R. Pitt

Module 4c: Soils and Amendments for Stormwater Quality Control

Abstract

The effects of urbanization on soil structure can be extensive. Infiltration of rain water through soils can be greatly reduced, plus the benefits of infiltration and biofiltration devices can be jeopardized. This paper is a compilation of results from several recent and on-going research projects that have examined some of these problems, plus possible solutions. Basic infiltration measurements in disturbed urban soils were conducted during the EPA-sponsored project by Pitt, *et al.* (1999a). The project also examined hydraulic and water quality benefits of amending these soils with organic composts. Prior EPA-funded research examined the potential of groundwater contamination by infiltrating stormwater (Pitt, *et al.* 1994, 1996, and 1999b). In addition to the information obtained during these research projects, numerous student projects have also been conduced to examine other aspects of urban soils, especially more detailed tests examining soil density and infiltration during lab-scale tests, and methods and techniques to recover infiltration capacity of urban soils. This paper is a summary of this information and it is hoped that it will prove useful to both stormwater practice designers and to modelers.

Introduction

This paper is a compilation of information from previous chapters in the Stormwater Modeling book series produced by Bill James as part of his annual series of conferences (Chapter 4 of Monograph 7, Pitt 1999, and Chapter 1, of Monograph 8, Pitt and Lantrip 2000), plus recent research. The role of urban soils in stormwater management cannot be underestimated. Although landscaped areas typically produce relatively small fractions of the annual runoff volumes (and pollutant discharges) in most areas, they need to be considered as part of most control scenarios. In stormwater quality management, the simplest approach is to attempt to maintain the relative values of the hydrologic cycle components after development compared to pre-development conditions. This usually implies the use of infiltration controls to compensate for the increased pavement and roof areas. This can be a difficult objective to meet. However, with a better understanding of urban soil characteristics, and how they may be improved, this objective can be more realistically obtained.

Whenever one talks of stormwater infiltration, questions of potential groundwater contamination arise. This paper therefore includes a short summary of our past work on investigating the potential of groundwater contamination through stormwater infiltration. This material is summarized from prior EPA-funded research, an updated book, and more recent review papers (Pitt, *et al*. 1994, 1996 and 1999b). This material shows that is possible to incorporate many stormwater infiltration options in urban areas, as long as suitable care is taken. These options should be considered in residential areas where the runoff is relatively uncontaminated and surface infiltration can typically be applied. In contrast, manufacturing industrial areas and subsurface injection should normally be excluded from stormwater infiltration consideration.

The bulk of this paper reviews our past and current investigations of the infiltration characteristics of disturbed urban soils. Several sets of tests have been conducted, both in the field and in the laboratory. We have found that typical soil compaction results in substantial reductions in infiltration rates, especially for clayey soils. Sandy soils are better able to withstand compaction, although their infiltration rates are still significantly reduced as a consequence of urbanization.

This paper also describes the results from a series of tests that have examined how the infiltrability of compacted soils can be recovered through the use of soil amendments (such as composts). Our work has shown that these soil amendments not only allow major improvements in infiltration rates, but also provide added protection to groundwater resources, especially from heavy metal contamination. Newly placed compost amendments, however, may cause increased nutrient discharges until the material is better stabilized (usually within a couple of years). Information collected during our work on stormwater filter media (Clark and Pitt 1999) has also allowed us to develop a listing of desirable traits for soil amendments and to recommend several media that may be good candidates as soil amendments.

Alternative stormwater management options are also examined using the Source Loading and Management Model (WinSLAMM) and this soil information. The use of biofiltration controls, such as roof gardens for example, can result in almost complete removal of roof runoff from the surface runoff component.

To put into perspective the recent infiltration tests, the following paragraphs briefly review the previous work (as reported in Monographs 7 and 8 of this series; Pitt 1999 and Pitt and Lantrip 2000).

Physical Processes of Infiltration

Infiltration of rainfall into pervious surfaces is controlled by three mechanisms, the maximum possible rate of entry of the water through the soil/plant surface, the rate of movement of the water through the vadose (unsaturated) zone, and the rate of drainage from the vadose zone into the saturated zone. During periods of rainfall excess, long-term infiltration is the least of these three rates, and the runoff rate after depression storage is filled is the excess of the rainfall intensity above the infiltration rate. The infiltration rate typically decreases during periods of rainfall excess. Storage capacity within the soil profile is recovered during periods when the drainage from the vadose zone exceeds the infiltration rate.

The surface entry rate of water may be affected by the presence of a thin layer of silts and clay particles at the surface of the soil and vegetation. These particles may cause a surface seal that would decrease a normally high infiltration rate. The movement of water through the soil depends on the characteristics of the underlying soil. Once the surface soil layer is saturated, water cannot enter soil faster than it is being draining into the vadose zone, so this transmission rate affects the infiltration rate during longer events. The depletion of available storage capacity in the soil due to urbanization-associated compaction affects the transmission and drainage rates. The storage capacity of soils depends on the soil thickness, porosity, and the soil-water content. Many factors including, soil texture, root development, soil insect and animal bore holes, structure, and presence of organic matter, affect the effective porosity of the soil.

