were analyzed in the upstream swale, infiltrated into the side slope of the swale, so no water reaches the center of the swale. This is because the K_{sat} value is high, with a geometric mean of 7.5 in/hr . But from the monitored data, flow was observed through the flume (upstream flume). The source of this flow is believed to be the concentrated flow through the pipe located near the flume, which is termed as “street flume” in [Figure 15](#), that discharges street runoff.

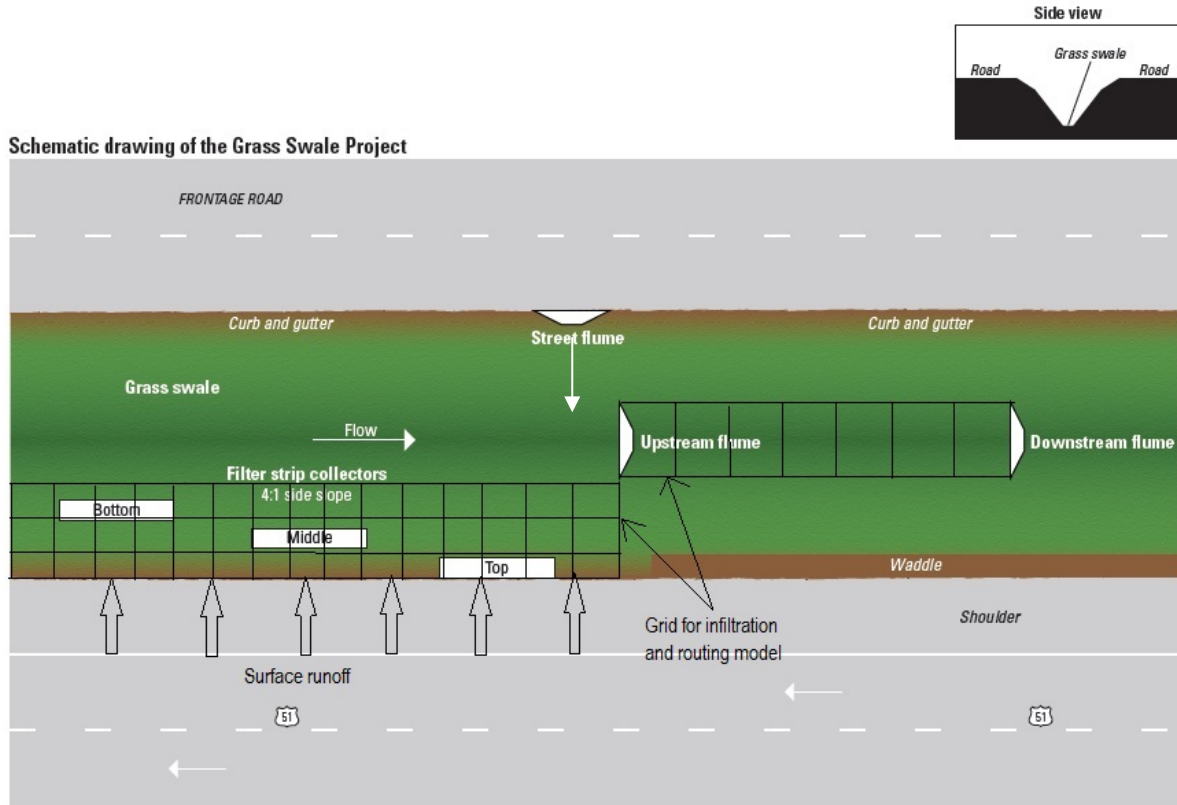


Figure 15. Schematic diagram of the swale located near Hwy 51, Madison, WI showing grid (78' x 4') and direction of flow.

The infiltration model (equation 7) and water balance equation (equation 6) were then used to model the downstream swale, which receives water only from the upstream swale through the flume (upstream flume). The geometric mean of K_{sat} of the downstream swale is 2.8 in/hr, the geometric mean of soil suction is 1.2 inches. The volume of water that flows through the downstream flume was predicted with the model, and the predicted runoff-rainfall ratio V_{out}/V_{rain} was compared with the monitored runoff-rainfall ratio V_{out}/V_{rain} for several rainfall events, which is summarized in Table 15.

Table 15. Comparing the predicted and monitored runoff-rainfall ratio. (Conversion: 1 *inch* = 2.54 *cm*; 1 *ft*³ = 0.0283 *m*³)

Rainfall event	Initial moisture content (%)	Total rainfall (<i>in</i>)	Rainfall intensity* (<i>in/hr</i>)	Monitored data				Predicted data		Monitored Vol. of water / Pred. Vol. of water
				Vol. of water through Street flume (<i>ft</i> ³)	Vol. of water through up-stream flume (<i>ft</i> ³)	Vol. of water through down-stream flume (<i>ft</i> ³)	Runoff/ Rainfall	Vol. of water through down-stream flume (<i>ft</i> ³)	Runoff/ Rainfall	
4/20/12	7	0.23	0.115	188	147	80	0.020	25	0.006	3.33
5/3/12	32	0.14	0.19	92	89	41	0.001	59	0.002	0.5
5/6/12	35	0.97	0.49	365	469	355	0.002	370	0.002	1
7/18/12	3	0.78	0.45	388	199	5	Close to 0	25	Close to 0	1
10/14/12	-	1.17	0.22	740	670	277	0.034	186	0.017	1.5
10/22/12	-	0.49	0.125	255	129	154	0.018	18	0.002	9
10/25/12	-	0.49	0.22	300	257	450	0.053	137	0.016	3

* Total rainfall / duration of storm

The soil moisture content data for 10/14/12, 10/22/12 and 10/25/12 was not available, so the average moisture content from the available data was assumed (12.2%). Note that the water from the street flume exceeded the downstream swale’s infiltration rate, such that the initial soil moisture of the downstream swale would not be as important.

For the storms with the greatest rainfall intensity, 5/3/12, 5/6/12 and 7/18/12, the volume of water through the street flume was lower than through the upstream flume. One interpretation of this data is that the upstream swale did not infiltrate all of the water in these storms. The differences on 7/18/12 were relatively small, and could be attributed to measurement error.

For rainfall events on 5/3/12, 5/6/12, 7/18/12 and 10/14/12 the monitored and predicted Runoff/Rainfall ratio is relatively close, especially considering that the ratio of volume of runoff to the volume of rainfall is so low, and any measurement errors would be magnified. For the 10/22/12 and 10/25/12 events, the measured volume of water in the downstream flume is higher than the volume of water in the upstream flume. This could be because of groundwater flow into the downstream swale or be due to measurement error.

2d. Prediction of Runoff from Minnesota Swales

The runoff model using [equation 7](#) was applied for Minnesota swales for a 1.1 *inch* 24 *hr* storm. Among five selected swales for infiltration measurements ([Table 8](#)) the swale located in

Hwy 212 was selected for this runoff analysis since this swale had the lowest geometric mean of K_{sat} values. The swale is located at the median of the highway, which has two lanes in each direction. The width of shoulder and each lane are 10 ft and 12 ft respectively and the ratio of impervious area to swale area is 1.7:1. Within a 15 ft long stretch the swale was divided into three cross sections in the longitudinal direction and each cross section was divided into eight cells. Twenty-four infiltration measurements were taken in this swale with the spatial variation of K_{sat} values are shown in Figure 16. The rainfall and stormwater that does not infiltrate along the cross section side slope is infiltration excess and reaches the center of the swale. By applying the runoff model it was computed that the entire volume of runoff infiltrated along the side slope of the swale, with an impervious area to side slope area ratio of 2.3:1. No water reaches at the center of the swale. Since Hwy 212 has the lowest geometric mean of K_{sat} values among five swales it is assumed that the rest of the swales are also able to infiltrate a 1.1 inch 24 hr storm.

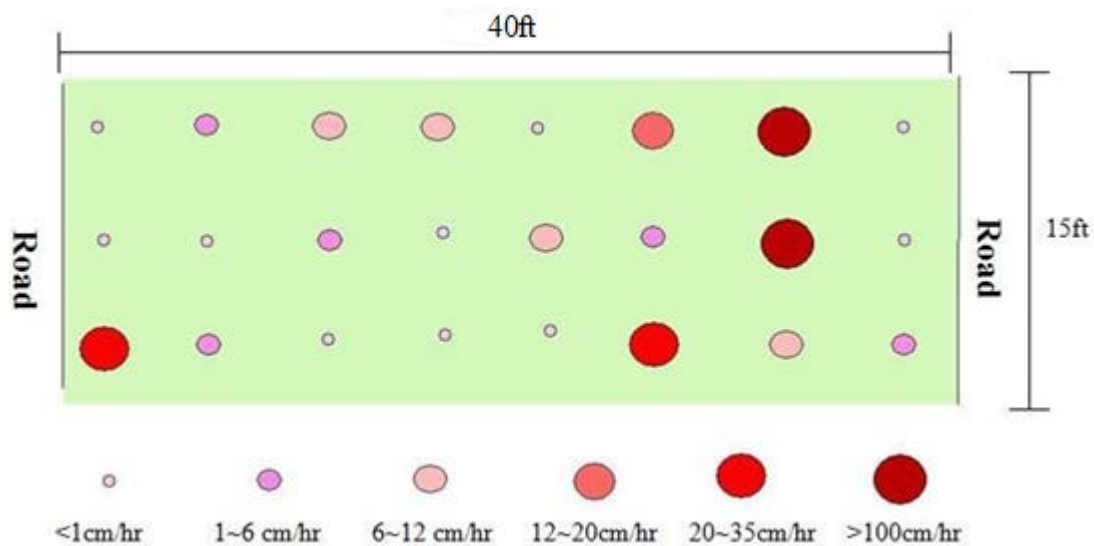


Figure 16. A plan view of the spatial variation of K_{sat} values in swale located in Hwy 212.

IV. Impact of Saturated Hydraulic Conductivity on the TR 55 Method (Task 7)

The impact of using saturated hydraulic conductivity, K_{sat} , to determine the Natural Resources Conservation Service (NRCS) curve number (CN) for a watershed modeled, was also investigated in this study.

1. Background

The NRCS Technical Release 55 (TR 55) (NRCS 1986) describes a method used by agencies across the United States and the world to model watershed runoff based on 24-hour precipitation amounts, land use and cover, and soil type. In TR 55 the depth of runoff, R , is estimated for pervious surfaces by:

$$R = \frac{(P-0.2S)^2}{(P+0.8S)} \quad (11)$$

where R = depth of runoff (*mm or inches*), P = precipitation depth (*mm or inches*), and S = the maximum depth of water storage in the watershed (*mm or inches*). The maximum depth of water storage, S , is the depth of rainfall the watershed can hold, adsorb, or otherwise abstract after runoff begins. In other words, it is the potential maximum abstractions after runoff begins. The value of S is estimated by:

$$S = \frac{1000}{CN} - 10 \quad (12)$$

where CN = the watershed curve number (or composite curve number). Curve numbers are a function of hydrologic soil group (HSG) (A, B, C, or D) and land use/cover and values are given in the TR 55 manual. For example, some CN values for urban areas are given in [Table 16](#).

[Equation 11](#) is derived by assuming:

$$\frac{F}{S} = \frac{R}{P-I_a} \quad (13)$$

where F = abstractions after runoff begins (*mm or inches*) and I_a = initial abstractions (*mm or inches*), which are defined as abstractions that occur before runoff begins. Also, by mass balance, it is known that:

$$P = F + I_a + R \quad (14)$$

and finally it is assumed that, for pervious soils, initial abstractions are 20% of S so that:

$$I_a = 0.2S \quad (15)$$

Substituting [equations 14](#) and [15](#) into [13](#) and rearranging to solve for R results in [equation 11](#).

Table 16. Example CN values for urban areas (NRCS 1986).

Cover description	Average percent impervious area ^{2/}	Curve numbers for hydrologic soil group			
		A	B	C	D
<i>Fully developed urban areas (vegetation established)</i>					
Open space (lawns, parks, golf courses, cemeteries, etc.) ^{3/} :					
Poor condition (grass cover < 50%)		68	79	86	89
Fair condition (grass cover 50% to 75%)		49	69	79	84
Good condition (grass cover > 75%)		39	61	74	80
Impervious areas:					
Paved parking lots, roofs, driveways, etc. (excluding right-of-way)		98	98	98	98
Streets and roads:					
Paved; curbs and storm sewers (excluding right-of-way)		98	98	98	98
Paved; open ditches (including right-of-way)		83	89	92	93
Gravel (including right-of-way)		76	85	89	91
Dirt (including right-of-way)		72	82	87	89

The TR 55 method is simple and easy to use and may be at least part of the reason for its widespread use. The method, however, should be considered accurate only to within +/- 30% for large, un-gaged watersheds and this accuracy decreases for smaller watersheds. Fennessey and Hawkins (2001) tested 37 watersheds and found that, compared to the TR 55 model results, runoff was either under or over predicted by more than 30% in 25 of the 37 watershed modeled, with 7 in error by up to several hundred percent and one by over 1300%.

The error in the model is based on several factors. For example, the assumption that initial abstractions are 20% of S was proposed as early as the 1950's and was based on runoff data from large and small watersheds across the United States (Plummer and Woodward undated). This relationship, however, is not valid for impervious surfaces, where $I_a \sim 0.1 in$. Other inherent assumptions include the soil moisture content at the beginning of the rainfall event. Typically, CN values such as those presented in Table 16 are for average moisture conditions. To incorporate variability in initial soil moisture, the TR 55 method adjusts CN's to a higher (for wet conditions) or lower value (for dry conditions). If the amount of rain received during the 5 days prior to the storm event of interest is more than (for dry conditions) or less than (for wet conditions) a corresponding threshold there is no adjustment made to the CN due to the antecedent moisture conditions. When an adjustment is made, it is not based on the actual amount of water in the soil and does not vary with rainfall amount once the threshold is surpassed. Also, others have claimed that the 5-day rainfall index is a subjective standard and not based on physical reality (Hope and Schulze 1982) and that it represents nothing more than error bands around the average conditions (Fennessey and Hawkins 2001). Finally, as a last example (but not an exhaustive list) of sources of error in the method, the relationship in equation 13 is only a generalization and is not an exact equality. Yet, it is considered as such in the derivation of equation 11. These and other issues (such as simplifying the runoff process)

lead to errors in TR 55 modeling results. The reader is urged to read the TR 55 manual to gain a full understanding of these assumptions and limitations so that the TR 55 method and the results are not applied incorrectly.

When using the TR 55 method, the hydrologic soil group (HSG) A, B, C, and D of a particular parcel of land is determined based on upper soil conditions. HSG values are usually found in United States Department of Agriculture (USDA) Soil Survey's or online at: <http://websoilsurvey.sc.egov.usda.gov>. Soils with a HSG rating of A have high permeability, B soils have slightly lower permeability values but are still considered well-drained, and C and D type soils are considered poorly drained with D soils having the lowest permeability. In general, highly permeable soils have low CN values and low permeable soils have high CN values. The maximum CN value is 100, which results in an S value of zero, but in practical terms the maximum CN value is 98 because even concrete surfaces, roads, roofs, and other impermeable surfaces can withhold a small amount of water and must be wetted before runoff occurs.

In summary, when the HSG and land use of a watershed is known, the CN can be determined. With the CN known, the value of S can be determined from [equation 12](#) and, with S , the depth of runoff, R , for a given precipitation depth, P , can be estimated using [equation 11](#). When a watershed contains multiple HSGs and land uses (and the land/use and soil type are evenly distributed throughout the watershed) a weighted average curve number (or composite curve number, CCN) can be computed and the depth of runoff estimated using the CCN.

2. Impact of Saturated Hydraulic Conductivity on the TR 55 Method

If the hydraulic saturated conductivity, K_{sat} , of the soil in a watershed was increased, one would expect less depth of runoff for a given precipitation amount because more water would infiltrate into the soil. This section describes how an increase in K_{sat} would impact the depth of runoff results (*i.e.* [equation 11](#)) of the TR 55 method.

If K_{sat} of a soil were increased the permeability of the soil would also increase and the HSG rating may be improved (e.g. move from D to C or from C to B, etc.) if the increase in K_{sat} were large enough. According to the NRCS (2007), soils with a depth to a water impermeable layer and high water table of more than 40 *inches* (100 *cm*) have HSGs that vary with K_{sat} according to [Table 17](#).

Table 17. Criteria for assignment of hydrologic soil groups when any water impermeable layer exists at a depth greater than 40 inches (NRCS 2007).

Hydrologic soil group	Hydraulic conductivity, K_{sat} (in/hr)
A	> 1.42
B	> 0.57 to ≤ 1.42
C	> 0.06 to ≤ 0.57
D	≤ 0.06

Thus, in order to determine the impact of increasing K_{sat} on runoff depths as estimated by the TR 55 method, the HSG and thus CN for a given land use from Table 16 can be varied as K_{sat} varies. For example, open space in good condition would have a HSG of D and a CN of 80 as long as the K_{sat} value was less than or equal to 0.06 in/hr. When the K_{sat} value was larger than 0.06 in/hr and less than or equal to 0.57 in/hr, the HSG would be C and the CN would change from 80 to 74. Increasing the value of K_{sat} would eventually cause the soil to have a HSG of B and then, when K_{sat} was larger than 1.42 in/hr, the HSG would be A and any further increase in K_{sat} would not affect the HSG or TR 55 runoff depth estimate.

Three scenarios or watershed land uses were investigated. Because the topic of this research project is roadside swales, the scenarios that were investigated for the impact of varying K_{sat} on the TR 55 estimated runoff depths were scenarios that could be used to represent roadways and/or swales. The investigated scenarios were 1) Open space in good condition (i.e. grass covering more than 75% of the area), 2) Paved road with open ditch, and 3) Gravel road. Curve number values used for the paved road with open ditch scenario were taken from NRCS (1986), which assumes pervious areas are open spaces in good condition and, for this scenario, assumed 75% of the area was impervious. For each investigation K_{sat} was varied from 0.01 in/hr to 2.0 in/hr and the precipitation depth was varied from 1 inch to 6 inches.

3. Results of Application to Roadside Runoff

The runoff depths estimated by equation 11 for K_{sat} values ranging from 2.0 in/hr down to 0.01 in/hr (i.e. HSGs A through D) for each of the three scenarios are shown in Figure 17 through Figure 19. Each figure shows runoff depths for HSGs A, B, C, and D for precipitation depths varying from 1 to 6 inches. HSG A corresponds to K_{sat} values greater than 1.42 in/hr, HSG B to K_{sat} values less than or equal to 1.42 in/hr and greater than 0.57 in/hr, and so on according to the information presented in Table 17. The calculations correspond to any watershed with the described soil type and land use and are applicable to roadside swales of any length. For all scenarios and all precipitation depths, runoff depths decrease as the HSG changes from D to C, from C to B, etc. with increasing values of K_{sat} . It should be noted that one listed limitation of the TR 55 method is that the method is not accurate for runoff depths less than 0.5 inches. When this occurs it is suggested that the TR 55 method not be used. Some results in which

runoff is less than 0.5 inches will still be discussed in this report but the reader should keep this limitation in mind.

3a. Effect of Increasing Saturated Hydraulic Conductivity on Runoff Depths Estimated by the TR 55 Method for Open Space in Good Condition

As shown in Figure 17 and Table 18 for open space in good condition, with a precipitation depth of 6 inches the runoff depth decreases from 3.8 inches (HSG D), to 3.2 inches (HSG C), to 2.0 inches (HSG B) and 0.4 inches (HSG A). This is a reduction in the estimated runoff depth of 16%, 53%, and 89%, as the HSG moves from D to C, then to B, and finally to A, respectively. For a precipitation depth of 4 inches, the runoff depth decreases from 2.0 inches (HSG D) to 1.6 inches (HSG C) to 0.8 inches (HSG B) and finally to zero inches (HSG A). This is a reduction in runoff depth of 20%, 60%, and 100%, respectively (all values are relative to the runoff depth for HSG A). For precipitation depths of 2 inches or less the runoff depths are approximately 0.5 inches or less. Thus, these results are graphed in Figure 17 but not discussed.

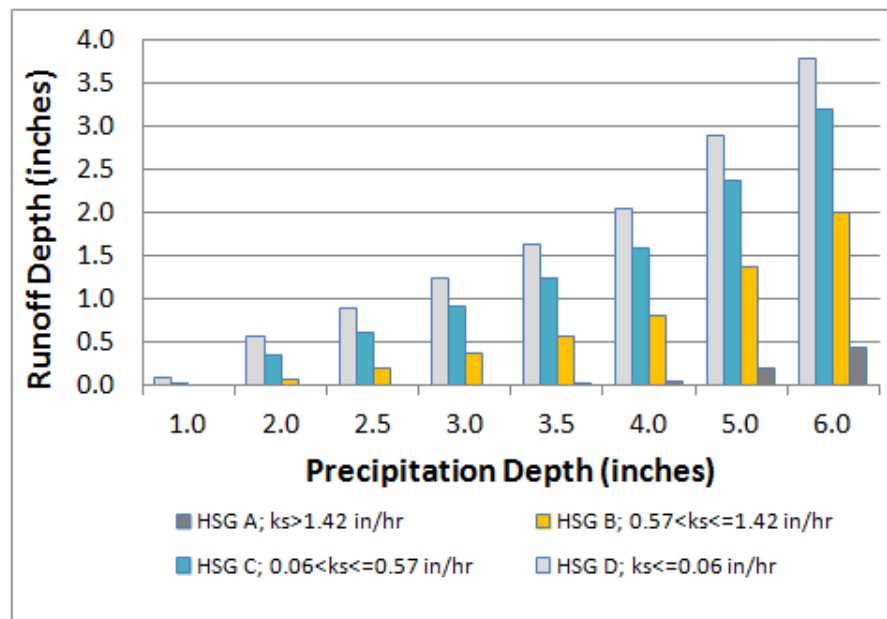


Figure 17. Effect of increasing saturated hydraulic conductivity on runoff depths estimated by the TR 55 method for open space in good condition.

Table 18. Effect of increasing saturated hydraulic conductivity on runoff depths estimated by the TR 55 method for open space in good condition. (Conversion: 1 inch = 2.54 cm)

	Listed values are Runoff Depth (inches)							
	Precipitation Depth (in)							
	6.0	5.0	4.0	3.5	3.0	2.5	2.0	1.0
HSG A; $K_{sat} > 1.42$ in/hr	0.45	0.20	0.05	0.01	0.00	0.00	0.00	0.00
HSG B; $0.57 < K_{sat} \leq 1.42$ in/hr	2.01	1.37	0.81	0.57	0.37	0.20	0.07	0.00
HSG C; $0.06 < K_{sat} \leq 0.57$ in/hr	3.18	2.36	1.60	1.24	0.91	0.61	0.35	0.02
HSG D; $K_{sat} \leq 0.06$ in/hr	3.78	2.89	2.04	1.64	1.25	0.89	0.56	0.08

3b. Effect of Increasing Saturated Hydraulic Conductivity on Runoff Depths Estimated by the TR 55 Method for a Paved Road with Open Ditch

As shown in Figure 18 and Table 19 for the paved road with open ditch scenario with 6 inches of precipitation, runoff depths vary from 5.2 inches (HSG D) to 5.1 inches (HSG C) to 4.7 inches (HSG B) and to 4.1 inches (HSG A). This is a reduction in the estimated runoff depth of 2%, 10%, and 21%, as the HSG moves from D to C, to B, and to A, respectively. For a precipitation depth of 2 inches, runoff depths vary from 1.3 inches (HSG D) to 1.2 inches (HSG C) to 1.0 inch (HSG B) and to 0.7 inches (HSG A). This corresponds to a 8%, 23%, and 46% reduction in runoff depth, respectively (all values are relative to the 1.3 inches of runoff depth for HSG D).