The infiltration of water into the surface soil is responsible for the largest abstraction (loss) of rainwater in natural areas. The infiltration capacity of most soils allows low intensity rainfall to totally infiltrate, unless the soil voids became saturated or the underlain soil is more compact than the top layer (Morel-Seytoux 1978). High intensity rainfalls generate substantial runoff because the infiltration capacity at the upper soil surface is surpassed, even though the underlain soil might still be very dry.

The classical assumption is that the infiltration capacity of a soil is highest at the very beginning of a storm and decreases with time (Willeke 1966). The soil-water content of the soil, whether it was initially dry or wet from a recent storm, will have a great effect on the infiltration capacity of certain soils (Morel-Seytoux 1978). Horton (1939) is credited with defining infiltration capacity and deriving an appropriate working equation. Horton defined infiltration capacity as "...the maximum rate at which water can enter the soil at a particular point under a given set of conditions" (Morel-Seytoux 1978).

Natural infiltration is significantly reduced in urban areas due to numerous factors: the decreased area of exposed soils, removal of surface soils and exposing subsurface soils, grading of soils through landscaping, and compaction of the soils during earth moving and construction operations. The decreased areas of soils are typically associated with increased runoff volumes and peak flow rates, while the effects of soil disturbance are rarely considered. Infiltration practices have long been applied in many areas to compensate for the decreased natural infiltration areas, but with limited success. Silting of the infiltration areas is usually responsible for early failures of these intended infiltration controls, although compaction from heavy traffic is also a recognized problem. More recently, "biofiltration" practices, that rely more on surface infiltration in extensively vegetated areas, are gaining in popularity and appear to be a more robust solution than conventional infiltration trenches. These biofiltration devices also allow modifications of the soil with amendments.

Groundwater Impacts Associated with Stormwater Infiltration

One of the major concerns of stormwater infiltration is the question of adversely impacting groundwater quality. Pitt, *et al*. (1994, 1996 and 1999b) reviewed many studies that investigated groundwater contamination from stormwater infiltration. They developed a methodology to evaluate the contamination potential of stormwater nutrients, pesticides, other organic compounds, pathogens, metals, salts and other dissolved minerals, suspended solids, and gases, based on the concentrations of the contaminant in stormwater, the treatability of the contaminant, and the mobility of the contaminant through the vadose zone. Stormwater salts, some pathogens, 1,3-dichlorobenzene, pyrene, fluoranthene, and zinc, were found to have high potentials for contaminating groundwater, under some conditions. However, there is only a minimal potential of contaminating groundwaters from residential area stormwaters (chlorides in northern areas remains a concern), especially if surface infiltration is used compared to subsurface disposal (such as by deep trenches or injection wells).

Prior to urbanization, groundwater recharge resulted from infiltration of rain and snowmelt through pervious surfaces, including grasslands and woods. This infiltrating water was relatively uncontaminated. With urbanization in humid areas, the permeable soil surface area through which recharge by infiltration could occur was reduced. This resulted in much less groundwater recharge and greatly increased surface runoff and reduced dry weather flows. In addition, the waters available for recharge generally carried increased quantities of pollutants. With urbanization, new sources of groundwater recharge also occurred, including recharge from domestic septic tanks, percolation basins, industrial waste injection wells, and from residential irrigation. In arid areas, groundwater recharge may actually increase with urbanization due to irrigation, resulting in increased dry weather base flows in urban streams.

Relative Risks Associated with Stormwater Infiltration of Various Contaminants

The following summary, from Pitt, *et al*. (1994, 1996, and 1999b), describe the stormwater pollutants which have the greatest potential of adversely affecting groundwater quality during stormwater infiltration. These prior publications contain several hundred references pertaining to the groundwater contamination potential of stormwater, and although they are too numerous to repeat here, they were of great help in preparing this synopsis.

Table 1 is a summary of the pollutants found in stormwater that may cause groundwater contamination problems for various reasons. This table does not consider the risk associated with using groundwater contaminated with these pollutants. Causes of concern include high mobility (low sorption potential) in the vadose zone, high abundance (high concentrations and high detection frequencies) in stormwater, and high soluble fractions (small fractions associated with particulates would have little removal potential using conventional stormwater sedimentation controls) in the stormwater. The contamination potential is the lowest rating of the influencing factors. As an example, if no pretreatment was used before percolation through surface soils, the mobility and abundance criteria are most important. If a compound was mobile, but was in low abundance (such as for Volatile Organic Compounds), then the groundwater contamination potential would be low. However, if the compound was mobile and was also in high abundance (such as for sodium chloride, in certain conditions), then the groundwater contamination would be high. If sedimentation pretreatment was to be used before infiltration, then most of the particulate-bound pollutants will likely be removed before infiltration. In this case, all three influencing factors (mobility, abundance in stormwater, and soluble fraction) would be considered important. As an example, chlordane would have a low contamination potential with sedimentation pretreatment, while it would have a moderate contamination potential if no pretreatment was used. In addition, if subsurface infiltration/injection was used instead of surface percolation, the compounds would most likely be more mobile, making the abundance criteria the most important, with some regard given to the filterable fraction information for operational considerations.

Table 1. Groundwater Contamination Potential for Stormwater Pollutants (Source: Pitt, *et al.* 1996)

	Compounds	Mobility (sandy/low organic soils)	Abundance in storm-water	Fraction filterable	Contamination potential for surface infilt. and no pretreatment	Contamination potential for surface infilt. with sediment-ation	Contamination potential for sub-surface inj. with minimal pretreatment
Nutrients	nitrates	mobile	low/moderate	high	low/moderate	low/moderate	low/moderate
Pesticides	2,4-D	mobile	low	likely low	low	low	low
	γ-BHC (lindane)	intermediate	moderate	likely low	moderate	low	moderate
	malathion	mobile	low	likely low	low	low	low
	atrazine	mobile	low	likely low	low	low	low
	chlordane	intermediate	moderate	very low	moderate	low	moderate
	diazinon	mobile	low	likely low	low	low	low
Other organics	VOCs	mobile	low	very high	low	low	low
	1,3-dichloro-benzene	low	high	high	low	low	high
	anthracene	intermediate	low	moderate	low	low	low
	benzo(a) anthracene	intermediate	moderate	very low	moderate	low	moderate
	bis (2-ethylhexyl) phthalate	intermediate	moderate	likely low	moderate	low?	moderate
	butyl benzyl phthalate	low	low/moderate	moderate	low	low	low/moderate
	fluoranthene	intermediate	high	high	moderate	moderate	high
	fluorene	intermediate	low	likely low	low	low	low
	naphthalene	low/inter.	low	moderate	low	low	low
	penta-chlorophenol	intermediate	moderate	likely low	moderate	low?	moderate
	phenanthrene	intermediate	moderate	very low	moderate	low	moderate
	pyrene	intermediate	high	high	moderate	moderate	high
Pathogens	enteroviruses	mobile	likely present	high	high	high	high
	Shigella	low/inter.	likely present	moderate	low/moderate	low/moderate	high
	Pseudomonas aeruginosa	low/inter.	very high	moderate	low/moderate	low/moderate	high
	protozoa	low/inter.	likely present	moderate	low/moderate	low/moderate	high
Heavy metals	nickel	low	high	low	low	low	high
	cadmium	low	low	moderate	low	low	low
	chromium	inter./very low	moderate	very low	low/moderate	low	moderate
	lead	very low	moderate	very low	low	low	moderate
	zinc	low/very low	high	high	low	low	high
Salts	chloride	mobile	seasonally high	high	high	high	high

This table is only appropriate for initial estimates of contamination potential because of the simplifying assumptions made, such as the likely worst case mobility measures for sandy soils having low organic content. If the soil was clayey and/or had a high organic content, then most of the organic compounds, because of their retardation characteristics, would be less mobile than shown on this table. The abundance and filterable fraction information is generally applicable for warm weather stormwater runoff at residential and commercial area outfalls. The concentrations and detection frequencies (and corresponding contamination potentials) would likely be greater for critical source areas (especially vehicle service areas) and critical land uses (especially manufacturing industrial areas).

With biofiltration through amended urban soils, the lowered groundwater contamination potential shown for surface infiltration with prior treatment, would generally apply. With gravel-filled infiltration trenches having no grass filtering or other pre-treatment, or with discharge in disposal wells, the greater groundwater contamination potentials shown for injection with minimal pretreatment would generally apply.

The stormwater pollutants of most concern (those that may have the greatest adverse impacts on groundwaters) include:

- nutrients: nitrate has a low to moderate groundwater contamination potential for both surface percolation and subsurface infiltration/injection practices because of its relatively low concentrations found in most stormwaters. However, if the stormwater nitrate concentration was high, then the groundwater contamination potential would also likely be high.

- pesticides: lindane and chlordane have moderate groundwater contamination potentials for surface percolation practices (with no pretreatment) and for subsurface injection (with minimal pretreatment). The groundwater contamination potentials for both of these compounds would likely be substantially reduced with adequate sedimentation pretreatment. Pesticides have been mostly found in urban runoff from residential areas, especially in dry-weather flows associated with landscaping irrigation runoff.

- other organics: 1,3-dichlorobenzene (a common polycyclic aromatic hydrocarbon found in stormwater, originating from fossil fuel combustion) may have a high groundwater contamination potential for subsurface infiltration/injection (with minimal pretreatment). However, it would likely have a lower groundwater contamination potential for most surface percolation practices because of its relatively strong sorption to vadose zone soils. Both pyrene and fluoranthene would also likely have high groundwater contamination potentials for subsurface infiltration/injection practices, but lower contamination potentials for surface percolation practices because of their more limited mobility through the unsaturated zone (vadose zone). Others (including benzo(a)anthracene, bis (2-ethylhexyl) phthalate, pentachlorophenol, and phenanthrene) may also have moderate groundwater contamination potentials, if surface percolation with no pretreatment, or subsurface injection/infiltration is used. These compounds would have low groundwater contamination potentials if surface infiltration was used with sedimentation pretreatment. Volatile organic compounds (VOCs) may also have high groundwater contamination potentials if present in the stormwater (likely for some industrial and commercial facilities and vehicle service establishments). The other organics, especially the volatiles, are mostly found in industrial areas. The phthalates are found in all areas. The PAHs are also found in runoff from all areas, but they are in higher concentrations and occur more frequently in industrial areas.