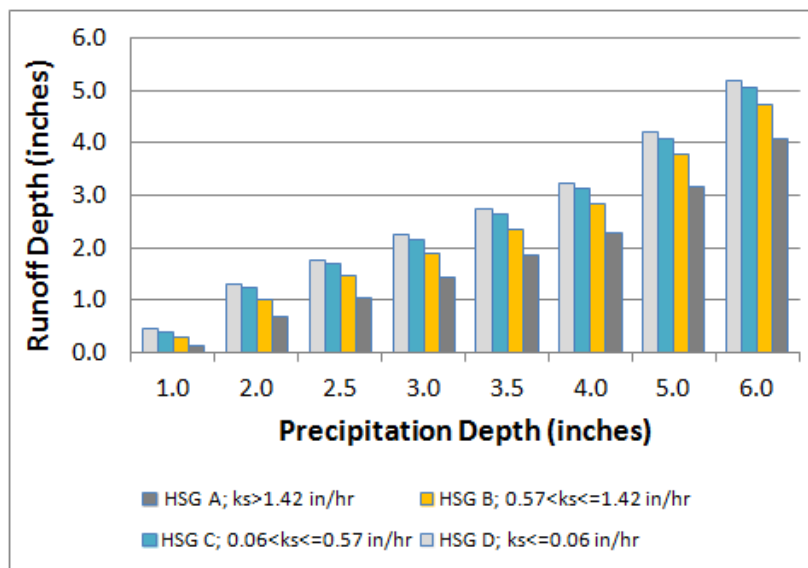


Figure 18. Effect of increasing saturated hydraulic conductivity on runoff depths estimated by the TR 55 method for a paved road with an open ditch.

Table 19. Effect of increasing saturated hydraulic conductivity on runoff depths estimated by the TR 55 method for a paved road with open ditch. (Conversion: 1 inch = 2.54 cm)

	Listed values are Runoff Depth (inches)							
	Precipitation Depth (in)							
	6.0	5.0	4.0	3.5	3.0	2.5	2.0	1.0
HSG A; $K_{sat} > 1.42$ in/hr	4.09	3.17	2.29	1.86	1.45	1.06	0.70	0.13
HSG B; $0.57 < K_{sat} \leq 1.42$ in/hr	4.74	3.77	2.82	2.36	1.90	1.45	1.03	0.28
HSG C; $0.06 < K_{sat} \leq 0.57$ in/hr	5.07	4.09	3.12	2.64	2.16	1.69	1.24	0.40
HSG D; $K_{sat} \leq 0.06$ in/hr	5.18	4.20	3.22	2.73	2.25	1.78	1.31	0.45

3c. Effect of Increasing Saturated Hydraulic Conductivity on Runoff Depths Estimated by the TR 55 Method for a Gravel Road

As shown in Figure 19 and Table 20 for the gravel road scenario with 6 inches of precipitation, runoff depths vary from 5.0 inches (HSG D) to 4.7 inches (HSG C) to 4.3 inches (HSG B) and to 3.4 inches (HSG A). This is a reduction in the estimated runoff depth of 6%, 14%, and 32%, as the HSG moves from D to C, to B, and to A, respectively. For a precipitation depth of 2 inches, runoff depths vary from 1.2 inches (HSG D) to 1.1 inches (HSG C) to 0.8 inches (HSG B) and to 0.4 inches (HSG A). This corresponds to a 8%, 33%, and 67% reduction in runoff depth, respectively (all values are relative to the 1.3 inches of runoff depth for HSG D).

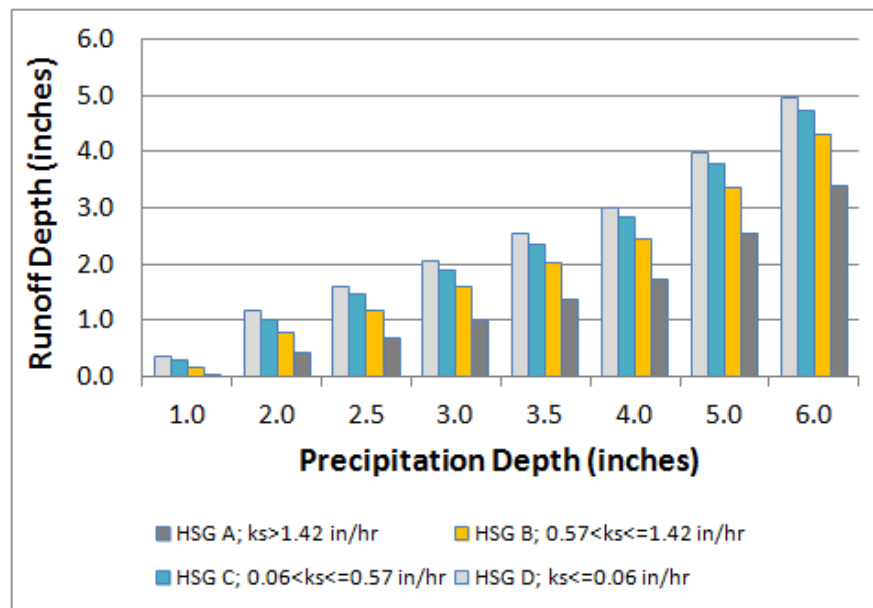


Figure 19. Effect of increasing saturated hydraulic conductivity on runoff depths estimated by the TR 55 method for a gravel road.

Table 20. Effect of increasing saturated hydraulic conductivity on runoff depths estimated by the TR 55 method for a gravel road. (Conversion: 1 inch = 2.54 cm)

	Listed values are Runoff Depth (inches)							
	Precipitation Depth (in)							
	6.0	5.0	4.0	3.5	3.0	2.5	2.0	1.0
HSG A; $K_{sat} > 1.42$ in/hr	3.38	2.54	1.74	1.37	1.02	0.69	0.41	0.04
HSG B; $0.57 < K_{sat} \leq 1.42$ in/hr	4.30	3.37	2.46	2.02	1.59	1.18	0.80	0.17
HSG C; $0.06 < K_{sat} \leq 0.57$ in/hr	4.74	3.77	2.82	2.36	1.90	1.45	1.03	0.28
HSG D; $K_{sat} \leq 0.06$ in/hr	4.96	3.98	3.02	2.54	2.07	1.61	1.16	0.36

The impact of increasing saturated hydraulic conductivity on open space in good condition was more significant than the impact on a paved road with an open ditch. This could be due, at least in part, to the fact that in the paved road scenario the abstractions of the impermeable surface of the road will not be increased by increasing the hydraulic conductivity. Thus, a significant portion of the watershed is not impacted by K_{sat} whereas in the open space scenario, 100% of the watershed is impacted.

When compared to the paved road with open ditch, the gravel road scenario was slightly more impacted by an increase in K_{sat} as the HSG moved from C to B and eventually to A. Again, this could be due to the fact that a large portion of the paved road watershed will not be impacted by an increase in K_{sat} because the paved surface is impermeable and its CN will remain at 98.

The impact of increasing K_{sat} was significantly more noticeable for the open space in good condition scenario than the gravel road scenario even though a gravel road, unlike a paved road, can infiltrate some water. In terms of the TR 55 method, this is due to the fact that as K_{sat} increases, the CN for an open space in good condition varies from 80 (HSG D) to 39 (HSG A) whereas the CN for a gravel road varies from 91 (HSG D) to 76 (HSG A). In practical terms, there will be more runoff from a gravel surface compared to a vegetated surface because the vegetation will tend to slow the runoff and allow more water to infiltrate.

V. Retaining Phosphate and Metals in Ditch Check Filters (Tasks 2 and 4)

1. Laboratory Batch Study of Enhancements (Task 2)

1a. Objectives

The objective of this task was to conduct a batch study on the capabilities of various media enhancements to remove dissolved zinc, cadmium, copper, lead, and phosphate. The enhancements were selected based on their demonstrated abilities to remove the target pollutants in several research studies (see Appendix A). The performances of the enhancements were determined using synthetic stormwater containing the target pollutants at levels typically found in natural stormwater runoff, based on data provided in a national-level study on stormwater runoff quality conducted by Maestre and Pitt (2005). The goal was to utilize the batch study results to select one or more enhancements for use in the pilot-scale study on ditch check filter.

1b. Potential Media Enhancement Materials

A total of 17 potential media enhancements including two iron-based materials (iron filings and steel wool), five ferrous oxide materials, one activated alumina material, six activated carbon materials, and C-33 sand were tested in the batch study. Some of the materials were chosen to target other pollutants of interest that were part of the batch study.

1c. Methods

For the batch study experiments, 500 mL capacity borosilicate glass bottles were used. The bottles were acid washed with 10% HCl, rinsed with ultrapure water (Milli-Q, 18.2 $M\Omega\cdot cm$), then acid-washed with 10% oxalic acid, rinsed again with ultrapure water, and then dried. Synthetic stormwater was prepared to represent median values for natural stormwater (Maestre and Pitt 2005) as listed in [Table 21](#).

Table 21. Characteristics of synthetic stormwater used in batch study.

Component	Target concentration	Source Compound
Cadmium (Cd^{2+})	0.5 $\mu g/L$	Cadmium Chloride
Copper (Cu^{2+})	8.0 $\mu g/L$	Cupric Sulfate
Lead (Pb^{2+})	3.0 $\mu g/L$	Lead Nitrate
Zinc (Zn^{2+})	112 $\mu g/L$	Zinc Chloride
Phosphate (PO_4^{3-})	120 $\mu g/L$	Potassium Hydrogen Phosphate
Alkalinity	135 mg/L as $CaCO_3$	Sodium Bicarbonate
Hardness	39 mg/L as $CaCO_3$	Magnesium Carbonate
pH	7.4	0.06 M Hydrochloric Acid
Conductivity	270 $\mu S/cm$	-

While the median conductivity in natural stormwater is $121 \mu\text{S}/\text{cm}$ (Maestre and Pitt 2005), the measured conductivity of the synthetic stormwater prepared was approximately $270 \mu\text{S}/\text{cm}$ due to the salts added to create the pollutant concentrations. The measured value is considered to be acceptable for the purpose of the study, and thus the conductivity was not adjusted in the synthetic stormwater.

For each media enhancement, three replicate bottles were used. Firstly, 500 mL of synthetic stormwater was added to each bottle. Using sterile plastic syringes, duplicate 20 mL samples were collected from each bottle as ‘initial’ samples to verify initial pollutant concentration and identify contaminated bottles. Then, 5 g of media was added to each bottle. A ‘blank’ bottle which did not contain any media was also included in the batch tests. All batch test bottles (with media) and blank (no media) were placed on a Labline Orbital shaker table set at 250 RPM for one hour (Figure 20). The ‘final’ samples were collected (as described above) after mixing for one hour. After collecting the final samples, pH and conductivity were measured again in each batch bottle.



Figure 20. Photograph showing sample bottles set up on a shaker table during the batch study experiments.

The initial and final samples were filtered through $0.45 \mu\text{m}$ membrane filters to remove particulates and stored in acid-washed glass sample bottles. The filtered samples were then acidified with 0.03 mL of 0.6 M HCl to adjust the pH to 2 or lower for laboratory analysis for soluble reactive phosphorus (phosphate) and metals. Samples were analyzed by the inductively coupled plasma- mass spectrometry method (Thermo Scientific XSERIES 2 ICP-MS) at the Research Analytical Laboratory at the University of Minnesota. If the measured pollutant concentration in a sample was below analytical detection, the detection limit value was assigned as the concentration for that sample.

1d. Results

i. Phosphate and Metals Removal Performance of the Enhancements

The results of the batch studies are shown in Figure 21. For each material, the pollutant removal is based on average of six sample concentration measurements. For final samples with concentrations at or below analytical detection, the pollutant removals are expressed as > 99%.

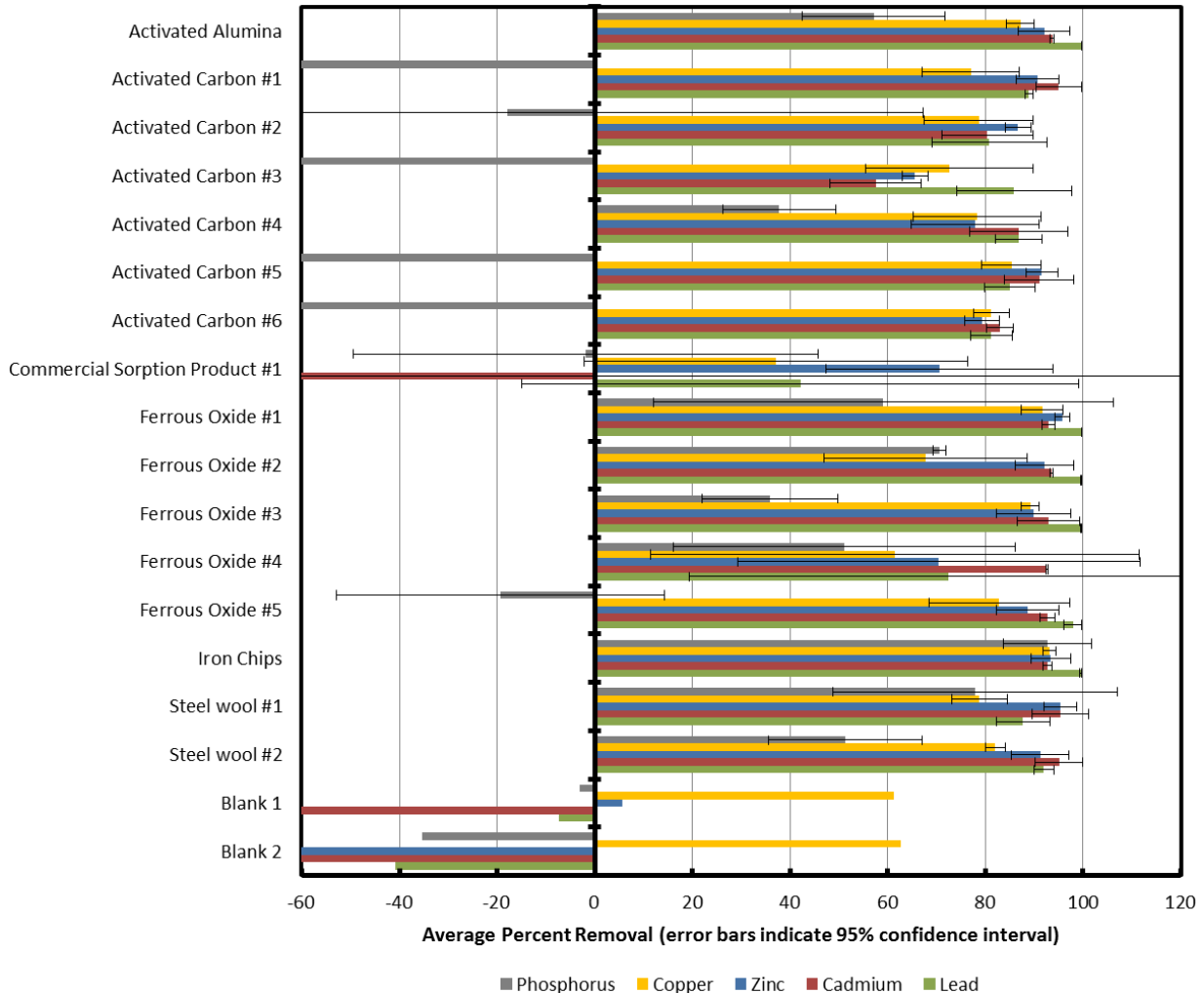


Figure 21. Batch study results on removal of phosphate and metals by various media enhancement materials. (Error bars are not shown for ‘Blank 1’ and ‘Blank 2’ since only one sample of each was included in the test).

Of the 17 enhancements tested, iron-based materials, activated alumina, and ferrous oxides showed the highest removal of phosphate and metals (Figure 21). While four ferrous materials reduced phosphorus and metals concentrations, one ferrous material exhibited

leaching of phosphorus. Activated carbon showed 77 – 95% removal for metals, however phosphate removal was poor exhibiting export of P (*i.e.* increased phosphate levels in the final samples) (Figure 21). Overall, iron-based materials (iron filings and steel wool) and activated alumina provided the highest potential for removal of both phosphorus and metals. The reductions in pollutant concentrations measured for iron-based materials and activated alumina are summarized in Table 22. While the metals removals were similar, iron materials exhibited higher phosphate removals compared to activated alumina. Therefore, the two iron materials can be considered to be the best media enhancements for high pollutant removal.

Table 22. Summary of phosphate and metals removals by the media enhancements that provided the highest pollutant removals during the batch study experiments.

Material	Percent pollutant concentration reduction				
	Phosphate	Copper	Lead	Zinc	Cadmium
Steel wool (2 types)	51 – 78	79 – 82%	88 – 92%	91 – 95%	95%
Iron filings	93%	93%	>99%	93%	93%
Activated alumina	57%	87%	>99%	92%	94%

ii. Impact on pH

The impact of addition of the media enhancement materials on the pH of the synthetic stormwater solution was assessed. The initial and final pH of the batch (bottle) solutions for all the enhancements are shown in Figure 22. The MPCA maximum pH for aquatic life and recreation waters was used as the threshold pH.

An increase in pH ranging between 0.44 and 1.9 units was noted for the materials tested. Activated carbon materials showed the highest increase in pH, resulting in the final pH exceeding the MPCA threshold of pH 9. The most effective materials for phosphate and metal removal (activated alumina, iron filings, ferrous oxides) increased the solution pH by 0.7 to 1.3 units. However, the final pH did not exceed the MPCA threshold for any of these materials.

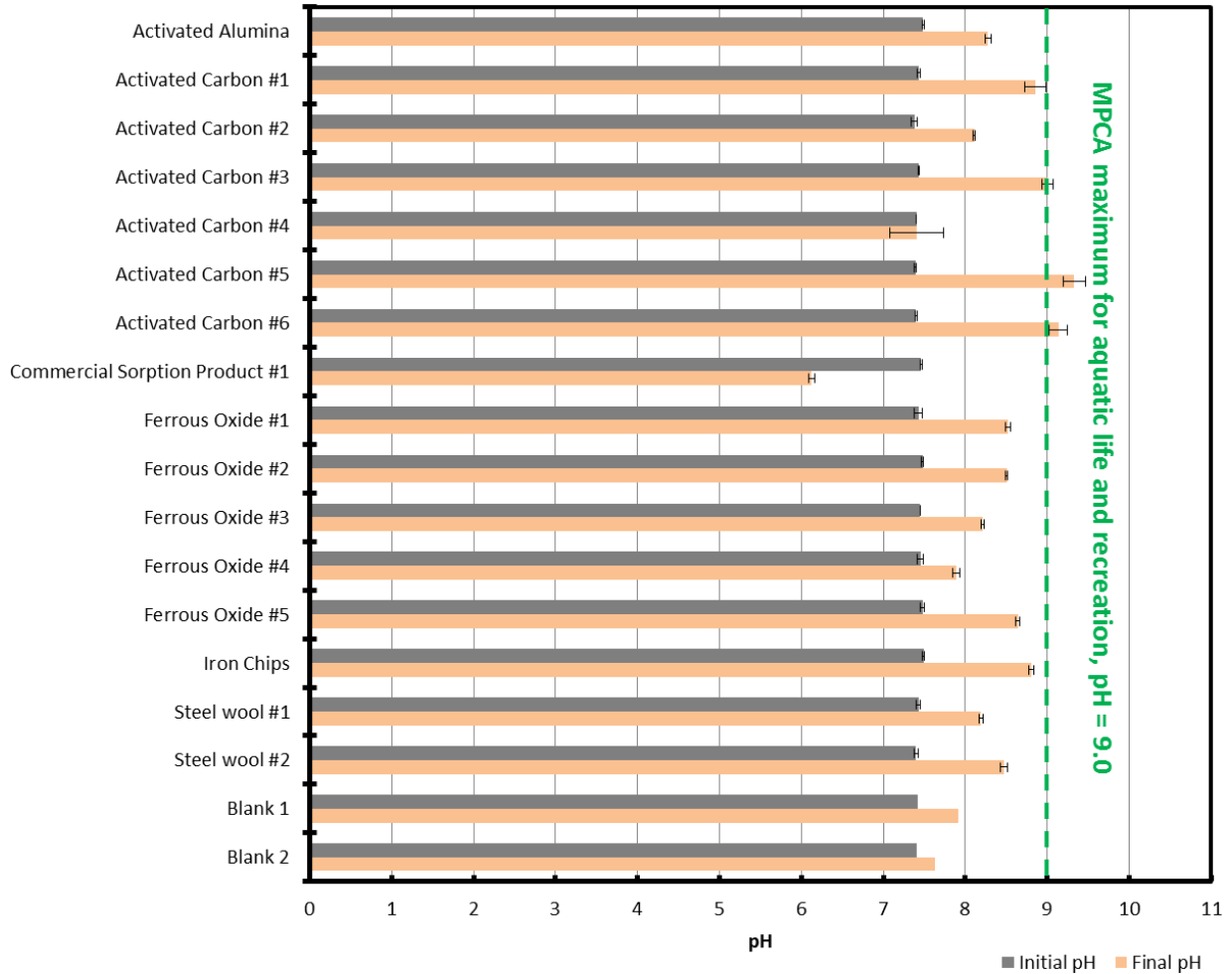


Figure 22. Effect on solution pH by various media enhancement materials tested in the batch study. (Error bars are not shown for ‘Blank 1’ and ‘Blank 2’ since only one sample of each was included in the test).

1e. Summary

Batch studies were conducted to determine the abilities of 17 different media enhancements in removing phosphate and heavy metals from water. The performance of an enhancement was evaluated based on the percent pollutant concentration reductions achieved and changes in pH observed. At the median concentrations and contact time of one hour, activated alumina and iron-based materials were most effective in removing metals, and iron-based materials provided the most effective phosphate removal in the synthetic stormwater. While activated alumina provided average removals of 57% for phosphorus and 87 – 99% for metals, the iron-based materials showed up to 93% removal for phosphorus and 79 – 99% for metals. The solution pH did not exceed the MPCA threshold of pH = 9 in the presence of these materials. Based on the batch study results, two iron-based materials, iron filings and steel wool, were selected as the media enhancement for the pilot-scale study.