- pathogens: enteroviruses likely have a high groundwater contamination potential for all percolation practices and subsurface infiltration/injection practices, depending on their presence in stormwater (likely if contaminated with sanitary sewage). Other pathogens, including *Shigella*, *Pseudomonas aeruginosa*, and various protozoa, would also have high groundwater contamination potentials if subsurface infiltration/injection practices are used without disinfection. If disinfection (especially by chlorine or ozone) is used, then disinfection byproducts (such as trihalomethanes or ozonated bromides) would have high groundwater contamination potentials. Pathogens are most likely associated with sanitary sewage contamination of storm drainage systems, but several bacterial pathogens are commonly found in surface runoff in residential areas.

- heavy metals: nickel and zinc would likely have high groundwater contamination potentials if subsurface infiltration/injection was used. Chromium and lead would have moderate groundwater contamination potentials for subsurface infiltration/injection practices. All metals would likely have low groundwater contamination potentials if surface infiltration was used with sedimentation pretreatment. Zinc is mostly found in roof runoff and other areas where galvanized metal comes into contact with rainwater.

- salts: chloride would likely have a high groundwater contamination potential in northern areas where road salts are used for traffic safety, irrespective of the pretreatment, infiltration or percolation practice used. Salts are at their greatest concentrations in snowmelt and early spring runoff in northern areas.

Prior Field Measurements of Infiltration in Disturbed Urban Soils

Early unpublished double-ring infiltration tests were conducted by the Wisconsin DNR in Oconomowoc, WI, as part of their Milwaukee River Priority Watershed Plan. These data, as shown in Table 2, indicated highly variable infiltration rates for soils that were generally sandy (NRCS A and B hydrologic group soils) and dry. The median initial rate was about 75 mm/h (3 in/h), but ranged from 0 to 600 mm/h (0 to 25 in/h). The final rates also had a median value of about 75 mm/h (3 in/h) after at least two hours of testing, but ranged from 0 to 400 mm/h (0 to 15 in/h). Many infiltration rates actually increased with time during these tests. In about 1/3 of the cases, the observed infiltration rates remained very close to zero, even for these sandy soils. Areas that experienced substantial disturbances or traffic (such as school playing fields), and siltation (such as in some grass swales) had the lowest infiltration rates.

Table 2. Ranked Oconomowoc Double Ring Infiltration Test Results (dry conditions)

Initial Rate (in/h)	Final Rate (after 2 hours) (in/h)	Total Range of Observed Rates (in/h)
25	15	11 to 25
22	17	17 to 24
14.7	9.4	9.4 to 17
5.8	9.4	0.2 to 9.4
5.7	9.4	5.1 to 9.6
4.7	3.6	3.1 to 6.3
4.1	6.8	2.9 to 6.8
3.1	3.3	2.4 to 3.8
2.6	2.5	1.6 to 2.6
0.3	0.1	<0.1 to 0.3
0.3	1.7	0.3 to 3.2
0.2	<0.1	<0.1 to 0.2
<0.1	0.6	<0.1 to 0.6
<0.1	<0.1	all <0.1
<0.1	<0.1	all <0.1
<0.1	<0.1	all <0.1

Source: unpublished data from the WI Dept. of Natural Resources

More recently, a series of 153 double ring infiltrometer tests were conducted in disturbed urban soils in the Birmingham, and Mobile, Alabama, areas (Pitt, *et al*. 1999a). The tests were organized in a complete 2^3 factorial design (Box, *et al*. 1978) to examine the effects of soil-water, soil texture, and soil density (compaction) on water infiltration through historically disturbed urban soils. Ten sites were selected representing a variety of desired conditions (compaction and texture) and numerous tests were conducted at each test site area. Soil-water content and soil texture conditions were determined by standard laboratory soil analyses. Compaction was measured in the field using a cone penetrometer and confirmed by the site history. From 12 to 27 replicate tests were conducted in each of the eight experimental categories in order to measure the variations within each category for comparison to the variation between the categories:

Category	Soil Texture	Compaction	Initial Soil-Water Content	Number of Tests
1	Sand	Compact	Saturated	18
2	Sand	Compact	Dry	21
3	Sand	Non-compact	Saturated	24
4	Sand	Non-compact	Dry	12
5	Clay	Compact	Saturated	18
6	Clay	Compact	Dry	15
7	Clay	Non-compact	Saturated	27
8	Clay	Non-compact	Dry	18

Soil infiltration capacity was expected to be related to the time since the soil was disturbed by construction or grading operations (turf age). In most new developments, compacted soils are expected to be dominant, with reduced infiltration compared to pre-construction conditions. In older areas, the soil may have recovered some of its infiltration capacity due to root structure development and from soil insects and other digging animals. Soils having a variety of times since development, ranging from current developments to those about 50 years old, were included in the sampling program. These test sites did not adequately represent a wide range of age conditions for each test condition, so the effects of age could not be directly determined. The WI Dept. of Natural Resources and the University of Wisconsin (Roger Bannerman, WI DNR, personal communication) have conducted some soil infiltration tests on loamy soils to examine the effects of age of urbanization on soil infiltration rates. Their preliminary tests have indicated that as long as several decades may be necessary before compacted loam soils recover to conditions similar to pre-development conditions.