2. Pilot Study (Task 4)

2a. Objectives

The objectives of the pilot-scale study were to design, install, and test a ditch check filter for removal of phosphate and metals (Cd, Cu, Pb, Zn) from stormwater runoff. The optimal design conditions determined from the literature review and the batch studies conducted earlier were incorporated in the pilot study. Through the pilot experiments, the goal was to determine the limits of hydraulic and pollutant loading rates for comparison to full-scale design parameters. Design conditions will be altered to determine optimal sizing and treatment criteria based on the results.

2b. Methods

The pilot experiments were conducted in a 20 *inch* flume at the St. Anthony Falls Laboratory (SAFL), University of Minnesota. Two ditch check filter prototypes were tested: a sand-iron filter and a steel wool filter. In the first filter prototype, the filter media consisting of a mixture of standard concrete sand (C-33) and iron filings was filled in a non-woven geotextile fabric bag [apparent opening size (AOS) = 80 US std. sieve] to form a ‘filter sock’. The particle size gradations of C-33 sand and iron filings are provided in [Table 23](#). The filter sock was placed in the flume and rip-rap was placed at 5:1 slope on either side of the filter sock. Although field installations typically employ Class II riprap at 10:1 slope, the pilot test used 5:1 slope to accommodate the filter setup in the flume. The different inclination adopted in the flume test was believed to have a negligible effect on the flow rate and effectiveness of the filter. The dimensions of the sand-iron filter prototype were 10 *inches* height, 20 *inches* width, and 12 *inches* length (in the direction of flow). Two media configurations were tested: first containing 95% sand and 5% iron filings, and second containing 92.5% sand and 7.5% iron (by weight).

Table 23. Particle size distributions of C-33 sand and iron filings utilized for the filter media mix in the pilot tests.

C-33 Sand			Iron filings			
Particle Size (mm)	Percent of total mass	Percent passing	Particle Size (mm)	Percent of total mass	Percent Minimum passing	Percent Maximum passing
1.0	12	100	4.75	2.5	100	
0.5	31	88	2.36	15	95	100
0.42	4	57	1.18	47.5	75	90
0.25	37	53	0.60	30	25	45
0.10	16	16	0.30	2.5	0	10
0.074	0	1	0.15	2.5	0	5
0.05	0	0				

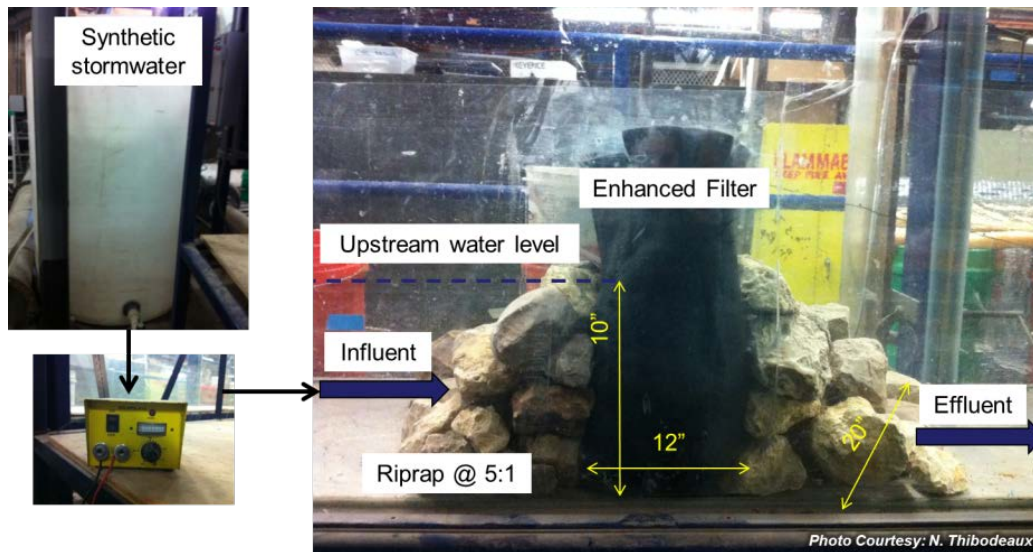


Figure 23. Experimental set up of the pilot-scale test on ditch check sand filter.

The experimental set up of the pilot test is shown in [Figure 23](#). For the test, synthetic stormwater was prepared in a 500 *gallon* tank by dosing tap water with appropriate amounts of phosphorus and metals and mixing the solution well. The composition of the synthetic stormwater was chosen to mimic the median pollutant levels in stormwater runoff reported by Maestre and Pitt (2005). Synthetic stormwater was pumped into the flume until the upstream water level reached the height of the filter (10 *inches*). The flow rate was then adjusted such that the upstream water level remained at this height. Once steady-state was achieved, water samples were taken from the upstream side (influent) and downstream side (effluent) every 20 *minutes*. The height of water in the flume and flow rate were measured periodically. Samples collected were filtered through a 0.45 μm membrane syringe filter to remove any particulates and then acidified for laboratory analysis for the target pollutant concentrations. Samples for metal analysis were acidified with trace metal grade nitric acid, and analyzed by the ICP-MS method. Samples for orthophosphate analysis were acidified to $\text{pH} = 2$ with sulfuric acid and refrigerated until laboratory analysis. Orthophosphate analysis was performed by ascorbic acid method in a Lachat analyzer (QuikChem FIA⁺ 8000 Series). The laboratory analytical detection limits of the phosphate and metals analyses methods are summarized in [Table 24](#). The containers used for collection and storage of samples were acid-washed, rinsed with distilled water, and dried prior to use.

Table 24. Laboratory analytical detection limits of phosphate and metals analyses methods.

	Phosphate	Copper	Lead	Zinc	Cadmium
Detection limit ($\mu\text{g/L}$)	1.0	0.12	0.01	0.1	0.01

The second ditch check filter prototype was constructed with steel wool (Figure 24). Steel wool reels of various grades (grades #0, #3, #0-3 combination, needle-punched, density 1200 and 2400 kg/m³; approx. 0.5 inches in thickness) were utilized. The grades are differentiated by the average width of steel wool fiber, for instance #00 is very fine (40 microns), #0 is fine (50 microns), and #3 is coarse (90 microns), and finer steel wool is denser.



Figure 24. Photograph of the steel wool filter installed in the flume for the pilot-scale study.

To construct the filter, the steel wool sheets were cut to a dimension of 14 inches length x 9 inches width, six such layers were enclosed in a non-woven geotextile fabric bag (AOS = 80 US std. sieve), and then encased by a layer of woven geotextile filtration fabric (AOS = 40 US std. sieve). This filter assembly (about 4 inches thick) was attached to an acrylic frame and installed in the 20 inch flume. The sides of the frame were tightly sealed with thum-gum to prevent leakage of water around the frame. No riprap was utilized with this filter setup. This is because the filter prototype was designed to be installed in an existing frame structure at a potential swales site. The synthetic stormwater used for the flume tests contained phosphate, lead, copper and zinc at median stormwater levels (Maestre and Pitt 2005); no cadmium was added to the mix. The experimental procedure including measurements, sample collection and laboratory analyses described for the sand-iron filter was followed for testing the steel wool filters. After the first test run was completed, the steel wool filters were air-dried for at least two weeks to allow rust formation and then tested again.

2c. Results

i. Performance of Sand-Iron Filter

Two ditch check sand filters (enhanced with 5% and 7.5% iron) were tested in the flume for a total of three runs. Two tests were conducted on the 5% iron filter (test durations 60 and 90

minutes, respectively) and one test was conducted on the 7.5% ion filter (test duration 105 minutes). The measured influent and effluent pollutant concentrations and pollutant removals data are summarized in Table 25. The metals removals were 10 – 45% for Zn (average = 25%) and 11 – 67% for Cd (average = 45%) during the tests conducted (Table 25). Increase in effluent concentrations (thus, negative removals) was observed for Cu, Pb and for some instances of Zn. Figure 25 shows the phosphate removal performance of the sand filters. The 5% and 7.5% iron filters showed phosphate removals ranging between 10 and 33% (average = 23%) during the tests.

Table 25. Measured influent (C_{in}) and effluent (C_{out}) pollutant concentrations and removals of phosphate and metals during the pilot tests on ditch check sand filter.

Filter	n	Phosphate			Zinc		
		C_{in} (mg/L)	C_{out} (mg/L)	Removal	C_{in} (μ g/L)	C_{out} (μ g/L)	Removal
5% Iron Test 1	3	0.45	0.36 – 0.41 (mean = 0.38)	11 – 21% (mean = 17%)	96	62 – 72 (mean = 67)	25 – 35% (mean = 30%)
5% Iron Test 2	8	0.44	0.29 – 0.35 (mean = 0.32)	21 – 33% (mean = 26%)	74	53 – 92 (mean = 61)	(-25) – 28% (mean = 30%)
7.5% Iron	8	0.45	0.33 – 0.38 (mean = 0.35)	16 – 26% (mean = 22%)	64	35 – 109 (mean = 62)	(-70) – 45% (mean = 3%)
Filter		Cadmium			Copper		
	n	C_{in} (μ g/L)	C_{out} (μ g/L)	Removal	C_{in} (μ g/L)	C_{out} (μ g/L)	Removal
5% Iron Test 1	3	0.11	0.05 – 0.08 (mean = 0.07)	25 – 53% (mean = 36%)	8.9	6.9 – 8.4 (mean = 7.8)	7 – 23% (mean = 14%)
5% Iron Test 2	8	0.65	0.22 – 0.58 (mean = 0.37)	11 – 66% (mean = 43%)	23	38 – 78 (mean = 59)	(-238) – (-87)% (mean = -153%)
7.5% Iron	8	0.70	0.23 – 0.54 (mean = 0.36)	23 – 67% (mean = 48%)	14	7.1 – 53 (mean = 59)	(-273) – 50% (mean = -153%)
Filter		Lead			Iron		
	n	C_{in} (μ g/L)	C_{out} (μ g/L)	Removal	C_{in} (μ g/L)	C_{out} (μ g/L)	Removal
5% Iron Test 1	3	5.9	4.2 – 5.2 (mean = 4.9)	11 – 28% (mean = 16%)	36	62 – 72 (mean = 67)	6 – 18% (mean = 12%)
5% Iron Test 2	8	0.98	0.43 – 2.8 (mean = 1.5)	(-182) – 56% (mean = -52%)	55	25 – 232 (mean = 109)	(-322) – 56% (mean = -98%)
7.5% Iron		n/a	n/a	n/a	n/a	n/a	n/a

*Negative removals indicate leaching of pollutant from the media; n/a: no data; n = number of samples collected

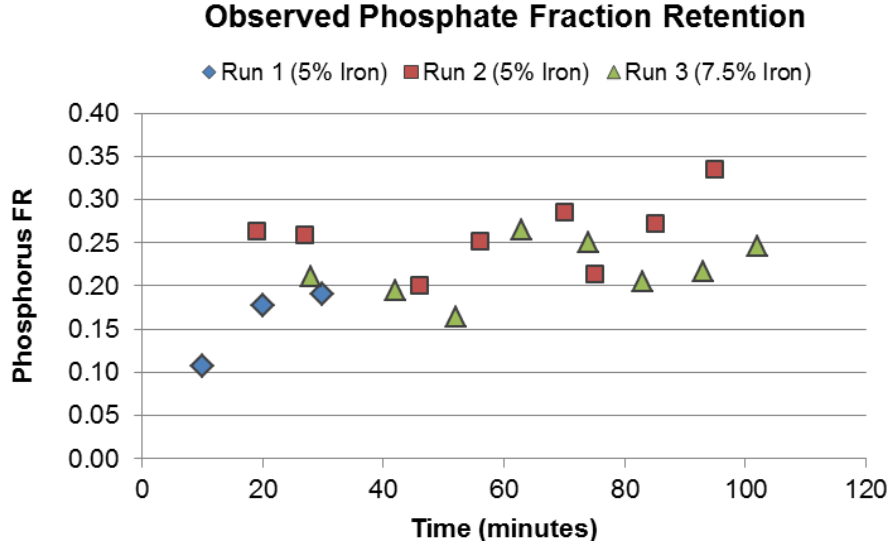


Figure 25. Retention of phosphate observed for the 5% iron ditch check sand filter during pilot tests. $FR = \left[\frac{(C_{in} - C_{out})}{C_{in}} \right]$

The observed phosphate removals were lower than expected, given that previous column study results showed 70 – 90% phosphate removal for the iron-enhanced sand filter (Erickson et al. 2005, 2007, 2012). During the flume tests, it was observed that a portion of the influent was bypassing or ‘leaking’ around the sides of the filter sock. This leakage was due to inadequate sealing of the filter sock against the glass walls of the flume. Attempts to eliminate the leakage were unsuccessful. The leakage resulted in mixing of the leakage (untreated water) with the water actually passing through the filter (treated water) on the downstream side of the filter. The effluent sampled was thus a mixture of treated and untreated water. Hence, it was hypothesized that the issue of leakage resulted in the low overall removals observed and the actual capacity of the ditch check filter to retain phosphate and metals could be higher.

In an effort to assess the actual effectiveness of the filter, computations were performed to quantify the leakage, flow through the filter, and pollutant concentrations in the leakage and treated water using the data collected. The flow rate through the filter (Q_{filter}) was estimated using Dupuit equation as follows:

$$q = \frac{K_{sat}}{2L} (h_o^2 - h_d^2) \quad (16)$$

$$Q_{filter} = q \times B \quad (17)$$

$$Q_{leakage} = Q_{total} - Q_{filter} \quad (18)$$

where, q is the specific discharge (ft^2/s), K_{sat} is the saturated hydraulic conductivity of the media (ft/s), h_o is upstream head (ft), h_d is downstream head (ft), L is length of filter in the

direction of flow (ft) and; B is the width of filter (ft). Leakage ($Q_{leakage}$) was then estimated as difference between measured total flow (Q_{total}) and estimated flow through filter (Q_{filter}) (equation 18).

The water velocity profile in the direction of flow and the travel time of water at each cross-section through the filter were computed. The cumulative sum of the travel times is the total contact time of water with the media ($t_{contact-media}$) in the ditch check filter.

Assuming the sampled effluent was well-mixed treated water and leakage, the measured effluent pollutant concentration ($\overline{C_{out}}$) can be expressed in terms of the flow-weighted pollutant concentrations in the treated and leakage water as:

$$\overline{C_{out}} = \frac{Q_{filter}C_{filter} + Q_{leakage}C_{leakage}}{Q_{total}} \quad (19)$$

where, $\overline{C_{out}}$ is the measured pollutant concentration (mg/L) in the effluent sampled, C_{filter} and $C_{leakage}$ are the pollutant concentrations (in mg/L) in the treated water and leakage, respectively.

The pollutant concentration in the treated water (C_{filter}) can be predicted using the pollutant retention model published by Erickson et al. (2005), as given in equation 20:

$$\frac{C_{filter}}{C_{in}} = 1 - [(\beta_0 e^{-\beta_1 \Sigma M})(1 - e^{-\beta_2 t_{contact}})] \quad (20)$$

where, C_{filter} = Treated effluent concentration (mg/L)

C_{in} = Influent concentration (mg/L)

β_0 = Coefficient related to the phosphate retention capacity of iron = 1.0 (for 5% steel)

β_1 = Coefficient related to the rate at which C_{out} approaches C_{in} (g Fe/g P) = 12 (for 5% steel)

ΣM = Instantaneous sum of phosphate mass retained (g P/g Fe)

β_2 = Coefficient related to the phosphate sorption rate constant (1/s) = 0.017 (for 5% steel)

$t_{contact}$ = Contact time with steel (s) = $\frac{\text{Volume of steel wool}}{\text{Flow rate}} = 131 \text{ s}$ (for 5% steel)

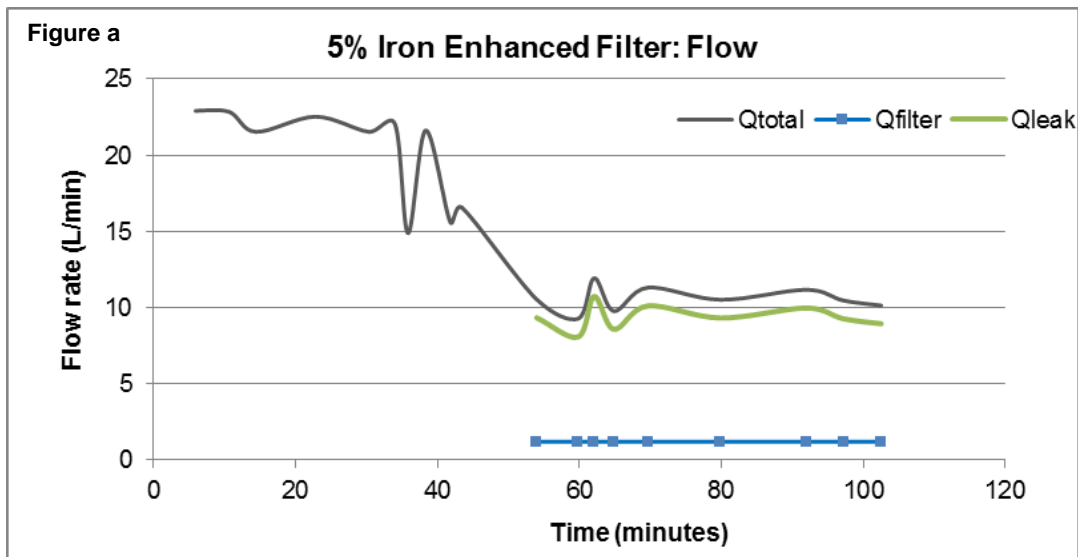
In equation 20, $t_{contact}$ is the residence time with steel only, expressed in terms volume of steel and flow rate (Erickson et al. 2005). However, similar computation method cannot be followed for the ditch check sand filter. The contact time with media (sand and steel/iron) is known for both Erickson et al. (2005) study and the pilot study. Therefore, a modification of the model in equation 20 is used where contact time with the media (sand and iron), $t_{contact-media}$, is used. In Erickson et al. (2005) study, the ratio of contact time with steel ($t_{contact}$) to contact time with media ($t_{contact-media}$) is 0.0793. Therefore, the exponent in second term of the model is ($\beta_2 \times 0.0793 \times t_{contact-media}$). Additionally, the flume tests conducted were short-term tests and the mass of phosphorus accumulated in the media (ΣM) can be assumed to be

insignificant, thereby approximating the exponential term ($e^{-\beta_1 \Sigma^M} \approx 1$). The resulting model to estimate the pollutant concentration in treated water is:

$$\frac{C_{filter}}{C_{in}} = 1 - [(1 - e^{-0.0013 t_{contact-media}})] \quad (21)$$

The set of equations 16 through 19, and 21 were solved to estimate the flow and pollutant removals through the filter for all the test runs.

Figure 26a shows the measured total flow and Dupuit estimate of flow through the filter for a selected test run on 5% iron filter. The flow through the filter was estimated only after steady state flow was established. The flow estimates shows that the leakage around the filter was $0.0046 \text{ ft}^3/\text{s}$ (7.8 L/min), which was significant compared to the flow through the filter of $0.0007 \text{ ft}^3/\text{s}$ (1.2 L/min). At the flow rate of $0.0007 \text{ ft}^3/\text{s}$ through the filter, the water-media contact time was 11.3 minutes. For this contact time with media, the model-predicted C_{filter} and $C_{leakage}$ for phosphate and zinc were computed and are shown in Figure 26b and Figure 26c. The corresponding average removals were 59% for phosphate and 59% for zinc.



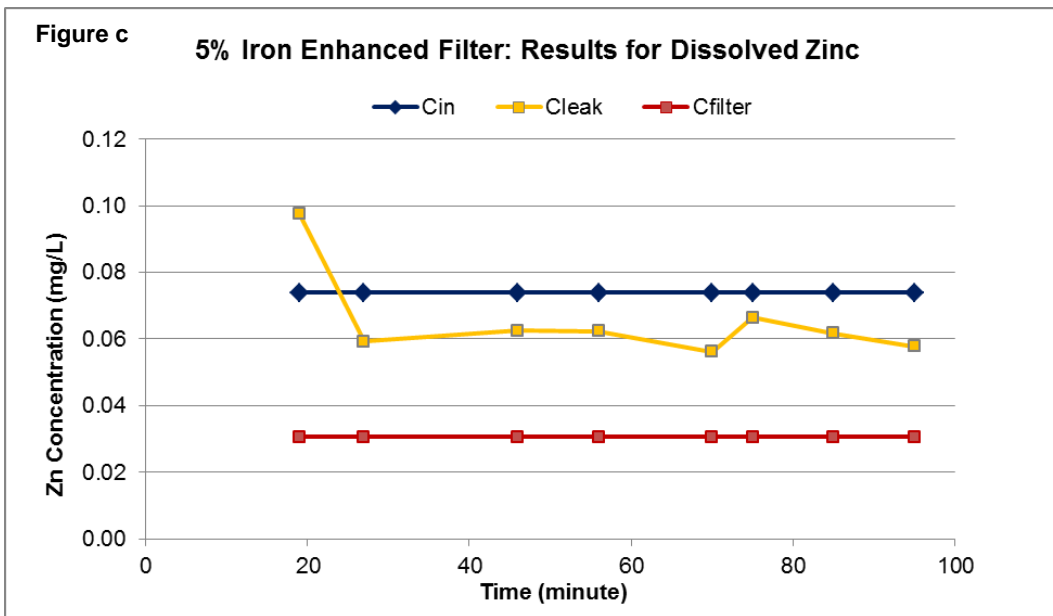
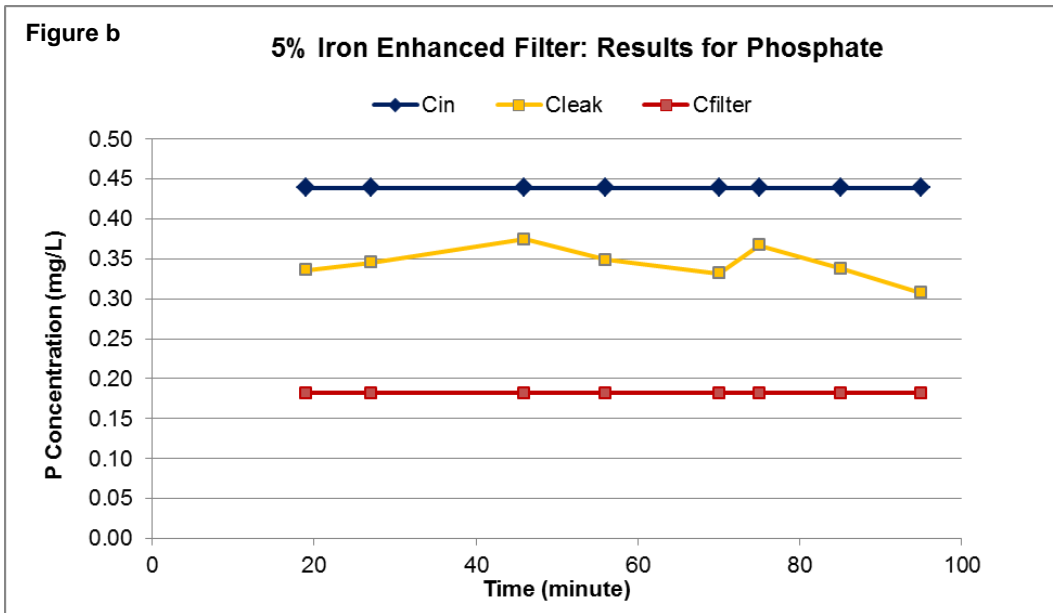


Figure 26. Results for flume test on the 5% iron ditch check filter: (a) Total and estimated flow through the filter. (b) Concentrations of phosphate in influent, filtered water, and leakage. (c) Concentrations of zinc in influent, filtered water, and leakage.