Three TURF-TEC Infiltrometers were used within a meter from each other to indicate the infiltration rate variability of soils in close proximity. These devices have an inner ring about 64 mm (2.5 in.) in diameter and an outer ring about 110 mm (4.25 in.) in diameter. The water depth in the inner compartment starts at 125 mm (5 in.) at the beginning of the test, and the device is pushed into the ground 50 mm (2 in.). Both the inner and outer compartments were filled with clean water by first filling the inner compartment and allowing it to overflow into the outer compartment. Readings were taken every five minutes for a duration of two hours. The incremental infiltration rates were calculated by noting the drop of water level in the inner compartment over each five minute time period.

The weather occurring during this testing phase enabled most site locations to produce a paired set of dry and wet tests. The dry tests were taken during periods of little rain, which typically extended for as long as two weeks with sunny, hot days. The saturated tests were conducted after through soaking of the ground by natural rain or by irrigation. The soil-water content was measured in the field using a portable soil water meter and in the laboratory using standard soil-water content methods. Saturated conditions occurred for most soils when the soil-water content exceeded about 20% by weight.

The texture of the samples were determined by ASTM standard sieve analyses (ASTM D 422 –63 (*Standard Test Method For Particle Size Analysis of Soils*). "Clayey" soils had 30 to 98% clay, 2 to 45% silt, and 2 to 45% sand. This category included clay and clay loam soils. "Sandy" soils had 65 to 95% sand, 2 to 25% silt, and 5 to 35% clay. This category included sand, loamy sand, and sandy loam soils. No natural soils were tested that were predominately silt or loam.

The soil compaction at each site was measured using a cone penetrometer (DICKEY-john Soil Compaction Tester Penetrometer). Penetrometer measurements are sensitive to water content. Therefore, these measurements were not made for saturated conditions and the degree of soil compaction was also determined based on the history of the specific site (especially the presence of parked vehicles, unpaved vehicle lanes, well-used walkways, etc.). Compact soils were defined as having a reading of greater than 300 psi at a depth of three inches.

Other factors that were beyond the control of the experiments, but also affect infiltration rates, include bioturbation by ants, gophers and other small burrowing animals, worms, and plant roots.

Figures 1 and 2 are 3D plots of the field infiltration data, illustrating the effects of soil-water content and compaction, for both sands and clays. Four general conditions were observed to be statistically unique, as listed on Table 3. Compaction has the greatest effect on infiltration rates in sandy soils, with little detrimental effects associated with higher soil-water content conditions. Clay soils, however, are affected by both compaction and soil-water content. Compaction was seen to have about the same effect as saturation on clayey soils, with saturated and compacted clayey soils having very little effective infiltration.

Figure 1. Three dimensional plot of infiltration rates for sandy soil conditions.

Figure 2. Three dimensional plot of infiltration rates for clayey soil conditions.

Table 3. Infiltration Rates for Significant Groupings of Soil Texture, Soil-Water Content, and Compaction Conditions

Group	Number of tests	Average infiltration rate (in/h) (total 2 h test durations)	COV
noncompacted sandy soils	36	13.5	0.4
compact sandy soils	39	1.5	1.3
noncompacted and dry clayey soils	18	9.3	1.5
all other clayey soils (compacted and dry, plus all wetter conditions)	60	0.2	2.4

The Horton infiltration equation was fitted to each set of individual site test data and the equation coefficients were statistically compared for the different site conditions. Because of the wide range in observed rates for each of the major categories, it may not matter which infiltration rate equation is used. The residuals are all relatively large and it is much more important to consider the random nature of infiltration about any fitted model and to address the considerable effect that soil compaction has on infiltration. It may therefore be best to use a Monte Carlo stochastic component in a runoff model to describe these variations for disturbed urban soils.

As one example of an approach, Table 4 shows the measured infiltration rates for each of the four major soil categories, separated into several time increments. This table shows the observed infiltration rates for each test averaged for different storm durations (15, 30, 60, and 120 minutes). Also shown are the ranges and COV values for each duration and condition. Therefore, a routine in a model could select an infiltration rate, associated with the appropriate soil category, based on the storm duration. The selection would be from a random distribution (likely a log-normal distribution) as described from this table.

Table 4. Soil Infiltration Rates for Different Categories and Storm Durations

	Sand, Non-compacted			
	15 minutes	30 minutes	60minutes	120 minutes
mean	19.5	17.4	15.2	13.5
median	18.8	16.5	16.5	15.4
std. dev.	8.8	8.1	6.7	6.0
min	1.5	0.0	0.0	0.0
max	38.3	33.8	27.0	24.0
COV	0.4	0.5	0.4	0.4
number	36	36	36	36

	Sand, Compacted			
	15 minutes	30 minutes	60minutes	120 minutes
mean	3.6	2.2	1.6	1.5
median	2.3	1.5	0.8	0.8
std. dev.	6.0	3.6	2.0	1.9
min	0.0	0.0	0.0	0.0
max	33.8	20.4	9.0	6.8
COV	1.7	1.6	1.3	1.3
number	39	39	39	39