The model-predicted pollutant removals for 7.5% iron enhancement were 60% for phosphate, zinc, and cadmium. As mentioned earlier, export of lead and copper was observed during all the flume tests. Therefore, the actual removal of these two metals by the filter could not be estimated using the data collected.

ii. Performance of Steel Wool Filter

Three types (grades) of steel wool filters were tested in the flume. For each filter, the first test was conducted on fresh non-rusted filter fabric. A second test was conducted after two or three weeks on selected rusted filter fabrics. Concentrations of phosphate were measured in all influent and effluent water samples, however metal analyses were performed for selected samples only. The measured influent and effluent phosphate concentrations and corresponding percent pollutant removals for all steel wool filters are summarized in Table 26. While fresh steel wool showed relatively low phosphate retention of 3 – 20%, the rusted filter fabrics showed increased phosphate retention.

Figure 27 shows the measured influent and effluent phosphate concentrations for a rusted steel wool filter (#0 grade). A steady state flow of $0.0007 \text{ ft}^3/\text{s}$ (8 L/min) was recorded during this test. The steel wool filter retained between 40 and 90% of input phosphate during the test and the average retention was about 65%. The metals concentrations were analyzed for this filter only. The retention of zinc by the filter was 41 – 74% (average = 61%) and that of copper was 50 – 79% (average = 66%). The lead concentrations were below analytical detection limit, hence the percent removal was not calculated.

Table 26. Measured influent and effluent concentrations and removals of orthophosphate through the steel wool filters during the pilot tests. Removal = $\frac{(C_{in}-C_{out})}{C_{in}} \times 100$

Filter name	n	Test 1: Phosphate Concentration (mg/L)		Percent Phosphate Removal	n	Test 2: Phosphate Concentration (mg/L)		Percent Phosphate Removal
		Influent	Effluent			Influent	Effluent	
Grade #0, 2400 kg/m ³	9	0.22	0.02 – 0.16 (mean = 0.08)	30 – 94 (mean = 65)	10	1.0	0.71 – 0.89 (mean = 0.79)	14 – 32 (mean = 21)
Grade #3, 2400 kg/m ³	6	0.20	0.14 – 0.19 (mean = 0.19)	4 – 33 (mean = 18)	7	0.57	0.45 – 0.47 (mean = 0.46)	16 – 23 (mean = 20)
Grade #0-3, 1900 kg/m ³	7	0.21	0.16 – 0.18 (mean = 0.17)	14 – 21 (mean = 17)	8	0.75	0.45 – 0.61 (mean = 0.56)	21 – 41 (mean = 26)

n = number of samples collected during the test

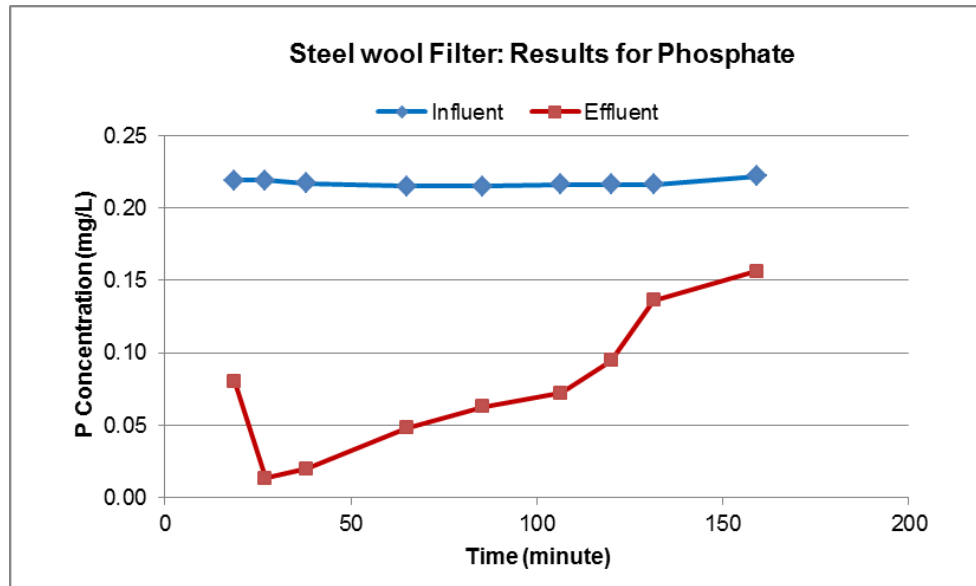


Figure 27. Concentrations of phosphate measured in the influent and effluent during the pilot test on steel wool filter.

As the filter rusted, the steel wool sheets exhibited disintegration into smaller pieces and clumping of the pieces into a hard mass. It is expected that the filter will eventually ‘wear out’ as the iron rusts washes away. As an example, 700 *lb* steel wool media has been estimated to last about 75 – 160 days for water at pH of 5 – 6 (Steven Bouse, GMT Inc., IL; personal communication). Although the rusted steel mass will capture pollutants in the stormwater, the structural life of a filter composed of only steel wool will be relatively short and may often require replacement, which may not be practical in practice. Therefore, the design of the steel wool filter prototype needs to be modified before use in a field-scale installation. One suggestion is to place the steel wool layers within a sand filter. This will ensure that the rusted steel wool will remain within the sand media and continue to adsorb pollutants even after disintegration. This filter will likely last longer. A modified steel wool filter design was not built and tested in the laboratory.

2d. Summary

The pilot experiments were developed to determine the optimal design and sizing of the ditch check filter for effective reduction of dissolved phosphorus and metal concentrations in stormwater. Sand filter media containing 5% and 7.5% iron filings (by weight) as enhancements and measuring 12 *inches* length (in the direction of the flow) was tested. Synthetic stormwater containing median levels of pollutants was passed through the filter, and pollutant concentrations in the influent and effluent water samples were measured to determine the percent pollutant removal through the filter. Although issues with leakage around the sand filter affected test results, the actual treatment capability of the filter was predicted using

models. For the design conditions tested, approximately 60% pollutant removal can be expected through the filter. In the iron-enhanced ditch check sand filter, the mechanisms of phosphorus and metal removal are adsorption of phosphate ion on ferric ion, and adsorption on and/coprecipitation of metal ions with iron hydroxide (Kishimoto et al. 2011).

The steel wool filter exhibited good removal of both phosphate and metals as well. As the steel wool rusted, the average removals were 65% for phosphate, 66% for copper, and 61% for zinc. However, the filter exhibited structural disintegration as steel wool rusted during the tests, suggesting that a filter consisting of only steel wool will not be a viable option. Therefore, a modification in design is required.

3. Development of Model for Pollutant Removal Performance (Task 7)

3a. Objectives

The objective of this task was to develop a model to predict the performance of ditch check filter at retaining dissolved phosphorus and metals (cadmium, copper, lead, and zinc). First, the hydraulics of the filter including flow through the filter and contact time of water with the filter media was modeled. Second, the pollutant retention in the filter media was modeled using an existing model developed by Erickson et al. (2012).

3b. Methods

i. Flow Model

The Dupuit equation (equation 22) computes the flow through the filter (Q ; ft^3/s) as a function of the saturated hydraulic conductivity of the media (K_{sat} ; ft/s), upstream head (h_0 ; ft) and downstream head (h_d ; ft) of water at the filter, and size of the filter *i.e.* length in the direction of flow (L ; ft) and width (B ; ft), given by:

$$Q = \frac{K_{sat}}{2L} (h_0^2 - h_d^2) \times B \quad (22)$$

For the ditch check filter, the profile of water level along the depth of the filter is a function of the head loss ($h_0 - h_d$), as shown in Figure 28. At a given distance x along the filter length, velocity of water (v) can be expressed in terms head of water and specific discharge ($q = \frac{Q}{B}$). Integrating the velocity at each cross-section along the length of the filter, the total contact time with the media (T_{media}) can be obtained, as shown in equations 23 through 26.

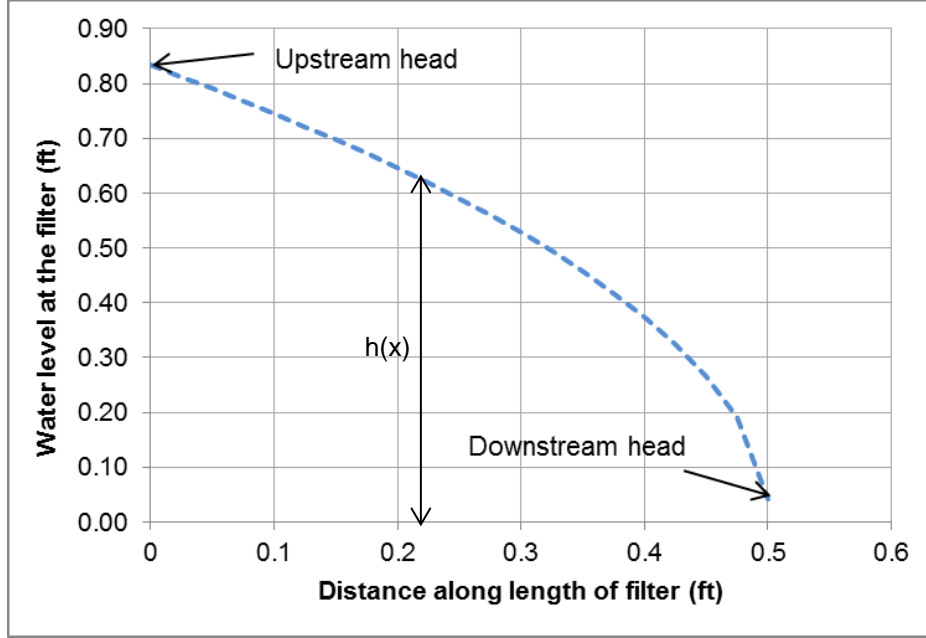


Figure 28. Profile of water head in the ditch check filter.

$$h(x) = f\{h_0, h_d, x\} \quad (23)$$

$$v = \frac{dx}{dt} = \frac{Q}{B \times h(x)} = \frac{q}{h(x)} \quad (24)$$

$$\int_0^L h(x) dx = \int_0^T q dt \quad (25)$$

$$T_{media} = \frac{4L^2}{3K_{sat}} \frac{(h_0^3 - h_d^3)}{(h_0^2 - h_d^2)^2} \quad (26)$$

Since the removal of pollutants is dependent on the contact with iron in the media, the contact time with iron only ($t_{contact}$) was calculated as (Erickson et al. 2012):

$$t_{contact} = T_{media} \times \text{porosity of media} \times \frac{\text{Surface area}_{iron}}{\text{Surface area}_{total}} \quad (27)$$

The value of saturated hydraulic conductivity of media was obtained from the Erickson et al. (2012) study, since similar media (95% C-33 sand and 5% iron filings) was utilized in their column experiments and the pilot studies. The surface area ratio calculation method outlined in Erickson et al. (2012) was followed.

ii. Pollutant Retention Model

A mathematical model was developed by Erickson et al. (2007, 2012) to predict phosphate retention in an iron-enhanced sand filter. This model estimates the retention of phosphate on iron as a function of influent concentration, contact time with iron in the media, and total mass of phosphate retained, and is of the form:

$$\frac{C_{out}}{C_{in}} = 1 - (\beta_0 e^{-\beta_1 \Sigma M})(1 - e^{-\beta_2 t_{contact}}) \quad (28)$$

where, C_{in} = Influent concentration (mg/L)

C_{out} = Effluent concentration (mg/L)

β_0 = Coefficient related to the phosphate retention capacity of iron = 0.984

β_1 = Coefficient related to the rate at which C_{out} approaches C_{in} (g Fe/g P) = 135

ΣM = Instantaneous sum of phosphate mass retained (g P/g Fe)

β_2 = Coefficient related to the phosphate sorption rate constant (1/s) = 54.6

$t_{contact}$ = Contact time between phosphate and iron (s)

The model coefficients indicated are for sand filter enhanced with 5% iron (Erickson et al. 2012).

The model in [equation 28](#) was utilized to predict the retention of phosphate and metals in the ditch check filter. The experimental data collected in the pilot study was used to fit the model to the data. The model predictions were compared to the observed data using the standard error function:

$$\text{Standard error} = \sqrt{\frac{1}{(n-2)} \times \Sigma(\text{data} - \text{model})^2} \quad (29)$$

3c. Results

Data from the pilot study on the ditch check sand filter, including filter dimensions, head loss, measured influent concentration, and estimated treated water concentration, were the inputs to the flow and pollutant retention models. The contact time with entire media was 11.3 *minutes*, from which the contact time with iron was computed as 1.1 *seconds*. The modeling exercise was performed for phosphate, zinc, and cadmium data. As mentioned in the pilot test results, effluent concentrations of lead and copper exceeded the influent concentrations, attributed to leaching of metals from the media. Therefore, pollutant removal modeling was not performed for these two metals.

First, the model of Erickson et al. (2012) ([Equation 28](#) where

$\beta_0 = 0.984$, $\beta_1 = 135$, and $\beta_2 = 54.6$) was used to predict the pollutant retention performance of the ditch check filter and were compared to that of the experimental data. The standard errors obtained were high; as an example, errors for phosphate modeling were 0.439

and 0.459 for 5% and 7.5% iron enhancements, respectively. Therefore, a new best-fit model coefficient, β_2 , was calculated such that the standard error was minimized (equation 29). A non-linear Newton method and model constraints discussed in (Erickson et al. 2012) were employed to solve for the best-fit coefficients ($\beta_0, \beta_1, \beta_2$). The coefficients $\beta_0 = 0.984$, $\beta_1 = 135$, and $\beta_2 = 0.919$ produced the least standard error of 0.044 for 5% iron enhancement and 0.046 for 7.5% iron enhancement for phosphate data. The standard errors (S.E.) were similar for the metals data (S.E. = 0.043 for Zn and S.E. = 0.045 for Cd). Therefore, the proposed model for predicting the performance of the ditch check sand filter in removing pollutants is:

$$\frac{C_{out}}{C_{in}} = 1 - (0.984 e^{-135 \Sigma M})(1 - e^{-0.919 t_{contact}}) \quad (30)$$

VI. Swale Maintenance Schedules, Costs and Efforts (Task 6)

1. Objective

The objective of this task is to document the best available knowledge on swale maintenance schedules, costs and effort. In addition, surveys on maintenance practices were distributed to appropriate sources, and the information gained combined into the best available knowledge. The best current knowledge and practices from published literature, technical manuals and handbooks, and survey result are incorporated to achieve the task objective.

2. Operation and Maintenance of Roadside Swales

Frequency of inspection and maintenance is dependent upon land uses and construction practices in the watershed and rainfall amounts and intensities. Visual inspection and associated maintenance, however, should be performed at least once per year with some recommending a minimum of two inspections per year. For example, the California Stormwater Quality Association (2003) recommends at least two inspections per year; one at the end of the wet season to schedule summer maintenance and one in the fall in preparation for the winter (with other inspections after heavy rainfall events). Also, the New Jersey Stormwater Management Manual draft (NJ DEP 2010) states that two inspections should occur annually with one occurring outside of the growing season.

Landphair et al. (2000) states that maintenance requirements for roadside swales are minimal, other than typical trash pickup and mowing. Landphair et al. (2000) also states that immediate replacement of any dead, dying, or missing vegetation is imperative. As discussed later, others recommend sediment removal, if necessary, as part of regular maintenance.

Some of the results from a survey on inspection frequency, effort, and complexity of stormwater treatment devices that was distributed to municipalities throughout Minnesota and Wisconsin are presented in [Table 27](#). Results show the percent of respondents who indicated the listed frequency or complexity along with the total number of respondents, *n*.

[Table 27](#) indicates that swales are most commonly inspected less than once a year with median annual staff hours spent on inspection and maintenance of 2 hours. It must be noted that this level of inspection is a representation of what is done in practice and may not optimize swale performance. Forty-six percent of respondents indicated that related maintenance issues were moderately complex or complicated. This was the highest combined percentage in these two categories for all stormwater treatment practices (ponds, filters, infiltration devices, etc.) although a similar value was also obtained for constructed wetlands.

Table 27. Inspection frequency and efforts related to swales.

	<i>Response</i>	<i>n</i>
Inspected less than once a year	58%	13
Inspected once a year	33%	13
Inspected twice a year	0%	13
Inspected more than twice a year	8%	13
Median annual staff hours spent on inspection and maintenance	2	11
Minimal complexity*	46%	14
Simple complexity*	8%	14
Moderate complexity*	38%	14
Complicated complexity*	8%	14

*Levels of complexity: minimal = stormwater professional or consultant is seldom needed; simple = stormwater professional or consultant is occasionally needed; moderate = stormwater professional or consultant is needed about half the time; and complicated = stormwater professional or consultant is always needed

Survey results also showed that factors that may reduce swale performance and require maintenance include the presence of sediment, trash, invasive vegetation, and other less common factors such as pipe clogging and bank erosion. The percentage of respondents that indicated these and other issues frequently caused reduced performance are given in [Table 28](#). Sediment buildup, invasive vegetation, and litter or debris were the primary factors indicated most frequently.

Table 28. Inspection frequency and efforts related to swales.

	<i>Response</i>
Sediment build up	21%
Litter/debris	26%
Pipe clogging	5%
Invasive vegetation	26%
Bank erosion	11%
Groundwater level	5%
Structural problems	5%

(*n* = 19)

3. Maintenance Activities

As previously discussed, publications suggest that swales should be inspected at least once per year with some suggesting more frequent inspections (e.g. twice per year and/or after every major rainfall). Components of inspections may include vegetation health and density, hydraulic conductivity of the soil, pools of standing water, trash racks, structures, low flow channels, riprap aprons, and cleanouts (NJ DEP 2010, CASQA 2003, University of Florida 2008). Many publications list mowing and litter/debris and/or sediment removal as the primary suggested maintenance activities or found that these actions were associated with swales that performed well for infiltration and/or contaminant removal (Barrett 2004a, 2004b, Claytor and Schueler 1996, US EPA 1999, Landphair 2000). The Minnesota Stormwater Manual suggests avoiding mowing the swales in the first few years until the vegetation is established. After establishment, it is suggested that vegetation be harvested and removed twice during each growing season, depending on the vegetation. Harvesting should be done only when the soil is dry to prevent damage to the vegetation, soil compaction, and ruts that will concentrate flow (MN Stormwater Steering Committee 2005). Also, sediment removal should not result in the removal of vegetation roots (NJ Stormwater Management Manual). CASQA (2003), however, states that mowing may not significantly impact the pollutant removal performance of a vegetated swale and suggests that mowing only once or twice a year for safety, aesthetics, or weed suppression may be enough. When cut, vegetation should be removed from the waterway. CASQA (2003) includes one of the more detailed and complete lists of maintenance activities and frequencies associated with vegetated swales. The CASQA example of a summary maintenance checklist is given (in modified form) in [Table 29](#).

The Virginia Stormwater Management Handbook (Virginia DCR 1999) suggests a "dense and vigorous" grass cover should be maintained in a grass swale and the NJ Stormwater Management Manual suggests that the vegetative cover be maintained over at least 95% of the swale. Both of these documents also suggest maintaining the grass at a height from 3 to 6 *inches*. Claytor and Schueler (1996), however, suggest that grass levels should be maintained at 3 to 4 inches for optimal filtering capability. The Idaho Department of Environmental Quality (2005) states that in order to prevent nutrient transport, grasses or emergent wetland-type plants should be cut to a low height at the end of the growing season. For other pollutant control objectives, the plants should be left at a height exceeding the design water depth by at least 2 inches.

Table 29. Inspection and maintenance activities and suggested frequencies.

Inspection Activities	Frequency
<ul style="list-style-type: none"> Inspect after seeding and after first major storms for damages 	Post construction
<ul style="list-style-type: none"> Inspect for signs of erosion, damage to vegetation, channelization of flow, debris and litter, and areas of sediment accumulation. Perform inspections at the beginning and end of the wet season. Additional inspections after periods of heavy rain are desirable. 	Semi-annual
<ul style="list-style-type: none"> Inspect level spreader (if present) for clogging, grass along side slopes for erosion and formation of rills or gullies, and sand/soil bed for erosion problems. 	Annual
Maintenance Activities	Frequency
<ul style="list-style-type: none"> Mow grass to maintain a height of 3-4 inches, for safety, aesthetic, or other purposes. Litter should always be removed prior to mowing. Clippings should be composted. Irrigate swale during dry season or when necessary to maintain Provide weed control, if necessary, to control invasive vegetation. 	As needed (frequent, seasonally)
<ul style="list-style-type: none"> Remove litter, branches, rocks, blockages, and other debris and dispose of properly Maintain inlet flow spreader (if applicable) Repair any damaged areas within a channel identified during inspections. Erosion rills or gullies should be corrected as needed. Bare areas should be replanted as necessary. 	Semi-annual
<ul style="list-style-type: none"> Correct erosion problems in the sand/soil bed of dry swales Plant an alternative grass species if the original grass cover has not been established. Reseed and apply mulch to damaged areas 	Annual (as needed)
<ul style="list-style-type: none"> Remove all accumulated sediment that may obstruct flow through the swale. Sediment accumulating near culverts and in channels should be removed when it builds up to 3 inches at any spot, covers vegetation, or it has accumulated to 10% of the original design volume. Replace the grass areas damaged in the process. Rototill or cultivate the surface of the sand/soil bed of dry swaled if standing water is present for more than 48 hours. 	As needed (infrequent)

From CASQA 2003

Mazer et al. (2001) suggest the following practices (listed in order of importance) to establish a good vegetative cover in a swale:

1. Avoid shading by adjacent vegetation, structures, or side slopes,
2. Minimize inundation during the dry season,
3. Install > 0.7 *ft* of moderately well-drained soil (*i.e.* sandy loam) and lightly compact it,
4. Hydroseed and irrigate during dry periods, if necessary,
5. Minimize fluctuating wet/dry periods,
6. Set bottom slope from 0.5 - 2.0% and maintain constant slope the entire swale length.

Also, the plant species should be adequate for the site conditions and appropriate soil stabilization methods (*e.g.* mulch, mats, blankets, etc.) should be used prior to the establishment of vegetation (MN Stormwater Steering Committee 2005).

Areas showing evidence of erosion should be immediately stabilized and reseeded. The area should be inspected once every two weeks until vegetation is established or for at least the first growing season (NJ Department of Environmental Protection, 2010). Also, grassed slopes should be inspected for rills and gullies and corrected, as needed (Claytor and Schueler 1996). According to Claytor and Schueler (1996) and Lake Superior Streams (2013), sediment should be removed when it has accumulated to approximately 25% of the original design volume or channel capacity. The Idaho Department of Environmental Quality (2005) states that for a bioswale, sediments should be removed when they have built up to 6 *inches* at any location, cover vegetation, or otherwise interfere with the operation of the swale. The California Stormwater Quality Association, however, states that sediment should be removed when it builds up to 3 *inches* (CASQA 2003).

Barrett (2004a, 2004b) found that the side slopes of a road-side swale can act much like a filter strip in that much of the runoff is filtered and/or infiltrates on the side slope. If infiltration rates into or sediment retention rates of a filter strip are unacceptable, the top-soil may have to be broken up and the surface reconstructed.

Tilling of the soil has been suggested (Lake Superior Streams 2013, Claytor and Schueler 1996, Virginia DCR 1999, CASQA 2003) to improve infiltration rates into the soil. This may be necessary after sediment has been removed if the swale does not infiltrate water at acceptable rates. Alternative actions include drilling or punching through the soil surface (Virginia DCR 1999).

Barrett et al. (1998) noted that a sediment lip may form at the edge of pavement and that this lip may prevent water from entering the swale. It was suggested that the edge of pavement should be higher than the adjacent soil in order to prevent formation of the sediment lip.

4. The Cost of Maintenance

The US EPA (1999) estimates that annual O&M costs for swales are about 5-6% of the total construction cost. The US EPA also notes that site and region specific parameters can vary greatly so this value is only a general guideline. Caltrans (2004) estimates that a bioinfiltration swale with a 4.9 acre watershed would incur 51 hours of labor per year (47 for maintenance, 3 for administration, and 1 for inspections) with equipment and materials costs (1999 dollars) of \$492. With a labor cost of \$44/hr, the total annual estimated O&M cost is \$2,736. Adjusting this value for regional price differences (US EPA 1999) and for 3% inflation per year results in an annual estimated O&M cost of \$3725 (2013 Minnesota dollars).

Hsieh and Yang (2007) state that the annual O&M costs associated with a grassed swale are 10% of the construction cost. The listed reference (Sample 2003), however, does not have any mention of O&M costs and, in fact, states "data on O&M costs are seldom available on a comprehensive basis and were not a part of this study." An email to Dr. Hsieh inquiring of the source of the 10% value has not yet received a response.

Landphair et al. (2000) estimated construction and O&M costs for a water quality swale serving watersheds of 1, 2, 3, 4, and 5 *acres*. Reported estimated annual O&M costs were 24-28% of the construction cost but the estimated cost included yearly check dam repair that ranged from 70-74% of the annual O&M costs. A properly constructed check dam, however, should require very little maintenance (Virginia DCR 1999). With the check dam repair costs subtracted from the costs estimates, annual O&M costs as estimated by Landphair et al. (2000) range from 8.7% (for the 1 acre watershed) to 6.4% of the original construction cost for the 4 and 5 *acre* watershed.

The SWRPC (1991) estimates annual O&M costs to be \$0.58/linear *ft* for a 1.5 deep, 10 *ft* wide swale to \$0.75/linear *ft* for a 3 ft. deep, 20 *ft* wide swale. Adding 3% for yearly inflation results in costs of \$1.11 and \$1.44 per linear *foot* for the 10 *ft* wide and 21 *ft* wide swale (in 2013 Minnesota dollars), respectively. No regional cost adjustment is necessary because the SWRPC is in the same region as Minnesota. The allocation of the estimated costs are shown in [Table 30](#). It should be noted that no cost for sediment removal is included in this estimate.

Table 30. Estimated O&M cost allocation for grassed swales.

	1.5 ft deep, 10 ft wide swale	3 ft deep, 20 ft wide swale
Program		
Administration & Inspection	26%	20%
Mowing	24%	28%
General Lawn Care	31%	37%
Debris & Litter Removal	17%	14%
Grass Reseeding	2%	1%

From SWRPC (1991)

Data obtained from the Minnesota Department of Transportation (MnDOT) indicated that in 2013 swale lengths of 60 *ft* and 130 *ft* received maintenance that included mowing and sediment removal at a cost of \$6 and \$13 per *ft*, respectively. The corresponding person-hours were 0.22 per *ft* for the 60 *ft* long swale and 0.28 per *ft* for the 130 *ft* long swale. Also, three concrete aprons (end sections attached to concrete pipes) were maintained (e.g. sediment removal) at a total cost of \$856 (\$285/apron) and a total of 17 person-hours. Finally, MnDOT estimates that mechanically breaking up the soil to a depth of 16-20 *inches* to increase infiltration into the swale would cost about \$420/*acre*.

5. Summary and Recommendations

There is limited information available on maintenance actions and corresponding frequencies in roadside swales that will optimize cost-effectiveness. Some experimental work has been documented but, in most cases, there has not been wide-spread, follow-up field verification of such work. Also, some field experiences have been documented but again, in most cases, there has not been follow-up work to verify the field experiences or determine the impact of all relevant variables. Even so, the work and experiences reported to date allow maintenance actions to be optimized to some extent. Thus, the following recommendations are intended to optimize the cost-effectiveness of road side swales in Minnesota and are made with the realization that they should be revised based on future experiences, other case studies, and research.

Recommendations:

1. Roadside swales should be inspected at least once per year.
2. Vegetation should not be mowed until it has become established. Mowing, when it does occur, should not be done under wet conditions when the soil may be soft. Also, mowed vegetation should be removed from the waterway. Mowing should be

performed at least once per year, depending on vegetation type and goals and objectives for stormwater quality improvement.

3. Inspection should include checks for:
 - a. Presence of trash, litter, and debris
 - b. Vegetation health and density and the presence of invasive vegetation
 - c. Standing water
 - d. Excessive sediment accumulation
 - e. Side slope instability or erosion
 - f. Structural integrity of culverts, aprons, and other structures
4. Based on inspection results, the following actions should be taken:
 - a. Trash, litter, and debris should be removed, if warranted.
 - b. Vegetation. Unhealthy, dead, or dying vegetation should be removed. If bare spots are present, vegetation should be reestablished. In both cases, the cause of the lack of healthy vegetation should be determined and remediated. For example, excessive water velocities can erode soil and/or damage vegetation and long periods of standing water can kill plants that are not tolerant to these conditions. It makes little sense to replace vegetation if the causal conditions exist and will exist in the future. Thus, the problem must be determined and removed prior to addressing the vegetation itself.
 - c. If standing water exists in a swale more than 2 days after a runoff event it can be a sign of a downstream obstruction and/or poorly draining soils. Downstream obstructions that are not part of the swale design (e.g. check dams) should be removed. If poorly draining soils are suspected, the soil should be tested (when dry) with an infiltrometer and/or permeameter to determine the saturated hydraulic conductivity of the soil. If hydraulic conductivity values are below 0.5 *in/hr* for hydrologic soil group A and B soils or are below 0.1 *in/hr* for C soils, consider removing sediment and/or mechanically breaking up the top 1 to 1.5 *ft* of soil to increase infiltration rates. It may not be possible to increase infiltration into D soils without the use of underdrains.
 - d. Sediment accumulation. Sediment over 3 *inches* deep should be removed. Sediment removal should not remove vegetation. If a sediment layer less than 3 *inches* is present and the area has problems with standing water, the area should be testing with an infiltrometer to determine the saturated hydraulic conductivity of the soil. If infiltration rates are lower than those previously expressed, the sediment should be removed. If sediment removal does not sufficiently increase infiltration, mechanically breaking up the soil should be considered.
 - e. Side slope instability or erosion. If it appears the side slope is unstable it should be stabilized immediately. If erosion is present, the source of erosion should be identified and eliminated and any affected vegetation replaced.

- f. Structural integrity. If any structures or their foundations appear to be out of alignment, damaged, or otherwise at risk, action should be taken to reestablish the structure to design conditions.

VII. Conclusions

Soils from 15 grassed roadside swales that were representative of Minnesota soils were sampled and soil textural analysis was performed on the soil samples. From these, five were selected for analysis of infiltration rates, chosen to represent the range of soil samples found in Minnesota swales. The near-surface (upper 8 inches) K_{sat} values of the five swales varied from 0.3 *in/hr* to 1.56 *in/hr*, and were roughly a factor of 2.8 (mean) or 1.5 (median) greater than the published mean values (Rawls, et al. 1983) for the types of soil. This may be due to roots creating macropores in the near-surface soil, which is normally not taken into consideration in laboratory soil permeability tests.

We can conclude that there is statistically significant evidence that soil type has an effect on the mean saturated hydraulic conductivity of a swale while soil moisture content and season do not have an effect. This is expected, because K_{sat} values should not be dependent upon soil moisture and should not change significantly with season, within the spatial variation of K_{sat} . This observation is a verification of the validity of the MPD infiltrometer in measuring K_{sat} .

To capture the spatial variability and estimate a geometric mean K_{sat} value with a sufficiently low uncertainty, 20 measurements of K_{sat} are recommended for each swale. Distance from the downstream may or may not have an effect on the geometric mean K_{sat} of the swale. Similar to what was found in the literature (Yonge 2000, Barrett 2004a, 2004b, Ahearn and Tveten 2008), the K_{sat} values at the side slope are observed to be higher than the center of the swale. This could be attributed to sedimentation in the center of the swale causing a reduction in K_{sat} values.

The swale located in Madison did not show any evidence of the effect of the distance from downstream on geometric mean K_{sat} of the swale. On the other hand, the swales located in Minnesota showed evidence that the distances from downstream can have a positive or negative effect on the cross-sectional geometric mean K_{sat} . This result implies that infiltration measurements should be spread over the swales of interest to obtain accurate results. There was no evidence that deposition at the lower reaches of the swale caused a lower K_{sat} in that region, possibly because the grass roots broke up the sedimentation and created macropores in the near-surface soil or because sedimentation did not favor the lower reaches of the swale.

For the Madison swale, the stormwater runoff-routing model, that takes into account the spatial variability of K_{sat} in the swale, delivered a good prediction of measured runoff/rainfall ratio for some rainfall events, while for some low intensity storm events (below 0.22 *in/hr*) the predictions were not as accurate, but still predicted runoff at the lower end of the swale. For the five representative Minnesota swales, the runoff model predicts that the side slopes of the swales infiltrate the entire runoff volume for a 1.1 inch 24-hour event. The ratio between the width of the road and the width of the side slope is 2.3:1 (Hwy 212) and higher for Minnesota

swales. Thus, all five of the roadside swales studied in detail are predicted to completely infiltrate a 1.1 inch, 24 hour runoff event with type II characteristics.

Field studies in the literature on swales show that infiltration of runoff volumes ranging between 9 to 100% can be achieved by swales. Factors such as soil permeability, initial moisture content, compaction of soil, presence of vegetation, *i.e.* plant or tree roots. Roadside swales do not have a high ratio of impervious area to swale area (5 to 10), so the infiltration rate does not need to be as large as an infiltration basin, with impervious to basin area ratios of up to 50. The evidence to date from Minnesota is that roadside swales are an effective means of infiltrating stormwater runoff.

The impact of increasing saturated hydraulic conductivity on runoff depths estimated by the TR 55 method was investigated for three scenarios that could be used to model runoff in or near a roadside swale. It was found that the impact of increasing K_{sat} was much more significant for open grass in good condition, which is 100% covered with vegetation, than the paved road with an open ditch and gravel road scenarios which were covered by only some or no vegetation. The gravel road was slightly more impacted by an increase in K_{sat} than the paved road with an open ditch. This could be due to the fact that the gravel road can infiltrate rainwater over its entire surface whereas the paved road cannot.

Application of the TR 55 method indicate that for a storm of 1 *inch* (a typical water quality storm design depth) runoff depths are, for all scenarios and soil types, less than 0.45 *inches*. Thus, the TR 55 method indicates that for smaller storms, much of the rainfall will be abstracted, before runoff, regardless of the hydrologic soil group. One must keep in mind, however, the large errors often encountered with the TR 55 method and that the stated limitation of that the TR 55 method is for runoff depths of greater than 0.5 *inch*.

Finally, for the paved road with an open ditch scenario, runoff depths predicted by the TR 55 method for a 1 *inch* storm range from 0.13 to 0.45 *inches*. This is contrary to field experimental results found by Barrett (2004a, 2004b) and the results found herein that indicated the side slopes of a swale can often infiltrate 100% of the road runoff for 1 *inch* storms. In addition, most of the K_{sat} values found in this report were associated with higher infiltrating hydrologic soil groups than the TR 55 specification for the soils. In addition, the runoff depth is less than 0.5 *inch*, so the accuracy of the TR 55 method is questionable, but it does demonstrate the need for more detailed and accurate modeling of runoff infiltration by swales. An analysis of CN for smaller storms is being undertaken in a current LRRB project, "Determination of Effective Impervious Area in Urban Watersheds."

The second major focus of this research study was on developing a ditch check filter that can be installed in swales to increase the capture of dissolved phosphorus and metals (copper, lead, zinc, cadmium) from stormwater runoff. Ditch check filters with enhanced filter media can be installed in the swales to increase the potential of swales to remove dissolved phosphorus and

metals from stormwater runoff. As previously discussed, cost-effective enhancing agents to remove dissolved phosphorus and metals must satisfy requirements that include rapid kinetics, high capacity, and low cost among others. Due to the relatively short contact time between stormwater runoff and the media of a stormwater filter, removal mechanism kinetics must be relatively rapid if significant contaminant removal is to occur. Also, a cost-effective enhancing agent must have a large contaminant capacity, otherwise maintenance costs associated with replacing the media or restoring its capacity will be cost-prohibitive. Finally, the material cost of an enhancing agent must be low enough such that the initial cost and, if replacement of the media will be required over the life of the practice, the replacement cost of the material does not render the use of the material cost-prohibitive.

Of the materials reviewed for phosphorus removal, ferrous-based materials, alumina, and water treatment residual are the potential materials for effective phosphate removal. Ferrous based materials, alumina, zeolites (if sodium leaching and increased pH can be minimized), and some activated carbons appear to be the most promising with regards to metal removal. Selected materials that show the most potential as enhancing agents for dissolved phosphorus and metal removal will be tested to further characterize their performance (i.e. capacity, effect on water quality parameters, etc.). Results will be used to develop, optimize and further design filters for field application.

First, batch studies were conducted to select the enhancement media for increased phosphorus and metal removal through the filter. Several potential enhancements were tested using synthetic stormwater and their effectiveness (percent removal) of pollutants determined after one-hour contact time. Of the 17 different materials tested, two iron-based materials (iron filings and steel wool) and alumina showed the highest potential as media enhancement. While alumina provided 60% phosphorus and 90% metal removals, iron filings and steel wool showed the highest phosphorus removal of 90% and metal removal of 90%. These two materials did not increase the final pH above 9, which is the MPCA maximum for aquatic life and recreation. Therefore, iron filings and steel wool were selected as the media enhancements for the pilot study on ditch check filter.

In the pilot study, two prototypes of a ditch check filter were designed, installed, and tested in a flume. The pilot experiments used estimated flow rates and optimal design conditions to determine the needed length (in the direction of the flow) of the ditch check filter and the amount of iron (percent by weight) as media enhancement for maximizing the filters' ability to remove the target pollutants. In the first prototype, a sand filter of dimension 12 *inches* length (in the direction of the flow) was tested. Two filter media compositions were tested in this prototype: one containing 95% C-33 sand 5% iron filings (by weight), and the other containing 92.5% C-33 sand and 7.5% iron filings (by weight). Although the experiments had issues with leakage around the filter and affected test results, and a model of pollutant removal was needed to analyze the experimental results, the model-predicted pollutant removal capacity was found to be approximately 60% for the design conditions tested. The second prototype, steel wool

filter, also showed good pollutant removal of 66% metals and 65% phosphate. However, the steel wool filter exhibited signs of disintegration after weeks of rusting. Hence, a modification in design was proposed to use steel wool embedded in a sand filter.

A model to predict the performance of the ditch check filter was developed. This comprised of modeling the flow through the filter and the pollutant retention in the filter. The pollutant retention is predicted based on the contact time with iron, instantaneous sum of pollutant mass adsorbed and the influent pollutant concentration..

Lastly, the maintenance cost and effort requirements for swales were investigated. Information based on existing experimental work and field experiences, and new survey data on the maintenance of swales and the cost involved was evaluated and combined. Recommendations on maintenance actions and corresponding frequencies in roadside swales that will optimize cost-effectiveness of roadside swales in Minnesota were provided. These recommendations should be revised based on future experiences, other case studies, and research.

In conclusion, swale infiltration capacity measurements and models, and new design standards and models for ditch check filters for swales were developed in this study. The results from this study can be used to design systems for use along roadways throughout Minnesota and the United States. Results can also be used to estimate load reductions due to infiltration and the installation of ditch check filters in Total Maximum Daily Load (TMDL) studies for future installations of roadside swales.

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Appendix A

Extended Literature Review on Performance of Swales for Pollution Prevention

Appendix A: Extended Literature Review on Performance of Swales for Pollution Prevention

This project focuses on stormwater runoff volume reduction by swales and dissolved phosphorus removal from stormwater runoff. Thus, the literature review contained in the main body of the report focused on these issues. Swales, however, can remove other contaminants such as total suspended solids (TSS), nitrogen, and metals and some have questioned (and investigated) the impact of various variables on the performance of swales, such as cold weather. This section (Appendix A) provides a review of available literature that has focused on such aspects of swale performance that are not contained in the main body of the report.

1. Additional Water Quality Treatment Performance of Swales

In addition to phosphorus, swales have the ability to remove other contaminants from stormwater runoff. Studies have investigated the performance of swales with respect to solids, nitrogen, and metal removal. This section reviews literature that has investigated the performance of swales with respect to retention or removal of these contaminants and variables that affect swale performance. It also discusses materials that have potential to increase dissolved metal removal.

1a. Suspended Solids Removal in Swales

Suspended solids can be removed from stormwater runoff flowing in a grassed swale by two mechanisms, 1) filtration by vegetation and, 2) sedimentation and capture on the bottom of the swale (Backstrom 2002b, Schueler 1987, Yu et al. 2001). Documented removal rates vary between close to 100% to large negative values (i.e. export of sediment from the swale). These large discrepancies may be due to many factors including varying total suspended solids (TSS) influent concentrations, differences in solid particle size and density, swale design and maintenance, measurement and/or sampling locations and techniques, and other reasons. For example, Barrett et al. (1998a) states that some studies that have reported swale TSS removal rates used test sections that had “influent” sampling locations located after much sediment had already been removed by the swale. As mentioned previously, the side slope leading into a swale can act as a filter strip and remove significant amounts of contaminants. Other studies (that are discussed in more detail below) support this notion.

According to Backstrom (2002b), the transport and deposition of particles in swales is dependent primarily on particle size, particle density, soil hydraulic conductivity, and grass density and height. The portion of the plants that are above ground are also thought to induce sedimentation of solids (Mazer et al. 2001). Deletic (2001) reported that, based on several studies by other investigators, the sediment removal effectiveness of grassed swales depends on grass density and thickness of the blades, slope of the surface, soil infiltration and

roughness, particle size and density, rainfall intensity and duration, and antecedent sediment and weather conditions.

Backstrom (2002a) investigated a total of nine swales; two artificial turf swales in the laboratory and seven natural grassed swales in the field. Each test used half-hour long, simulated runoff events with synthetic stormwater. The synthetic stormwater was prepared by street sweeping and mixing the collected sediment with water.

Swales with thin turf and short grass had the lowest TSS removal (~80%) whereas swales with fully developed turf achieved TSS removal rates of over 90% for all studies. Thus it was concluded that grass height and spacing are important variables that affect removal efficiency. Also, no relationship was found between swale length and removal efficiency. The authors attributed this lack of correlation to heavy particles dropping out of the water on the upstream portions of the swale.

The laboratory swales were found to capture particles of all sizes equally, perhaps due to large water velocities above the grass. In addition, the laboratory swale with longer grass captured more sediment (>90%) than swales with short grass or thin vegetation (80%). Field swales with dense turf and larger infiltration rates removed more sediment and had no signs of sediment resuspension. Based on previous studies, however, Backstrom did report that sediment resuspension and erosion can occur. Field swales also did show a correlation between removal and particle size with large particles experiencing more removal. This lead Backstrom to conclude that, with respect to TSS removal in the field swales, sedimentation was more important than grass filtration.

Backstrom (2002a) developed a model for sediment removal which exhibited a correlation between particle size and trapping efficiency. Swale residence time could not explain all the variation in the data. Backstrom suggested that infiltration rate and swale bottom slope could help explain this variation. It was also suggested that hydraulic residence time could be used as a design parameter for some swales but this would not necessarily be valid for swales with relatively large infiltration rates. Finally, Backstrom proposed the possibility that large particles are captured mostly by sedimentation and relatively small particles are captured by adsorption onto blades of grass.

Deletic (1999) also studied sediment trapping but used artificial turf and synthetic stormwater. The synthetic stormwater was prepared using sediment from an estuary that was mixed with clean water. Two different grass densities, four flow rates, five flow depths, and a range of sediment concentrations were tested. TSS concentrations were found to decrease exponentially with distance along the flow path until a constant value was reached. The rate of the decrease and the constant value were found to be a function of particle size and density along with the flow depth and velocity. The model results did not agree with the Kentucky Model (Tollner et al. 1976) which models the transport of sediment along a grassed surface in four distinct zones;

in zone 1 all sediment is transported, in zone 2 some sediment is trapped, in zone 3 all sediment is transported as bed load, and in zone 4 all sediment is trapped. Because results did not fit the Kentucky model, a new, simplified model based on a dimensionless fall number, flow depth, and other variables was developed.

In further research, Deletic (2001) modeled runoff generation and sediment transport over grass using a modified form of the Green-Ampt equation for infiltration and a kinematic model for overland flow. The model, which was calibrated and verified against field data, is physically based (i.e. not empirical) and accounts for surface depressions but it does not account for particle infiltration, resuspension, or capture of sediment in ponds. A sensitivity analysis on the model showed that sediment capture (and volume reduction) depends mostly on the following variables, which are listed in order of importance: particle size, grass strip length, particle density, soil saturated hydraulic conductivity, and grass density. Later, Deletic (2005) developed another equation (the Aberdeen Equation) for capture efficiency on steady, uniform, overland flow without infiltration. The equation is only applicable for runoff with small to moderate TSS concentrations.