	Clay, Dry Non-compacted			
	15 minutes	30 minutes	60minutes	120 minutes
mean	9.0	8.8	10.8	9.3
median	5.6	4.9	4.5	3.0
std. dev.	9.7	8.8	15.1	15.0
min	0.0	0.0	0.0	0.0
max	28.5	26.3	60.0	52.5
COV	1.1	1.0	1.4	1.6
number	18	18	18	18

	All other clayey soils (compacted and dry, plus all saturated conditions)			
	15 minutes	30 minutes	60minutes	120 minutes
mean	1.3	0.7	0.5	0.2
median	0.8	0.8	0.0	0.0
std. dev.	1.6	1.4	1.2	0.4
min	0.0	0.0	0.0	0.0
max	9.0	9.8	9.0	2.3
COV	1.2	1.9	2.5	2.4
number	60	60	60	60

Figures 3 through 6 are probability plots showing the observed infiltration rates for each of the four major soil categories, separated by these event durations. Each figure has four separate plots representing the storm event averaged infiltration rates corresponding to four storm durations from 15 minutes to 2 hours. As indicated previously, the infiltration rates became relatively steady after about 30 to 45 minutes during most tests. Therefore, the 2 hour averaged rates could likely be used for most events of longer duration. There is an obvious pattern on these plots which show higher rates for shorter rain durations, as expected. The probability distributions are closer to being log-normally distributed than normally distributed. However, with the large number of zero infiltration rate observations for three of the test categories, log-normal probability plots were not possible.

Figure 3. Probability plots for infiltration measurements for noncompacted, sandy soil, conditions.

Figure 4. Probability plots for infiltration measurements for compacted, sandy soil, conditions.

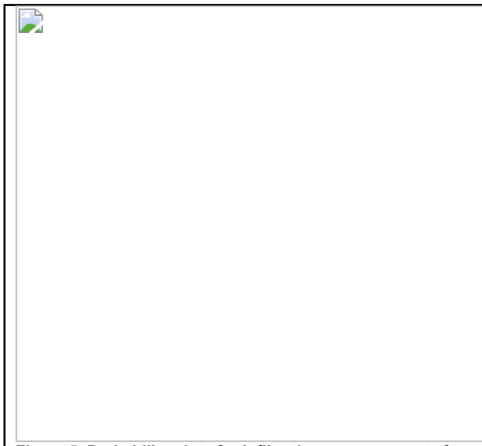

Figure 5. Probability plots for infiltration measurements for dry-noncompacted, clayey soil, conditions.

Figure 6. Probability plots for infiltration measurements for wet-noncompacted, dry-compacted, and wet-compacted, clayey soil conditions.

The soil texture and compaction classification would remain fixed for an extended simulation period (unless the soils underwent an unlikely recovery operation to reduce the soil compaction), but the clayey soils would be affected by the antecedent interevent period which would define the soil-water level at the beginning of the event. Recovery periods are highly dependent on site-specific soil and climatic conditions and are calculated using various methods in continuous simulation urban runoff models. The models assume that the recovery period is much longer than the period needed to produce saturation conditions. As noted above, saturation (defined here as when the infiltration rate reaches a constant value) occurred under an hour during these tests. A simple estimate of the time needed for recovery of soil-water levels is given by the USDA’s Natural Resources Conservation Service (NRCS) (previously the Soil Conservation Service, SCS) in TR-55 (McCuen 1998). The NRCS developed three antecedent soil-water conditions as follows:

- Condition I: soils are dry but not to the wilting point
- Condition II: average conditions
- Condition III: heavy rainfall, or lighter rainfall and low temperatures, have occurred within the last five days, producing saturated soil.

McCuen (1998) presents Table 5 (from the NRCS) that gives seasonal rainfall limits for these three conditions. Therefore, as a rough guide, saturated soil conditions for clay soils may be assumed if the preceding 5-day total rainfall was greater than about 25 mm (one inch) during the winter or greater than about 50 mm (two inches) during the summer. Otherwise, the “other” infiltration conditions for clay should be assumed.

Table 5. Total Five-Day Antecedent Rainfall for Different Soil-Water Content Conditions (in.)

	Dormant Season	Growing Season
Condition I	<0.5	<1.4
Condition II	0.5 to 1.1	1.4 – 2.1
Condition III	>1.1	> 2.1

Recent Laboratory Controlled Compaction and Infiltration Tests

Laboratory Test Methods

Previous research (Pitt, *et al.* 1999a), as summarized above, has identified significant reductions in infiltration rates in disturbed urban soils. The tests reported in the following discussion were conducted under more controlled laboratory conditions and represent a wider range of soil textures and known soil density values compared to the previous field tests.

Laboratory permeability test setups were used to measure infiltration rates associated with different soils having different textures and compactions. These tests differed from normal permeability tests in that high resolution observations were made at the beginning of the tests to observe the initial infiltration behavior. The tests were run for up to 20 days, although most were completed (when steady low rates were observed) within 3 or 4 days.

Test samples were prepared by mixing known quantities of sand, silt, and clay to correspond to defined soil textures, as shown in Table 6. The initial sample moistures were determined and water was added to bring the initial soil water to about 8% by weight, per standard procedures (ASTM D1140-54), reflecting typical “dry” soil conditions and to allow water movement through the soil columns. Table 7 lists the actual soil water levels at the beginning of the tests, along with the actual dry bulk soil densities and indications of root growth problems.