While the previous discussion focused on the theory of sedimentation and filtration processes that occur in swales, other studies have determined the removal efficiency of swales (or the side slope embankment leading to the swales) and the effect of various variables through monitoring of natural and synthetic rainfall and/or runoff events. Backstrom (2003) reviewed and provided an overall summary for three such studies. In the summary, Backstrom stated that 79-98% of TSS removal occurred during simulated runoff events. In these studies the parameter that best explained TSS removal was the influent TSS concentration. No significant removal occurred below a TSS concentration of 40 mg/L (which agreed with other studies) and below certain minimum concentrations (not stated) swales can act as a source rather than a sink of sediment. For swales with dense, thick vegetative growth no relationship was found between TSS removal and bottom slope, grass height, side slope, or infiltration capacity but more TSS were removed during the growing season. Also, during the growing season less retention of nutrients was observed. This trend was attributed to decaying plant matter that acted as a source of nutrients.

Barrett et al. (1998a) compared the performance of two roadside medians with different vegetation, bottom slopes, and other variables and found that the medians had similar TSS removal rates of 87% and 85%. As part of this investigation it was noted that, at some locations, there were sediment deposits at the road/median interface that prevented runoff from entering the median. As a result, runoff could not flow freely onto the median and was sometimes diverted by the sediment to a curb and gutter system. It was suggested that this phenomenon could be prevented by ensuring that the soil next to the road surface have a slightly lower elevation than the road and by performing periodic maintenance to remove accumulated sediment. Barrett (2008) reviewed data on 14 swales in the International BMP database and found a median TSS removal of 60% with a range from 6% to 70%. The

bioswales (not roadside swales) studied by Mazer et al. (2001) were documented to have TSS removal efficiencies from 60-99%.

In the study by Ahearn and Tveten (2008), which measured contaminant removal along the grassed surface next to the roadway, it was found that TSS removal varied from 59% to 82% at 2 meters from the edge of pavement and from 93% to 96% at 4 meters from the edge of pavement. As Ahearn and Tveten (2008) point out, these removal rates are similar to those found in other studies performed on roadside embankments in Texas and California.

Yonge (2000) studied filter strips adjacent to roadways for their ability to remove typical contaminants found in stormwater runoff. These filter strips were essentially the shoulder of the road that the runoff had to flow over to drain into a roadside swale. Three different test plots, each with a different soil type, were monitored. TSS removal achieved by the filter strips ranged from 20% to 80% with an average value of 72%. Thus, as suggested by other studies, a significant amount of TSS removal from road runoff can occur before the runoff actually enters a roadside swale.

Barrett (2004) also investigated the pollutant removal of two roadside swales in Austin, Texas. As supported by the previously mentioned studies by Ahearn and Tveten (2008) and Yonge (2000), it was determined that most of the pollutant removal occurred along the side slope before the runoff was carried downstream by the swale. Thus, Barrett (2004) suggests that roadside swales be designed with triangular cross-sections with longer side slopes rather than the more common trapezoidal cross-section.

Caltrans (2003) performed a 2-year study in which the performance of vegetative slopes adjacent to roadways was investigated for contaminant removal and infiltration capacity. It was found that these slopes, which were not designed to treat road runoff, performed just as well as buffer strips that were designed and maintained specifically to improve runoff quality. It was found that below 80% vegetative cover on the slope, contaminant removal (of TSS and other contaminants) dropped off significantly, although removal still occurred at vegetative covers of 65%. The vegetative slopes consistently resulted in a significant reduction in the concentration of TSS. In another Caltrans study (2004) swales were found to remove 76% of suspended solids.

Kaighn and Yu (1996) monitored two roadside swales south of Charlottesville, VA. The two swales had differences in slopes (2% vs. 5%) and vegetation characteristics but the biggest difference was that one of the swales had a check dam that slowed the flow of water. The swale with the check dam removed 49% of TSS while the site without the check dam removed 30% of TSS. The difference in removal was attributed to the check dam, not the other differences between the swales.

[Table A- 1](#) summarizes TSS removal performance results for the previously discussed studies.

Table A- 1. Summary of total suspended solids removal performance.

Study	TSS Removal (%)	Comments
Kaighn and Yu (1996)	49	Roadside swale with checkdam
	30	Roadside swale without checkdam
Barrett et al. (1998a)	85, 87	2 roadside medians with different characteristics were investigated
Yonge (2002)	20 - 80	Filter strips adjacent to roads (average = 72%)
Mazer et al. (2001)	60 - 99	Investigation of bioswales (not roadside swales)
Caltrans (2004)	76	Roadside swale
Ahearn and Tveten (2008)	59 - 82	At 2 m from EOP
	93 - 96	At 4 m from EOP
Barrett (2008)	6 - 70	Review of International BMP database (median = 60%)

Note: EOP = Edge of Pavement

Kearfott et al. (2005) investigated the performance of vegetated side slopes of rural highways in Texas for their pollutant removal performance. It was determined that the higher the vegetation density the more pollutant removal occurred regardless of the bottom slope and that most TSS removal occurred within the first 2 to 4 meters from the edge of the roadway. A vegetative cover of 90% resulted in the most pollutant removal but significant removal was observed in sites with only 80% vegetative cover.

Liebens (2001) analyzed the sediments found in swales and found that they contain higher fractions of silts and clays and less sand than control sites. One possible explanation is that relatively large particles are captured by grass (and possibly the side slope of the swale) before the runoff reaches the swale flow path. This study also determined that clay content may be associated with larger metal concentrations due to the large surface areas of clays and their ability to capture metal ions through cation exchange processes. In some agricultural areas, however, swales with higher clay content did not have higher metal concentrations. This apparent contradiction may be due to lower metal production rates in agricultural areas or older, more weathered soils in agricultural areas that have lower cation exchange capacities.

i. Resuspension of Settled Solids

To investigate the potential for resuspension of solids and other factors, Barrett et al. (1998a and 1998b) monitored two highway medians that essentially acted as grassed swales. Data from several rainfall/runoff events were collected and the TSS concentrations of the runoff from the median were found to be less than the concentrations of runoff directly leaving the road. This indicated that, for each storm monitored, there was net TSS removal. The authors did note, however, that one median showed signs of erosion and, if improperly designed such that erosion occurs, medians (or swales) could be a source of sediment.

Deletic (2005) studied resuspension of sediment in a laboratory using flumes with artificial turf to simulate natural grass. Experiments were performed at different bottom slopes, grass densities, flow rates, sediment concentrations, and sediment densities. The resuspension of previously deposited sediment by clean water flowing on the turf was found to be negligible. This finding did not deviate from the conclusion of Cutierrez and Hernandez (1996) that grass itself tends to constrain detachment of settled solids. Mazer et al. (2001) also mentioned that plant roots tend to stabilize sediment deposits. Deletic did find, however, an increase in TSS concentration in the clean water but that was attributed to the washing of sediment off the blades of grass and not resuspension of solids.

1b. Removal of Nutrients and Metals in Swales

By observing data in the International BMP Database, Barrett (2008) investigated the performance of 14 swales. For these swales no significant removal of nitrogen or phosphorus was observed but Barrett suggests that the removal of grass clippings after mowing could remove nutrients from the swale and prevent nutrient release upon decomposition. Significant zinc removal was observed and it was found that the zinc removal efficiency correlated well with the influent zinc concentration. Metal removal reported by Barrett (2008) is shown in [Table A- 2](#). Barrett (2008) also found that reductions in chemical oxygen demand (COD) were only found when influent COD concentrations were greater than 80 mg/L. Barrett et al. (1998b) discussed possible fates of metals once they have been removed from stormwater by vegetation including vegetation in roadside swales. The potential fates are:

- a. Residence in an insoluble form, i.e., attached to particulate matter in the soil matrix;
- b. Uptake of soluble metals by plants;
- c. Uptake by animals who consume plants with accumulated metals;
- d. Leaching of soluble metals from the soil into groundwater;
- e. Removal from the filter strip to receiving waters by runoff from subsequent storm events; and
- f. Removal from the filter strip by wind action on particulates containing metals.

Table A- 2. Metal removal reported by Barrett (2008).

Constituent	Removal (%)	
	Range	Median
Total Zinc	8 to 87	60
Dissolved Zinc	22 to 58	40
Total Copper	-3 to 75	62
Dissolved Copper	-17 to 54	24

As part of the same report (Barrett et al. 1998b), the metal deposition rate for zinc and lead on the two highway medians were found to be less than one-tenth of the allowable rate for crop lands. Based on the lead deposition rates, Barrett et al. (1998b) estimated it would take 244 years at one site and 1204 years at the other site for the lead soil concentrations to exceed regulatory limits. It was also estimated that zinc soil concentrations would exceed regulatory limits after 570 years at the first site and 304 years at the other site.

Barrett et al. (1998b) performed experiments at different water depths on vegetated laboratory swales using synthetic stormwater created to represent typical stormwater runoff pollutant concentrations in the Austin, Texas area. Underdrains were used to collect stormwater that infiltrated into and through the swale. Upon reaching the underdrain the water had infiltrated through the top layer of Buffalo grass, 16 cm of topsoil, and 6 cm of gravel. The highest removal efficiencies in the laboratory swales were for suspended solids, zinc, and iron. The ranges of all removal efficiencies (%) for various water depths, given in Table A- 3, indicate that removal primarily occurs in the first 20 m of the vegetated swale. Dissolved Phosphorus and dissolved metals were not measured.

Table A- 3. Range of removal efficiencies (%) for a laboratory vegetated swale (from Barrett et al. 1998b). TSS = Total suspended solids, COD = Chemical oxygen demand, TKN = Total Kjeldahl nitrogen, Total P = Total phosphorus.

Contaminant	Distance along swale (m)				Underdrain
	10	20	30	40	
TSS	35-59	54-77	50-76	51-75	73-87
COD	13-61	26-70	26-61	25-79	39-76
Nitrate	(-5)-7	(-5)-17	(-28)-(-10)	(-26)-(-4)	(-8)-(-10)
TKN	4-30	20-21	(-14)-42	23-41	24-41
Total P	25-49	33-46	24-67	34-45	55-65
Zinc	41-55	59-77	22-76	66-86	47-86
Iron	46-49	54-64	72	76	75

The removal efficiencies for the swale at a water depth of 7.5 cm (hydraulic residence time of 8.8 minutes at 40 m) were similar to a swale studied by the Municipality of Metropolitan Seattle (hydraulic residence time of 9.3 minutes). Because these two swales varied in length, slope, and vegetative cover but had similar residence time and removal efficiencies, Barrett claims that this supports the use of residence time as a primary design parameter. The data suggests that a hydraulic residence time of 9 minutes can result in removal of over 80% of TSS over a range of other swale parameters. Removal of TSS was also greater during the growing season than during the winter months but removal of the other contaminants did not appear to be a function of season. The seasonal dependence of TSS removal was attributed to thicker

grass density during the growing season. At these times new grass grew amongst the old, dead grass and the combination resulted in a significantly higher grass density.

The removal efficiency of TSS, COD, total phosphorus, and metals increased with length along the swale but this increase in removal efficiency declined as the distance along the swale increased. For these contaminants, most of the removal occurred in the first 20 m of the swale. For example, over the range of water depths tested (4, 7.5, and 10 cm), the TSS removed in the first 20 m ranged from 80% to 105% of the total TSS removal at 40 m.

Barrett et al. (1998b) also found that TSS removal efficiency dropped in the grassed laboratory swale as water depth increased but the removal of other contaminants was not strongly correlated. This suggests that the filtration action of the blades of grass increases removal of solid particles and, as water depth increases above the grass height, this mechanism has less of an effect. Barrett (2008) also investigated the performance of vegetative buffer strips (as previously stated, Barrett claimed that the side slopes of swales perform as buffer strips) by observing the BMP database data and found little relationship between the grass type, grass height, and TSS removal.

A study by Caltrans (2003) found that vegetative slopes next to roadways consistently reduced total metal concentrations and, contrary to Barrett (2004), dissolved metal concentration reduction occurred frequently. Also, dissolved solids concentrations consistently increased and organic carbon concentrations occasionally increased with distance from the edge of the roadway whereas nutrient concentrations remained unchanged with distance from the edge of the roadway. As was found in other studies, total mass loads of all contaminants transported by the runoff decreased with increasing distance from the edge of the roadway due to the infiltration of stormwater. As shown in [Table A- 4](#), Caltrans (2004) report that vegetated swales can remove a significant fraction of nitrogen and total and dissolved metals.

Table A- 4. Removal efficiencies of roadside vegetated swales (Caltrans 2004).

Contaminant	Percent Removal
NO ₃ -N	65
TKN	67
Total Copper	82
Total Lead	85
Total Zinc	89
Dissolved Copper	76
Dissolved Lead	80
Dissolved Zinc	87

A study by Kearfott et al. (2005) investigated the performance of vegetated side slopes of rural highways in Texas, and found that significant reductions in lead, copper, and COD all occurred within 8 m of the roadway edge but no consistent increases or decreases were found for nutrients. Also, total zinc concentrations were found to increase with distance from the roadway but this was thought to be due to the leaching of zinc from the galvanized metal that was part of the collection apparatus.

Lancaster (2005) measured metal concentrations as a function of distance from the edge of the roadway pavement. At this site total zinc concentrations decreased with increasing distance from the edge of pavement, total copper concentrations decreased slightly, and total cadmium and lead concentrations showed no significant change.

Kaighn and Yu (1996) investigated one roadside swale with a check dam and one without, and found that the swale with a check dam removed 3%, 33%, and 13% of chemical oxygen demand (COD), total phosphorus, and total zinc, respectively. In the swale without a check dam COD concentrations increased by 6%, total phosphorus increased by 0.4%, and 11% of total zinc was removed. The difference in COD and phosphorus removal rates between the two swales was attributed to the check dam. This led the authors to conclude that the use of check dams in roadside swales can increase pollutant removal because the check dam increases both infiltration and particle settling.

Wang et al. (1981) investigated lead (Pb) removal in a grassed swale and found that a 20 meter long swale achieved 60% lead removal and a 60 meter long swale achieved 90% removal. By monitoring four roadway locations that were 41 years old, Ahearn and Tveten (2008) found that total metal concentrations were reduced as runoff moves from the edge of pavement down the embankment but the dissolved metal and nutrient concentrations often increased. For example, at all monitoring locations dissolved copper concentrations increased from the edge of pavement to either 2 meters or 4 meters from the edge of pavement. When total mass loading is considered, however, the total mass of dissolved metals and nutrients decrease due to the large infiltration rates of the embankments that resulted in less total flow volume. Mazer et al. (2001) summarizes a handful of studies on bioswales and road side swales and reported metal removal efficiencies of 21% to 91% and phosphorus removal efficiencies of 7.5% to greater than 80%.

It is believed that optimum metal sorption to soil particles occurs between a pH of 5.5 and 7.0 whereas precipitation of metal hydroxides occur at higher pH values. Thus, Muthukrishnan and Oleske (2008) investigated the effect of adding lime to soil on the adsorption of metal from infiltrated stormwater. A 0.5% lime addition by weight resulted in pH change from less than 5.0 to 7.2 after 144 hours. The amount of increase in pH units and the corresponding time required will vary with each soil, but this study shows that stabilization times can be relatively long. It can also serve as a caution with respect to the ease of overdosing before the final, stabilized pH value is obtained. In other studies, Gray et al. (2006) added lime to soil and

raised the pH to about 6. Gray found the concentrations of aqueous Zn, Cd, Pb, nickel (Ni), and Cu were reduced by 54%, 36%, 76%, 34%, and 86%, respectively, due to the added lime. Also, Zurayk et al. (1997) added 2% to 4% lime to the soil (by weight) which raised the pH and significantly increased phosphorus removal. The effect of lime addition and the increase in pH is a function of initial pH, amending agent (i.e. the lime), and the natural resistance of the soil to pH change. Besides pH, other properties that can affect the performance of the swale included soil texture, organic matter, cation exchange capacity, and soil phosphorus status.

When runoff infiltrates into the soil it takes with it the dissolved contaminants. Studies have shown that most dissolved metals become bound to soil media in the first 20-50 cm below the surface (Weiss et al. 2008). Some metals, along with phosphorus and nitrogen, are also used as nutrients by plants and are assimilated into plant biomass.

2. Factors Affecting Performance of Swales

The performance of a swale depends on many factors including the health and abundance of the vegetation within the swale. Work by Mazer et al. (2001) on bioswales (not roadside swales), may be helpful in understanding optimum conditions for healthy vegetation within grassed, roadside swales. Mazer et al. (2001) report several causes of poor vegetative cover in swales including standing water in the swale for prolonged periods of time, high flow velocities, large fluctuations in surface water depth and soil moisture, excessive shade, and improper installation. Improper installation could be the result of poor design or poor construction practices. For the eight bioswales in western Washington State investigated by Mazer et al. (2001) heavy shade was more important than other environmental factors that limited vegetation. The second most important factor was inundation with water. If water was present for more than 35% of the summer the bioswale had significantly less vegetation. Shade may not be a serious problem for most roadside swales because right-of-ways are usually clear from overhanging vegetation due to line-of-sight regulations. It is, however, important to ensure that any swale is not inundated with water for extended periods of time. This may be especially important to consider when developing a ditch check filter that will slow the conveyance of water within the swale, as in this study.

With regards to estimating contaminant removal in a swale, Wong et al. (2006) modeled the removal of TSS, total nitrogen (TN), and total phosphorus (TP) along the length of swales using a series of completely stirred tank reactors (CSTR's) and the first order removal equation:

$$q \frac{dc}{dx} = -k(C - C^*) \quad (\text{Equation A- 1})$$

where q is the hydraulic loading rate ($m^3/m^2\text{-yr}$), x is the fraction of the distance from swale inlet to outlet, C is the concentration of the contaminant, C^* is the background concentration of the contaminant, and k is a fitted decay rate constant (m/yr). The results of the model, as compared to data collected by Walsh et al. (1997) on actual swales, are shown in [Figure A- 1](#).

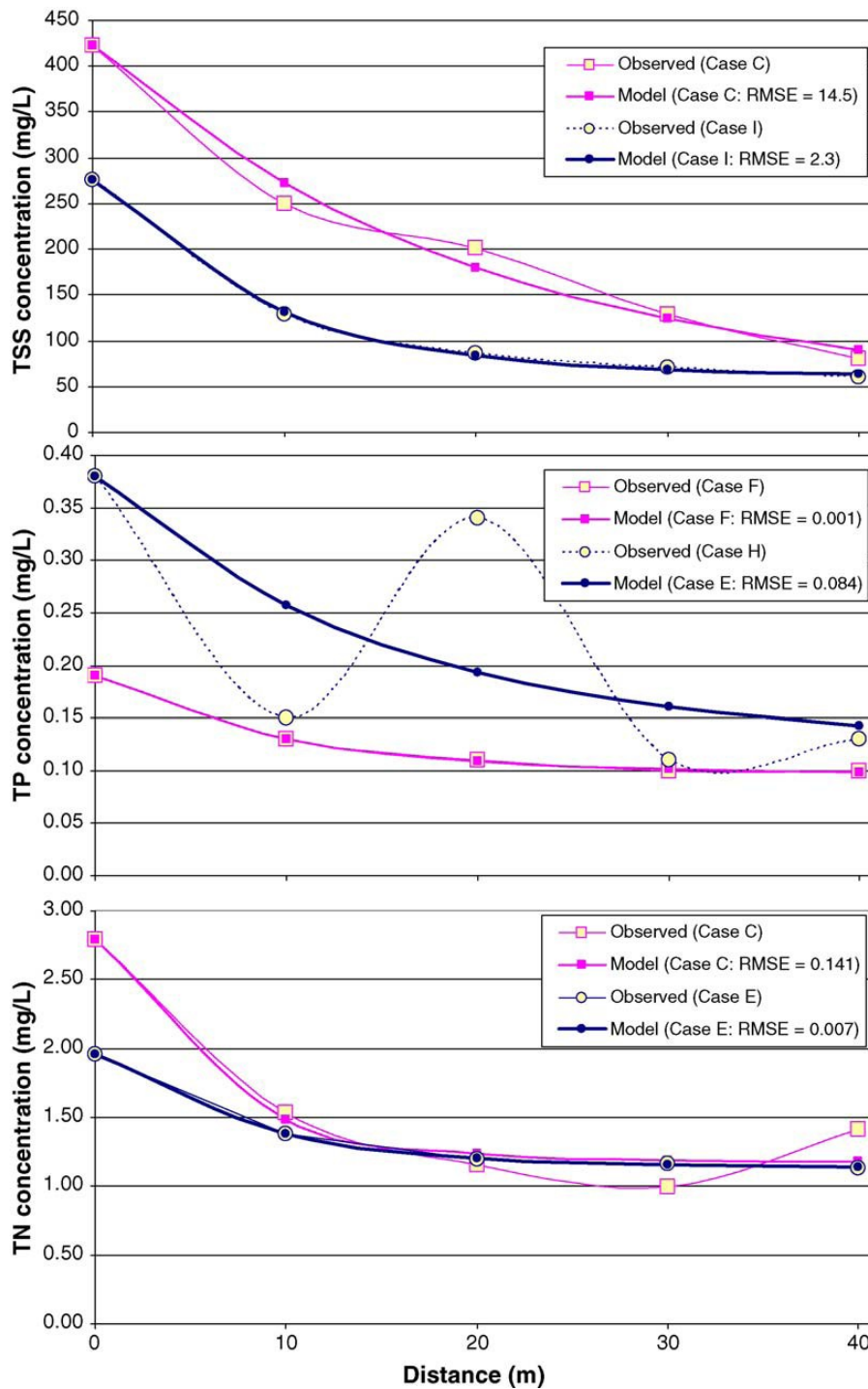


Figure A- 1. Examples of model application to swale performance data from Wang et al. (2006).

As summarized by Backstrom (2003), several authors have proposed design guidelines to optimize the performance of swales with respect to stormwater treatment. Ferguson (1988) recommended a swale length greater than or equal to 60 m, water velocities less than 0.15 m/s, and a residence time of at least 9 minutes. Other authors (e.g. Barrett et al. 1998a), however, found that swale length is not an important design parameter as long as the road runoff is allowed to flow directly down the side slope into the swale. The data suggest that under these conditions the side slope acts as a filter strip and removes most of the contaminants before the runoff begins to flow in the swale parallel to the road. Yu et al. (2001) recommend a swale length of 75 m and a bottom slope of no more than 3%. Backstrom (2003), however, suggests that recommending a certain maximum slope may be misleading because, depending on the condition of the vegetation within the swale, small water velocities can exist in swales with moderate slopes.