Table 6. Test Mixtures During Laboratory Tests

	Pure Sand	Pure Clay	Pure Silt	Sandy Loam	Clayey Loam	Silt Loam	Clay Mix
% Sand	100			72.1	30.1	19.4	30
% Clay		100		9.2	30.0	9.7	50
% Silt			100	18.7	39.9	70.9	20

Table 7. Soil Water and Density Values during Laboratory Tests

			Root Growth Potential Problems (NRCS 2001)				
Soil Types	Compaction Method	Dry Bulk Density Before Test (g/cm^3)	Ideal Bulk Density	Bulk Densities that may Affect Root Growth	Bulk Densities that Restrict Root Growth	Before Test Water Content (%), by weight	After Test Water Content (%), by weight
Silt	Hand	1.508		X		9.7	22.9
	Standard	1.680		X		8.4	17.9
	Modified	1.740			X	7.8	23.9
Sand	Hand	1.451	X			5.4	21.6
	Standard	1.494	X			4.7	16.4
	Modified	1.620		X		2.0	16.1
Clay	Hand	1.242		X		10.6	N/A
Sandy Loam	Hand	1.595		X		7.6	20.2
	Standard	1.653		X		7.6	18.9
	Modified	1.992			X	7.6	9.9
Silt Loam	Hand	1.504		X		8.1	23.0
	Standard	1.593		X		8.1	27.8
	Modified	1.690		X		8.1	27.8
Clay Loam	Hand	1.502		X		9.1	24.1
	Standard	1.703			X	9.1	19.0
	Modified	1.911			X	9.1	14.5
Clay Mix	Hand	1.399		X		8.2	42.2
	Standard	1.685			X	8.2	N/A
	Modified	1.929			X	8.2	N/A

Three methods were used to modify the compaction of the soil samples: hand compaction, Standard Proctor Compaction, and Modified Proctor Compaction. Both Standard and Modified Proctor Compactions follow ASTM standard (D 1140-54). All tests were conducted using the same steel molds (115.5 mm tall with 105 mm inner diameter, having a volume of 1000 cm^3). The Standard Proctor compaction hammer is 24.4 kN and has a drop height of 300 mm. The Modified Proctor hammer is 44.5 kN and has a drop height of 460 mm. For the Standard Proctor setup, the hammer was dropped on the test soil in the mold 25 times on each of three soil layers, while for the Modified Proctor test, the heavier hammer was also dropped 25 times, but on each of five soil layers. The Modified Proctor test therefore resulted in much more compacted soil. The hand compaction was done by gentle hand pressing to force the soil into the mold with as little compaction as possible. A minimal compaction effort was needed to keep the soil in contact with the mold walls and to prevent short-circuiting during the tests. The hand compacted soil specimens therefore had the least amount of compaction. The head for these permeability tests was 1.14 meter (top of the water surface to the top of the compaction mold). The water temperature during the test was kept consistent at 75^{o}F.

As shown on Table 7, a total of 7 soil types were tested representing all main areas of the standard soil texture triangle. Three levels of compaction were tested for each soil, resulting in a total of 21 tests. However, only 15 tests resulted in observed infiltration. The Standard and Modified Proctor clay tests, the Modified Proctor clay loam, and all of the clay mixture tests did not result in any observed infiltration after several days and those tests were therefore stopped. The “after test” water contents generally corresponded to the “saturated soil” conditions of the earlier field measurements.

Also shown on Table 7 are indications of root growth problems for these soil densities, based on the NRCS Soil Quality Institute 2000 report, as summarized by the Ocean County Soil Conservation District (NRCS 2001). The only soil test mixtures that were in the “ideal” range for plant growth were the hand placed and standard compacted sands. Most of the modified compacted test mixtures were in the range that are expected to restrict root growth, the exceptions were the sand and silt loam mixtures. The rest of the samples were in the range that may affect root growth. These tests cover a wide range of conditions that may be expected in urban areas.

Laboratory Test Results

Figures 7 through 11 show the infiltration plots obtained during these laboratory compaction tests. Since the hydraulic heads for these experiments was a little more than 1 m, the values obtained would not be very applicable to typical rainfall infiltration values. However, they may be comparable to biofiltration or other infiltration devices that have substantial head during operation. The final percolation values may be indicative of long-term infiltration rates, and these results do illustrate the dramatic effects of soil compaction and texture on the infiltration rates.

Figure 7. Sandy soil laboratory infiltration test results.

Figure 8. Sandy loam soil laboratory infiltration test results.

Figure 9. Silty soil laboratory infiltration test results.

Figure 10. Silty loam soil laboratory infiltration test results.

Figure 11. Clayey loam soil laboratory infiltration test results.