3. Performance of Swales in Cold Climates

The effect of cold weather on the performance of stormwater BMPs is a concern in cold climates. The impact of frozen soil on infiltration, short-circuiting of ponds caused by ice cover, and ice formation on BMP structures such as inlets and outlets, etc. has caused many to question the ability of stormwater BMPs to perform as designed during winter months in cold regions. Furthermore, it has been shown that the positive effects of vegetation (i.e. flow retardation and pollutant uptake) have less impact during the winter (Backstrom and Viklander 2000). These concerns have spurred investigations into the impact of below freezing temperatures on BMPs. Studies related to the performance of swales and their associated removal mechanisms in cold weather are reviewed below.

In addition to the impact of cold weather on the stormwater BMP itself, stormwater and snow melt runoff during the winter in cold regions can vary significantly from warm weather runoff. Backstrom and Viklander (2000) performed a literature review on cold climate BMP performance and found that snowflakes get more polluted in the air than rain drops because of their larger surface area and slower fall velocity which allows the snow to absorb more pollutants. After the snow reaches the ground it can be further polluted by dry and wet deposition and gas adsorption. The snowpack can also act as a filter of fine particles and, one can assume, collect them within the snowpack. This is supported by data that shows suspended solid concentrations in snowmelt can be two to five times greater than in rain runoff. It has also been shown that cold automobile engines pollute more than warm engines. For example, cold engines produce two to eight times more particulates than warm engines.

Al-Houri et al. (2009) investigated the impacts of frozen soils on infiltration. Infiltration capacity and hydraulic conductivity are closely linked to the soil water content at the time of freezing. Using the average soil water content to characterize this phenomenon, however, can be erroneous because of unequal internal distribution of water. During infiltration water preferentially resides in larger pore spaces due to the water surface tension and as a result

water is kept out of smaller spaces (Miller 1973). When water drains from the soil, however, it drains more rapidly from the larger pore spaces so that they become filled with air more quickly than small pore spaces. As a result, the longer the time allowed for soil-water redistribution after infiltration ceases and freezing begins, the more air spaces there will be during the next infiltration event. Thus, the time allowed for water-soil redistribution can be used to estimate the soil infiltration capacity of frozen soils.

Al-Houri et al. (2009) studied a loam soil and a sandy loam soil. Both soils were tested under frozen and unfrozen conditions. The soil was allowed to drain for time spans of 2, 4, 8 and 24 hours before freezing commenced. For each specimen, data was collected on three different sections; the top, middle, and bottom sections. The data showed that the loam soil with only a 2-hour drain time before freezing exhibited a decrease in the hydraulic conductivity, K , of the soil by one to two orders of magnitude. Longer drain times resulted in a smaller reduction in hydraulic conductivity, especially for the top sections. For example, the 24-hour drain time specimens had a reduction in K of only 3% to 70%.

For the loam soil, the frozen two hour drain time specimen had an average (of top, middle, bottom sections) hydraulic conductivity that was 5% of the unfrozen specimens. The four and 24-hour tests showed frozen average hydraulic conductivities of 21% and 30%, respectively. The sandy loam soils 24 hour drain time test, by contrast, resulted in an average hydraulic conductivity that was 4% of the unfrozen specimen.

Roseen et al. (2009) compared the annual, summer, and winter contaminant removal efficiencies of various LID practices with respect to influent and effluent event mean concentrations. Roseen et al. determined that, compared to most other LID practices, vegetated swales exhibited larger variations in performance between the summer and winter seasons. [Figure A- 2](#) shows the annual, summer, and winter influent and effluent event mean concentrations for stone and grass swales. The difference between these two kinds of swales is that the bottom of a stone swale is lined with stones instead of vegetation. [Table A- 5](#) lists the annual and seasonal removal efficiencies (RE) and efficiency ratio (ER) for the two kinds of swales.

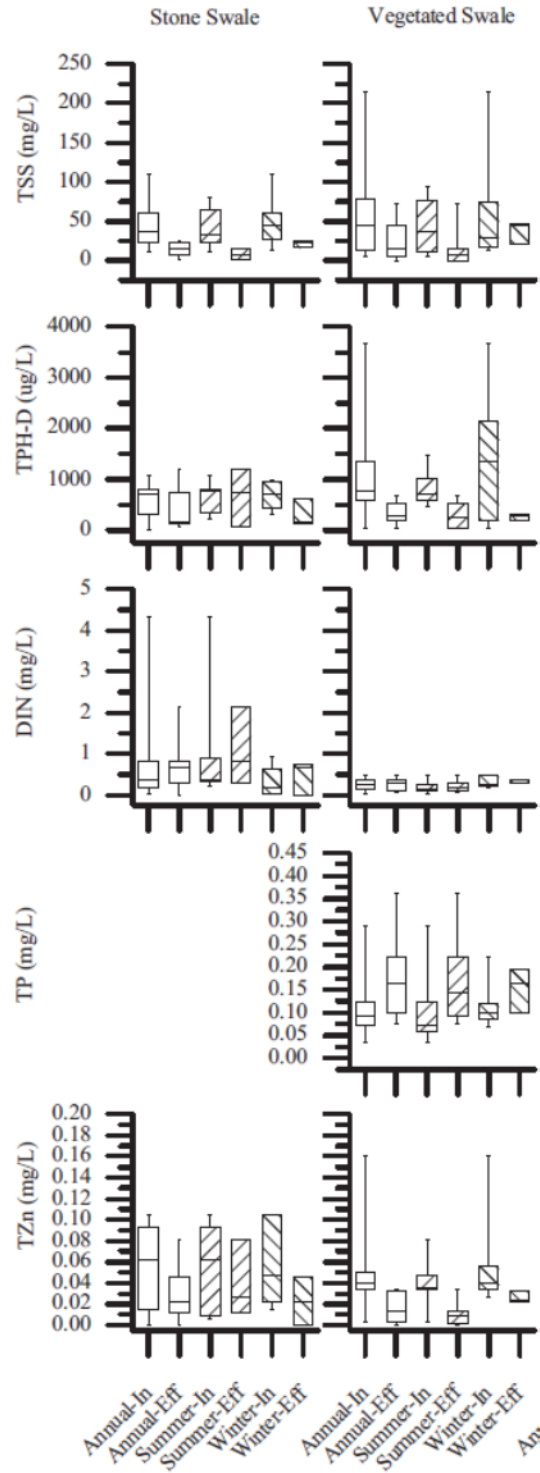


Figure A- 2. Annual and seasonal influent and effluent EMCs for a stone swale and a vegetated/grassed swale. Lines show max and min values, boxes show 75th and 25th percentiles, and the horizontal line in the box indicates the median value (From Rosen et al. 2009).

Table A- 5. Annual and seasonal efficiency ratios (ER) and removal efficiencies (RE) for swales (Data from Roseen et al. 2009). RE is calculated as $RE = 1 - (EMC_{outlet}/EMC_{inlet})$ and the reported value is the median of the entire data set. ER is calculated as $ER = 1 - (\text{average } EMC_{outlet}/\text{average } EMC_{inlet})$. TSS = Total suspended solids, TPH-D = Total petroleum hydrocarbons-diesel, DIN = Dissolved inorganic nitrogen (nitrate, nitrite, and ammonia), Total Zn = Total zinc, and Total P = Total phosphorus.

Contaminant	LID Practice	Annual		Summer		Winter	
		ER (%)	RE (%)	ER (%)	RE (%)	ER (%)	RE (%)
TSS	Stone Swale	68	50	85	80	64	8
	Veg. Swale	52	60	69	68	36	13
TPH-D	Stone Swale	28	33	67	9	72	52
	Veg. Swale	53	67	83	60	75	77
DIN	Stone Swale	-2	-72	-81	--	-34	-11
	Veg. Swale	65	-13	61	-5	1	-13
Total Zn	Stone Swale	48	64	18	72	64	56
	Veg. Swale	72	88	73	92	58	17
Total P	Stone Swale	0	61	31	61	--	--
	Veg. Swale	-38	-95	-40	--	--	-64

4. Improving Dissolved Metal Removal

Urban stormwater runoff typically contains elevated levels of metals in dissolved or particulate form. The metals can originate from vehicle wear, tire and asphalt abrasion, buildings, and other sources (Wu and Zhou 2009). Metals of primary concern (based on toxicity and occurrence) in stormwater are cadmium, copper, lead and zinc (Jang et al. 2005, Rangsviek and Jekel 2005) with up to 50% being in dissolved form (Morrison et al. 1984). Copper, lead, and zinc are detected in urban runoff over 90% of the time (Pitt 1996) and, where erosion is minimal, soluble pollutants can exceed 80% of the total pollutant load (Faucette et al. 2009). The particulate fraction of metals is not readily bioavailable and can be removed via settling or filtration. Thus, metals that occur primarily in particulate form (such as lead), while toxic, may not typically be of concern in stormwater treatment because they can be removed through settling or filtration. The dissolved fraction, however, typically passes through stormwater treatment devices and into receiving water bodies. With up to 50% of cadmium, copper, and zinc being in dissolved form, attention is being given to the development of methods that can remove these metal ions from stormwater runoff. One possible solution is to add material (i.e. an enhancing agent) to soil or sand filter media that will adsorb or otherwise retain dissolved metal cations.

Metal cation removal from aqueous solution may occur through four different mechanisms; cation exchange, cementation, adsorption, and metal hydroxide precipitation (Rangsviek and Jekel 2005, Zhang et al. 2008). The mechanisms are pH dependent and more than one mechanism can occur at the same time, even on the same sorbent. In cation exchange

processes, positively charged ions on the sorbent are displaced by metal cations that have a higher affinity for the site. Thus, metal cations are removed solution and previously sorbent-bound cations enter solution. At very low pH, metal cations must compete with hydrogen ions for the exchange sites and metal removal tends to be reduced (Kwon et al. 2010). Cementation involves redox sensitive compounds being reduced to insoluble forms (Rangsivek and Jekel 2005), surface adsorption occurs when ions are adhered preferentially to the surface of the adsorbent, and metal hydroxide precipitation occurs at high pH values where the metal ions will form a precipitate with free hydroxide ions (OH⁻).

There are many materials with the ability to adsorb metal cations in aqueous solutions including various forms of iron, alumina, activated carbon, compost, mulch, wood bark, grass clippings, and many other plants or plant parts. An ideal enhancing agent for the removal of metal ions from stormwater would:

- a. Be inexpensive and readily available
- b. Be easily mixed into the infiltration media without any health or safety issues
- c. Have relatively fast reaction/sorption kinetics because contact times between the water and enhancing agent are relatively short
- d. Not alter the pH or other water quality parameters to unacceptable values
- e. Not dissolve and be washed away or plug the filter media
- f. Have a high capacity for metal retention
- g. Not release other contaminants such as phosphorus, nitrogen, or other metals.

After a discussion of dissolved metal removal capabilities of natural soils, studies that have investigated various enhancing agents and their ability to remove metal ions from solution are reviewed below.

4a. Natural Soils

Many natural soils have demonstrated the ability to remove metal ions from solution and most metals adsorb to soil particles in the top 20-50 cm of soil (Mikkelsen et al. 1997, Dierkes and Gieger 1999, Dechesne et al. 2004, Zimmerman et al. 2005, Sun and Davis 2007). Elliot et al. (1986) investigated soil adsorption of Cd, Cu, Pb, and Zn and found that for all metals, a higher pH resulted in more metals being sorbed by the soil. Possible explanations include the adsorption of metal-hydroxo complexes (Metal-OH⁺), hydrolysis of aluminum on exchange sites, and finally competition between protons and metal cations and acid-catalyzed dissolution of reactive oxide sites that may occur at low pH values.

In batch and column experiments, Plassard et al. (2000) investigated the adsorption of Cd, Pb, and Zn by a soil with a carbonate content of 227 mg/g. In a review of previous studies, Plassard et al. (2000) states that cadmium can be retained by the formation of an ideal surface solution with cobalt carbonate (CoCO₃), and zinc and copper may be precipitated as hydroxides or hydroxycarbonates. Another author found that the reaction between metals and

carbonates results in metal incorporation into the solid lattice, not surface adsorption. All of Plassard's experiments were performed at pH = 8.2 with highly buffered soils. In the batch experiments at metal concentrations of 5×10^{-4} mol/L, all metals were bound to the soil after 4 hours, mostly to the acid-soluble fraction of soil. Increases in initial metal concentration, however, lead to weaker retention. For example, in batch experiments at a cadmium concentration of 5000 $\mu\text{mol/L}$, 27% of Cd remained in solution after 4 hours. Cadmium, it must be noted, behaves differently than other metals as it reacts more slowly and, thus, is typically retained at a lower rate. Metals were removed to a lesser extent in the column experiments, probably due to preferential flow paths and/or shorter contact times. Plassard et al. (2000) concluded that even in situations where the removal rates were high the removal forces (precipitation and cation exchange) were weak and that there is a risk of metal release when conditions change.

Elliot et al. (1986) investigated the effect of organic matter on metal adsorption by performing batch studies with a 24-hour contact time. It was determined that when organic matter was removed from two organic soils, zinc adsorption was greater than cadmium adsorption. When organic matter was not removed, however, cadmium adsorption was greater than zinc adsorption. Based on these results it appears that organic matter prefers adsorption with cadmium, although Elliot et al. (1986) point out that other studies have found equal adsorption tendencies between cadmium and zinc. In some soils all metal adsorption capabilities might be due to organic matter but this is not true for all soils. The investigators also state that for soils without organic matter, the addition of organic matter may increase metal adsorption under acidic conditions and that this will enhance cadmium removal more than zinc.

Blecken et al. (2009) investigated the effect of the saturated (or submerged) zone in soil media on dissolved contaminant removal by the soil. Submerged zones, in combination with organic carbon zones, have been shown by other authors to support nutrient and metal removal, especially copper. Eighteen laboratory biofilters were constructed out of PVC pipe; some filters contained a submerged zone and some did not (Blecken et al. 2009). Lead and zinc removal was large and cadmium was always removed to levels below the detection limit regardless of whether a submerged zone was present. Copper removal, however, was 12% greater in filters with a submerged zone. The authors reasoned that the submerged zone creates anoxic (or partially anoxic) conditions which cause more of these metals to adsorb to sediments. After three weeks of drying, elevated metal concentrations were found in filter effluent but no effect was observed for shorter drying times. Filters with a submerged zone also did not discharge lead, even after long dry periods. Decreased metal removal after drying could be attributed to leaching of metals from the soil, mobilization of fine sediment, and cracking of the soil and creation of preferential flow paths. Blecken et al. (2009) also observed that metals were removed in the top portion of the soil because below a depth of 20 cm all metal concentrations were below detection limits.

Lassabatere et al. (2004) investigated the effect of geotextiles on contaminant removal, including metal (Cd, Pb, Zn) adsorption, in calcareous soils. It was determined that geotextile fabric can filter particles, trigger development of microorganisms that may interact with contaminants, and modify or control flow. While the geotextile material did not retain metals, it did modify mechanisms of metal retention in the surrounding soil. When geotextile material was placed dry it tended to homogenize the flow. This resulted in better contact between the soil and water and more metal removal. All tests were performed on calcareous sand and the results may not apply to other soil types or other metals.

4b. Activated Carbon

Van Lienden et al. (2010) reviewed the ability of six commercial and six non-commercial activated carbon (AC) materials to remove copper and zinc ions from water. Non-commercial AC included those derived from straw, hulls, and shells of almonds, pistachios, pecans, and walnuts. The performance of each AC was investigated in the laboratory using batch studies with a contact time of 72 hours and a prepared solution containing the metal ions. The batch studies were followed by experiments using actual stormwater runoff samples. Most of the ACs tested, including all of the commercial products, demonstrated a greater affinity for copper than for zinc. The AC that was derived from rice hulls had the largest capacity to remove zinc whereas a commercial AC had the largest copper removal capacity. When tested in actual stormwater, however, the copper sorption dropped by up to 80% due to competition between metal ions and other substances in the water. The authors concluded that 1 kg of AC derived from rice hulls could treat up to 46 m³ of California highway stormwater runoff that contained the 90th percentile concentration of copper. The zinc treatment volume, however, would be 7 m³ and would control the design of a stormwater filter. With regards to copper removal, the rice-derived AC could treat 20.9 times as much California highway runoff than the best commercially AC tested and 4.9 times as much water with respect to zinc removal.

Genc-Fuhrman et al. (2007) tested a host of enhancing agents including a commercially supplied granulated activated carbon (GAC) for their ability to remove arsenic, cadmium, chromium, copper, nickel, and zinc from solution using batch studies with a 48 hour contact time. The GAC in their study ranked behind alumina, bauxsolcoated sand, granulated ferric hydroxide, and fly ash in terms of overall removal effectiveness. The fly ash, however, increased pH values to over 9, whereas the GAC increased pH valued to between 8 and 9.

The ability of activated carbon impregnated with the surfactants sodium dodecyl sulfate (SDS), sodium dodecyl benzene sulfonate (SDBS), or dioctyl sulfosuccinate sodium (DSS) to remove cadmium ions from solution was investigated by Ahn et al. (2009) using batch studies with a 48 hour contact time. Impregnating the surface increased the number of active cation exchange sites; it was theorized that the new sites could be the heads of the surfactants arranged towards the water. All modified GAC removed Cd well, even at pH =2, with removal rates increasing linearly with pH. It was determined that SDS was the most effective enhancing agent of those

tested. Also, the surfactants covered up acid groups and decreased acidity. Finally, Ahn et al. (2009) concluded that the kinetics of the adsorption reactions is well described by a pseudo-second order kinetic model.

4c. Alumina

Alumina, or aluminum oxide (Al_2O_3), can remove heavy metals from solution by precipitation and adsorption. There are many different grades of alumina but in a batch study experiment (48 hour contact time) by Genc-Fuhrman et al. (2007) that investigated the ability of 11 different sorbents to remove arsenic, cadmium, chromium, copper, nickel, and zinc ions, alumina was determined to have the largest overall removal efficiency of all sorbents. The alumina used by Genc-Fuhrman et al. (2007) was supplied by Haldor-Topsoe in Denmark and was a waste product of their manufacturing process of catalysts (email communication with Professor Anna Ledin, co-author). In this study the alumina outperformed activated bauxsol-coated sand, bark, bauxsol-coated sand, fly ash, granulated activated carbon, granulated ferric hydroxide, iron oxide-coated sand, natural zeolite, sand, and spinel (MnAl_2O_4). It was observed that as the pH and initial metal concentration increased, oversaturated minerals and salts also increased. Thus, precipitation, not adsorption, may have been the primary removal process in these tests. Regardless of the mechanism, the alumina removed all metals to levels below Dutch emission limit values.

The high removal rates of alumina were attributed to the high surface area of alumina and favorable surface reactions with metal ions. Alumina removes cations through mechanisms such as surface complexations with hydroxide groups, pore diffusion, and adsorption. Although alumina was deemed the best overall performer, some sorbents did outperform alumina with respect to removal of certain metals.

Baumgarten and Kirchhausen-Dusing (1997) presented a chemical reaction model to predict the concentration of aluminum ions, other metal ions, and pH as a function of the quantities of alumina, volume of liquid, metal concentrations, and other variables. They also described adsorption with a Henry isotherm.

4d. Compost

Compost is generally derived from leaves, grass, and woody debris and do not contain food or animal waste. It is the organic fraction of compost that has the ability to adsorb metal ions from solution. The organic material contains binding sites such as amine, carboxyl, phenolate, and thiol groups that can complex or exchange metals and remove them from water (Nwachukwu and Pulford 2008). Precipitation may also occur, usually by an anion such as phosphate, which forms an insoluble salt with the metal, or by raising the pH to precipitate the metal as its hydroxide.

Nwachukwu and Pulford (2008) tested bonemeal, coir (commercially available blocks of dried coconut husks), compost, green waste compost, peat, and wood bark for their ability to adsorb lead, copper, and zinc in batch laboratory studies with a contact time of one hour. The materials with the three largest metal capacities were, in order of decreasing capacity, green waste compost, coir, and compost. The Langmuir sorption maxima were approximately 87 mg Pb/g (coir and green waste compost), 30 mg Cu/g (compost and green waste compost), and 13 mg Zn/g (compost and green waste compost) all in 0.001 M Ca(NO₃)₂. The affinity of metals for the compost materials was consistent with other materials tested (in this and other studies) and was in the order of Pb > Cu > Zn.

Adsorption was dependent on ionic strength of the solution and competition with other metals. For example, larger background salt concentrations resulted in less lead removal by compost, coir, wood bark, and green waste compost. Also, testing all metals in one combined solution reduced sorption by all materials due to competition. Lead sorption, when in competition, was reduced by about 40-50%, copper was reduced by 60% to 70%, and zinc sorption was decreased by variable amounts under competition.

Seelsaen et al. (2006) used batch studies with a 24 hour contact time to test compost, sand, packing wood, ash, zeolite, recycled glass, and Enviro-media (a manufactured material containing either selected organic matter or a blend of organics, minerals, specialized aggregates, soil and other ingredients) for their ability to remove copper, lead, and zinc ions from solution. The relevance of the Enviro-media mixes is that based on its performance, the authors concluded that a combination of sand and alternative materials could be used to treat stormwater. With regards to metal removal, compost was ranked the best but it also leached large amounts of dissolved organic carbon (DOC) into the water.

Faucette et al. (2009) investigate the use of socks filled with compost to remove sediment, NH₄, NO₃, fecal bacteria, metals, and petroleum hydrocarbons. Removal efficiencies for soluble metals tested (Cd, Cr, Cu, Ni, Pb, and Zn) ranged from 17 to 72%.

Composted Manilla grass was investigated by Kahn et al. (2009) to determine whether pH and moisture content (MC) values that create large volume reduction in compost (compost's main purpose), also promote cadmium, copper, lead, and zinc cation removal. The experiments were performed using laboratory batch studies with a contact time of 5 hours. 30% MC and an initial pH of 8 resulted in compost with the largest cation exchange capacity whereas 30% MC and an initial pH of 7 had the maximum volume reduction. Kahn et al. (2009) concluded that 30% MC was optimum for maximizing volume reduction and cation exchange capacity at all initial pH values.

Compost, although shown to be effective at removing metal ions, releases or has the potential to release phosphorus and nitrogen into the water. Phosphorus or nitrogen are typically the limiting nutrient for plant and algae growth and therefore are often a target for removal from

stormwater. Thus, using an enhancing agent that can release phosphorus and/or nitrogen may be more detrimental than beneficial. A two-stage filter, however, in which the upstream first stage removes metals but releases nutrients followed by a second stage that removes nutrients may be effective.

4e. Crustacean Shells

Shells from crabs, shrimp, and other crustaceans are composed of chitin (poly-N-acetylglucosamine), protein, and CaCO_3 and have been shown to have a relatively large capacity to remove heavy metals from aqueous solutions. Chitin complexes investigated by Robinson-Lara and Brennan (2010) removed aluminum ions more quickly (3 days) than lactate (12 days) and spent mushrooms (37 days) and was the only complex able to remove manganese (73% removal). Aluminum removal was consistent with the formation of hydroxides and/or alunite ($\text{KAl}_3(\text{SO}_4)_2(\text{OH})_6$). Iron was also removed by chitin which was consistent with the precipitation of Fe(III) oxides and Fe(II) sulfides, as well as surface adsorption. Finally, the chitin complex increased the pH from 3 to neutral in three days.

Bailey et al. (1999) discuss chitosan, which is a complex that can be produced chemically from chitin, is found naturally in some fungal cell walls, and has a large capacity for metal adsorption. It is estimated that one to four million pounds of chitosan could be produced for \$1-2/lb. Chitosan, however, is highly soluble in water but can be made essentially insoluble by cross-linking it with glutaraldehyde.

The adsorption capacity of chitosan varies with crystallinity, affinity for water, percent deacetylation, and amino group content but studies have found extremely large capacities of 136 to over 500 mg Cd/g, 27.3 mg Cr(VI)/g, several hundred mg Hg/g, and almost 800 mg Pb/g. In other studies N-acylation was shown to increase the porosity of chitosan (which is naturally non-porous) which increased metal capacity by 20% but cross-linking to reduce solubility was shown to decrease capacity by 37%. Other researchers have increased the capacity of chitosan by substitution of functional groups such as organic acids.

4f. Ferrous-based Materials

Iron based enhancing agents, which have the capacity to remove dissolved phosphorus from solution, have also demonstrated the ability to remove metal ions. For example, Namasivayam and Ranganathan (1995a) used Fe(III)/Cr(III) hydroxide, a waste product from the fertilizer industry, to remove cadmium, nickel, and lead ions from solution. In batch studies with contact times of 1 or 5 hours the amount of metals sorbed increased as the sorbent dose increased and particle size decreased. Contrary to some studies (Nwachukwu and Pulford 2008), metal adsorption was not affected by changing the ionic strength of the solution. When all metals were combined in one solution, lead demonstrated the strongest affinity for the sorbent, followed by cadmium and then nickel, due to competition. Namasivayam and Ranganathan (1995a) also performed fixed bed experiments in which 3 grams of adsorbent was packed into

a 1.4 cm inside diameter glass column (2.5 cm bed height) and a solution containing one of the three metal ions (200 mg Pb/L, 80 mg Cd/L, or 50 mg Ni/L) was pumped through the adsorbent at a rate of 10 mL/minute. The breakthrough curves are shown in Figure A- 3.

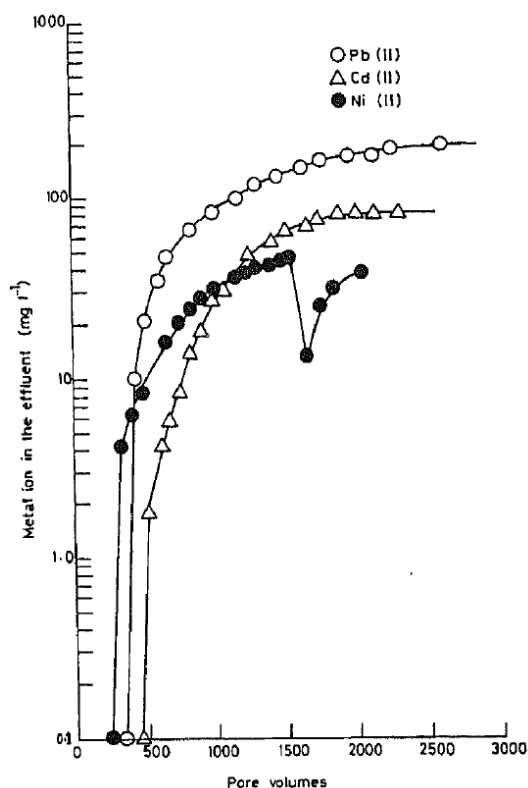


Figure A- 3. Breakthrough curves for the adsorption of metals on waste Fe(III)/Cr(III) hydroxide (From Namasivayam and Ranganathan (1995a)).

In a paper investigating only cadmium ion removal, Namasivayam and Ranganathan (1995b) reported that the mass of cadmium ions removed ranged from approximately 20 to 40 mg Cd/g sorbent as the concentration of the sorbent increased from 50 to 140 mg/L. The time required to reach equilibrium was approximately 300 hours. Adsorption also increased with temperature; equilibrium values of about 25 mg/g were reported at 20°C and about 33 mg/g at 40°C. The effect of temperature was attributed to increased pore sizes in the adsorbent and/or activation of the adsorbent surface. Adsorption also appeared to be endothermic and spontaneous and, as is typical of cation adsorption of metal oxides, increased with pH. Finally, as pH was lowered, previously sorbed metals ions were desorbed; 70% of the cadmium ions desorbed at pH = 3.8, the smallest pH value tested.

Rangsivek and Jekel (2005) used zero valent iron (ZVI), or Fe^0 , to remove copper and zinc ions from solution in batch studies with a contact time of 48 hours. Their ZVI was in the form of scrap iron obtained from an ASTM A284 grade steel cylinder that was 98% Fe^0 and was ground into particles that ranged from 0.4 to 1.25 mm in size. Most of the copper removed was determined to be reduced in a thermodynamically favored reaction by Fe^0 and transformed into insoluble Cu^0 or Cu_2O . Unlike copper, zinc removal required the precipitation of iron oxides that acted as adsorption sites. Zinc removal also increased as dissolved oxygen, ionic strength, pH, and temperature increased but the rate of zinc removal was half that of copper. Metal uptake rates decreased with increasing amounts of dissolved organic carbon as these substances form complexes with metals and Fe^{2+} .

Rangsivek and Jekel (2005) examined the removal processes of ZVI in detail and found that large molecular organics can compete with metal ions for adsorption sites and that the long-term performance of ZVI may be limited or governed by the formation of layers of iron and cuprous oxides. Also, under acid conditions with no dissolved oxygen, only 0.88 mg iron was required to remove 1 mg Cu^{2+} but in stormwater runoff that value increased to about 5 mg of iron. The difference was attributed to iron consumption by dissolved oxygen and accumulated intermediate products such as Fe^{3+} . The presence of DO, however, significantly increased zinc removal as DO increased the corrosion of iron.

Copper removal was not significantly affected by temperature but zinc removal increased by a factor of seven as the temperature increased from 5 to 35°C. The impact of temperature was attributed to increased iron dissolution rates at higher temperatures which resulted in more iron being available to remove zinc. It also appeared that some copper (7% to 9%) formed metal-DOC complexes that could not be removed by the ZVI. The DOC also reduced copper removal rates by a factor of two due to the formation of complexes and competitive adsorption. Zinc removal was reduced by a factor of 4.6 in the presence of DOC. Finally, ionic strength had little impact on copper removal but zinc removal increased from 75% in deionized water to greater than 95% in water with large ionic strength.

In a comparison of 11 different sorbents, Genc-Fuhrman et al. (2007) ranked granulated ferric hydroxide (GFH) only behind alumina and bauxsol-coated sand in terms of overall metal ion removal performance. The GFH was a commercially available product sold under the name Ferrosorp Plus and was later investigated for its ability to remove As, Cd, Cr, Cu, Ni, and Zn from solution (Genc-Fuhrman et al. 2008). The GFH increased the pH of the water from 6.7 to 8.2 and complete breakthrough occurred after 100 days (or about 800 bed volumes). In this study, humic acid was found to reduce the uptake of metals; Cr was most negatively affected with 100 mg/L of humic acid reducing its removal by 81%. Adsorption capacities were also found to decrease in the absence of light because, in the presence of organic matter such as humic acid, light can reduce the Fe(III) in the GFH to Fe(II).

Wu and Zhou (2009) also investigated the ability of Ferrosorp Plus and Ferrosorp RWR to remove arsenic, cadmium, chromium, copper, nickel, and zinc from solution using batch studies with a contact time of 48 hours. The Ferrosorp Plus leached chromium (14.3 $\mu\text{g/L}$) and zinc (120 $\mu\text{g/L}$) in control batches of clean water. The Ferrosorp products had a large affinity for cadmium, nickel, and zinc but were not as effective at removing copper, arsenic, and chromium. The authors suggested that a Ferrosorp/zeolite mixture could be effective for stormwater treatment; the Ferrosorp would remove cadmium, nickel, and zinc and the zeolite could remove copper (and possibly other metals). The performance of Ferrosorp, however, could be reduced when used in actual stormwater runoff due to the formation of metal/organic complexes and competition between metal ions and other substances for adsorption sites.

4g. Magnesium

Zhu et al. (2009) used chromatographic silica gel as the host matrix and impregnated it with magnesium chloride. Granules were then dried for 3 hours and calcined at 773 $^{\circ}\text{K}$ for 4 hours. The result was a material that had a capacity for metal adsorption 15 to 30 times greater than silicon dioxide. At initial copper and nickel concentrations of 50 mg/L, more than 90% of the metal ions were removed after 8 hours at a pH value of 5 or above. Removal rates decreased as pH decreased below 5 as metal ions had to compete with hydrogen ions for sorption sites. If the metal removal process was all cation exchange, the removal would have been completed in a matter of minutes, not hours. Thus, the authors note that a process that is slower than ion exchange is also involved in the uptake of metals. Adsorption capacities also increased with temperature indicating endothermic adsorption processes are occurring.

4h. Manganese and Manganese Oxide

Shi et al. (2008) used ground, natural manganese ore (rhodochrosite) to remove lead and copper from solution. Like other adsorption studies, pH was found to greatly impact the adsorption process. Adsorption equilibrium was attained within 60 minutes and capacities were on the order of 1-5 mg/g for copper and 1-11 mg/g for lead. The authors concluded that diffusion of the metal ions into the bulk crystal was so difficult that it essentially did not occur and that all adsorbed metal ions were on the surface of the particles.

Manganese oxide coated media (with the coating chemically applied under laboratory conditions) has been shown to remove metal ions from water and has similar adsorption properties as granular activated carbon and iron-oxide coated sand (Liu et al. 2004). Removal mechanisms include adsorption, surface complexation ion exchange, and filtration (Liu et al. 2005). In their study, Liu et al. (2004) coated polymeric media with manganese oxide and found that the coating significantly increased metal removal. Over 50% of the metals (Cd, Cu, Pb, and Zn) were removed in the first 30 minutes with 90% removal occurring within 5 hours. Under conditions where the metals were combined and competing for adsorption sites, the amount of metals adsorbed in decreasing order were $\text{Pb} > \text{Cu} > \text{Cd} > \text{Zn}$. Also, at pH = 7 the capacity for Pb removal was found to be several times larger than that at pH = 6.

In a later column study, Liu et al. (2005) compared manganese oxide coated materials (polymeric, cementitious materials, and sand) with iron oxide coated sand, silica sand, granular activated carbon, and cementitious material and found that the manganese oxide coated materials had the best overall behavior for removal of Pb, Cu, Cd, and Zn. In these experiments the loading rates ranged from 20 to 80 L/m²-minute, which is typical for stormwater BMPs.

4i. Plant Biomass

Various plant materials that have been investigated have demonstrated the ability to remove metal ions from aqueous solutions. For example, Al-Asheh et al. (2000) demonstrated that pine bark has the ability to remove Cd, Cu, and Ni ions from water. This study focused on which isotherm models fit best in single and multi-metal solutions. Freundlich and Sips models most accurately modeled solutions with one metal ion. Solutions containing a combination of two metal ions, however, fit the Extended-Langmuir, Extended-Freundlich, and ideal adsorption solution theory (IAST) best. In general, bark is effective because of its large tannin content.

The polyhydroxy polyphenol groups of tannin enable adsorption to occur as ion exchange takes place between metal cations and phenolic hydroxyl groups (Bailey et al. 1999). Tannins can also discolor water by releasing soluble phenols. Pretreatment with acidified formaldehyde, acid, base, or formaldehyde can prevent the water from becoming discolored. Peanut skins, walnut expeller meal, and coconut husks have similar metal removal capacity as bark. Other plant material with demonstrated metal removal capability include seaweed, peat moss, modified orange peel, mulch (from various types of bark), barley straw, manila grass, and dried wheat stem (Bailey et al. 1999, Jang et al. 2005, Pehlivan et al. 2009, Tan and Xiao 2009, Kahn et al. 2009, Feng et al. 2009). The above mentioned orange peel was modified by hydrolysis with a copolymer which increased the adsorption capacity for copper to 289 mg/g which was 6.5 times greater than the non-modified peel (Feng et al. 2009). The kinetic experiments on the modified orange peel showed that saturation was attained after 5 minutes and pH between 4 and 6 appeared to be the most adequate.

The main parameters affecting copper and lead adsorption to barley straw were initial metal concentration, amount of adsorbent, contact time, and pH. The percent lead and copper adsorbed increased with increasing pH with the maximum adsorption occurring at a pH of 6. Equilibrium capacities were 4.64 mg/g for copper and 23.3 mg/g for lead after two hours (Pehlivan et al. 2009).

In experiments reviewed by Bailey et al. (1999), seaweed had a capacity of 67 mg Cd/g seaweed and peat moss, due to large cation exchange capacity (CEC), had relatively large metal removal capacities of approximately 5 mg Cd/g, 44 - 119 mg Cr(VI)/g, and 20 - 230 mg Pb/g.

Algae's (*Synechocystis* sp.) cadmium ion removal capabilities was investigated by Ozturk et al. (2009) who studied dried algae, immobilized and dried algae, and immobilized and live

algae. Immobilized algae were in the form of alginate gel beads. Dried algae had the largest capacity (75.7 mg Cd/g) compared to immobilized dried (4.9 mg Cd/g) and immobilized live algae (4.3 mg Cd/g) with equilibrium being established in approximately 15 minutes. The optimum pH and temperature for removal was 7 and 25° C, respectively. Capacities dropped off at temperatures higher than 25° C; a phenomenon that was attributed to larger temperatures affecting the stability and configuration of cell walls and ionization of chemical functional groups. Also at small pH values, surface ligands are associated with H_3O^+ that restrict the approach of metal cations.

Tan and Xiao (2009) investigated cadmium uptake by dried wheat stems that had been passed through a 100 mesh and were methanol esterified. In these experiments 200 mg of wheat stem was mixed with 50 mL of solution at cadmium concentrations ranging from 0.1 to 1.2 mmol/L. Adsorption increased with increasing pH from 2 to 5 but remained constant at a value of 0.030 mmol Cd/g from pH =5 to pH = 8, the maximum pH investigated. Kinetics were relatively rapid with 90% of cadmium being removed in the first 10 minutes and 95% removed in the first 20 minutes. As with other studies, carboxyl groups (COO^-) were found to be important in the cadmium binding process as were hydroxyl (OH^-) and O^- ions.

4j. Zeolites and Clays

Zeolites and clays have adsorption capabilities that result from a net negative charge on the structure of fine-grain silicate minerals. In zeolites this negative charge results from Al_3^+ replacing Si_4^+ in the tetrahedral (Bailey et al. 1999). The negative charge is neutralized by the adsorption of positively charged species, giving zeolites and clay the ability to attract and hold metals cations. In zeolites, Na^+ , Ca_2^+ , and K^+ occupy spaces within the lattice structure and can be replaced by metals in an ion exchange process. Bailey et al. (1999) reported clay adsorption capacities of 4 - 16.5 mg Cd/g, 0.5-50 mg Cr(VI)/g, and 0.22 to 58 mg Pb/g with the value for Cr(VI) being the largest reported value for all substances reviewed. Fly ash has also been reported to have metal adsorption capability but a portion of fly ash (about 6%) has been documented to be clay (Zhang et al. 2008).

Genc-Fuhrman et al. (2007) used batch studies (48 hours) to test naturally occurring Clinoptilolite, one of the most abundant of the naturally occurring zeolites, and found that its affinity for metal removal is in the order of $\text{Cu} > \text{Cd} > \text{Zn} > \text{Ni} > \text{As} > \text{Cr}$. Pitcher et al. (2004) investigated the ability of a synthetic zeolite and a natural zeolite, mordenite, to remove Cd, Cu, Pb, and Zn from synthetic stormwater runoff and actual motorway runoff using batch studies with a contact time of 10 minutes. The synthetic zeolite removed greater than 91% of all metals from both solutions but increased sodium levels to 295 mg/L, which is significantly above the freshwater standard of 170 mg/L. The synthetic zeolite also increased the pH to between 8.5 and 9.0 regardless of the starting pH. Finally, the synthetic zeolite also removed calcium. This could be important because hardness is thought to be a factor in reducing the toxicity of metals.

In Pitcher et al. (2004) mordenite was less effective than the synthetic zeolite tested; the mordenite removed 42% to 89% of all metals in synthetic stormwater and only 6% to 44% in motorway runoff. As with many other sorbents the preference for adsorption was $Pb > Cu > Zn \sim Cd$. Seelsaen et al. (2006) tested seven different sorbents including one zeolite and found that zeolite removed 95% of zinc and 52% of copper in batch tests using synthetic stormwater at concentrations of 5 mg Cu/L and 27 mg Zn/L and a contact time of 24 hours. These concentrations are about 50 times greater than those typically found in stormwater runoff. Thus, the removal efficiencies may not directly correlate to typical stormwater metal concentrations.

4k. Other Materials

Mahmoud et al. (2010) used silica gel coated with chelating agents to remove Cd, Cr, Cu, Fe, Mn, Ni, Pb, and Zn from solution in batch studies with a contact time of 20 minutes which was long enough for equilibrium to be achieved. Silica gel was used because it is mechanically and thermally stable, experiences less swelling and shrinking, and is less susceptible to microbial effects compared to other organic polymers. It was found that optimum metal recovery occurred in the pH range of 7 to 8.5, except for Cr(III) which experienced maximum removal at pH = 6.

Hu et al. (2010) coated Fe₃O₄ microspheres with SiO₂ to remove lead and mercury ions from industrial wastewater. Lead removal was greater than 95% at a pH of 6 (contact time was not reported). Mercury removal was slightly less than lead removal but still greater than 90%. The microspheres, due to their iron core, can be removed from solution by a magnet and can be regenerated in a weak acidic solution and reused.

Munaf et al. (2009) used columns packed with perlite, a volcanic rock that expands up to 30 times its original size when heated quickly from 800 to 1100o C and has surface hydroxyl groups, to remove Cd, Cr, Cu, and Zn from solution. Chemical analysis of the perlite revealed that it was approximately 72% - 77% SiO₂, 13% - 17% Al₂O₃, 4%-6% K₂O, with the remaining constituents being miscellaneous materials. Loading rates and contact times were not reported. At a pH value of 6, Cr, Cu, and Zn removal ranged from 75% - 90% and Cd removal was 75%. At a pH of 1.0, 83% - 99% of all metals desorbed.

Fly ash consisting of 38% SiO₂, 18.4% silt, and 6% clay was used by Zhang et al. (2008) to remove Cu, Pb, and Zn from solution in batch (contact time of 24 hours) and column studies. The columns had a length of 14.3 cm with influent being pumped up through the column at 3 cm/hr. It was determined that a sand/fly ash mixture in a bioretention cell had the capacity to removal heavy metals from stormwater runoff for over 900 years. The mixture with 5% fly ash, however, initially increased the water pH to over 10 and the value remained above 9 after treatment of 240 pore volumes.

Other materials reviewed or discussed briefly in the literature include bonemeal (Nwachukwa and Pulford 2008), sand, recycled glass, packing wood (Seelsaen et al. 2006), sawdust waste, bagasse pith waste from sugar mills, maize cob waste, wool waste, and blast furnace slag (El-Geundi 1997).

Contact times with conventional sand media in stormwater filters can be on the order of one to a few hours. Accounting for the fact that any enhancing agent added to filter media with the intent of capturing dissolved metals will probably be a small fraction (typically <10%) of the media by mass, contact times with the enhancing agent may be as low as three to five minutes (Erickson et al. 2010). Thus, in order to achieve significant levels of dissolved metal removal, removal kinetics must be relatively rapid. As previously discussed, the enhancing agent must also be readily available, cost-effective, and not alter water quality parameters to unacceptable levels. This most likely means that the enhancing agent will not need to be modified in intensive laboratory procedures. Of the materials reviewed, ferrous based materials, alumina, zeolites (if sodium leaching can be minimized), and some activated carbons appear to warrant further investigation.

5. Summary

Field studies on swales show that infiltration of runoff volumes ranging between 9 to 100% can be achieved by swales. Factors such as soil permeability, initial moisture content, compaction of soil, presence of vegetation, i.e. plant or tree roots. Significant removals of pollutants such as suspended solids, phosphorus, and metals have been shown in several field studies. While particulates can settle into swale vegetation, removal of dissolved fraction of pollutants occurs due to infiltration of water into the swales.

Ditch check filters with enhanced filter media can be installed in the swales to increase the pollution control potential of swales. As previously discussed, cost-effective enhancing agents to remove dissolved phosphorus and metals must satisfy requirements that include rapid kinetics, high capacity, and low cost among others. Due to the relatively short contact time between stormwater runoff and the media of a stormwater filter, removal mechanism kinetics must be relatively rapid if significant contaminant removal is to occur. Also, a cost-effective enhancing agent must have a large contaminant capacity, otherwise maintenance costs associated with replacing the media or restoring its capacity will be cost-prohibitive. Finally, the material cost of an enhancing agent must be low enough such that the initial cost and, if replacement of the media will be required over the life of the practice, the replacement cost of the material does not render the use of the material cost-prohibitive.

Of the materials reviewed for phosphorus removal, ferrous based materials, alumina, and water treatment residual are the potential materials for effective phosphorus removal. Ferrous based materials, alumina, zeolites (if sodium leaching and increased pH can be minimized), and some activated carbons appear to be the most promising with regards to metal removal. Organic

material such as mulch, plant biomass, and activated carbon may also be effective in metal removal but these materials may also leach phosphorus, and other organics including nitrogen into the runoff. Thus, if organic material is to be used to capture metal ions a two-stage filter may be necessary. The organic enhancing agent could be in the upstream or first stage where it would retain metal ions and possibly release nutrient ions. The second stage would contain a material that could capture and retain nutrients, including those released by the enhancing agent in the first stage.

Fly-ash and carbonate based materials that rely on metal precipitation for removal typically either require a high pH environment or raise the pH of the water to levels that would prohibit their use when treating stormwater runoff. Crustacean shells and similar shell-based enhancing agents tend to dissolve in solution and therefore would be washed away if used in a stormwater filter. Although chemical alteration can reduce their solubility it also reduces their effectiveness and may be cost-prohibitive.

Selected materials that show the most potential as enhancing agents for dissolved phosphorus and metal removal will be tested to further characterize their performance (i.e. capacity, effect on water quality parameters, etc.). Results will be used to develop, optimize and further design filters for field application.