Another series of controlled laboratory tests were conducted to better simulate field conditions and standard double-ring infiltration tests, as shown in Table 8. Six soil samples were tested, each at the three different compaction levels described previously. The same permeability test cylinders were used as in the above tests, but plastic extensions were used to enable small depths of standing water on top of the soil test mixtures (4.3 inches, or 11.4 cm, maximum head). Most of these tests were completed within 3 hours, but some were continued for more than 150 hours. Only one to three observation intervals were used during these tests, so they did not have sufficient resolution or enough data points to attempt to fit to standard infiltration equations. However, as noted previously, these longer-term averaged values may be more suitable for infiltration rate predictions due to the high natural variability observed during the initial field tests. As shown, there was very little variation between the different time periods for these tests, compared to the differences between the compaction or texture groupings. Also, sandy soils can still provide substantial infiltration capacities, even when compacted greatly, in contrast to the soils having clays that are very susceptible to compaction.

Table 8. Low-Head Laboratory Infiltration Tests for Various Soil Textures and Densities (densities and observed infiltration rates)

	Hand Compaction	Standard Compaction	Modified Compaction
Sand (100% sand)	Density: 1.36 g/cm^3 (ideal for roots)	Density: 1.71 g/cm^3 (may affect roots)	Density: 1.70 g/cm^3 (may affect roots)
	0 to 0.48 hrs: 9.35 in/h	0 to 1.33 hrs: 3.37 in/h	0 to 0.90 hrs: 4.98 in/h
	0.48 to 1.05 hrs: 7.87 in/h	1.33 to 2.71 hrs: 3.26 in/h	0.90 to 1.83 hrs: 4.86 in/h
	1.05 to 1.58 hrs: 8.46 in/h		1.83 to 2.7 hrs: 5.16 in/h

Silt (100% silt)	Density: 1.36 g/cm^3 (close to ideal for roots) 0 to 8.33 hrs: 0.26 in/h 8.33 to 17.78 hrs: 0.24 in/h 17.78 to 35.08 hrs: 0.25 in/h	Density: 1.52 g/cm^3 (may affect roots) 0 to 24.22 hrs: 0.015 in/h 24.22 to 48.09: 0.015 in/h	Density: 1.75 g/cm^3 (will likely restrict roots) 0 to 24.20 hrs: 0.0098 in/h 24.20 to 48.07: 0.0099 in/h
Clay (100% clay)	Density: 1.45 g/cm^3 (may affect roots) 0 to 22.58 hrs: 0.019 in/h 22.58 to 47.51 hrs: 0.016 in/h	Density: 1.62 g/cm^3 (will likely restrict roots) 0 to 100 hrs: <2X10-3 in/h	Density: 1.88 g/cm^3 (will likely restrict roots) 0 to 100 hrs: <2X10-3 in/h
Sandy Loam (70% sand, 20% silt, 10% clay)	Density: 1.44 g/cm^3 (close to ideal for roots) 0 to 1.17 hrs: 1.08 in/h 1.17 to 4.37 hrs: 1.40 in/h 4.37 to 7.45 hrs: 1.45 in/h	Density: 1.88 g/cm^3 (will likely restrict roots) 0 to 3.82 hrs: 0.41 in/h 3.82 to 24.32 hrs: 0.22 in/h	Density: 2.04 g/cm^3 (will likely restrict roots) 0 to 23.50 hrs: 0.013 in/h 23.50 to 175.05 hrs: 0.011 in/h
Silty Loam (70% silt, 20% sand, 10% clay)	Density: 1.40 g/cm^3 (may affect roots) 0 to 7.22 hrs: 0.17 in/h 7.22 to 24.82 hrs: 0.12 in/h 24.82 to 47.09 hrs: 0.11 in/h	Density: 1.64 g/cm^3 (will likely restrict roots) 0 to 24.62 hrs: 0.014 in/h 24.62 to 143.52 hrs: 0.0046 in/h	Density: 1.98 g/cm^3 (will likely restrict roots) 0 to 24.62 hrs: 0.013 in/h 24.62 to 143.52 hrs: 0.0030 in/h
Clay Loam (40% silt, 30% sand, 30% clay)	Density: 1.48 g/cm^3 (may affect roots) 0 to 2.33 hrs: 0.61 in/h 2.33 to 6.13 hrs: 0.39 in/h	Density: 1.66 g/cm^3 (will likely restrict roots) 0 to 20.83 hrs: 0.016 in/h 20.83 to 92.83 hrs: 0.0066 in/h	Density: 1.95 g/cm^3 (will likely restrict roots) 0 to 20.83 hrs: <0.0095 in/h 20.83 to 92.83 hrs: 0.0038 in/h

Soil Amendments to Improve Urban Soil Performance

A growing area of research is the investigation of the use of soil amendments to improve the infiltration performance of urban soils, and to provide additional protection against groundwater contamination.

Soil Modifications to Enhance Infiltration

Turf scientists have been designing turf areas with rapid infiltration capabilities for playing fields for many years. It is thought that some of these design approaches could be used in other typical urban areas to enhance infiltration and reduce surface runoff. Several golf course and athletic field test sites were examined in Alabama during our study to document how turf areas can be constructed to enhance infiltration (Pitt, *et al*. 1999a). These areas were designed to rapidly dry-off following a rain to minimize downtime due to excessive soil-water levels. Turf construction techniques were reviewed at three sites: an intramural playing field at the University of Alabama at Birmingham (UAB), the UAB practice football field, and a local golf course. The UAB intramural field has a simple drainage design of parallel 100 mm (4in.) wide trenches with a filter fabric wrapped pipe laid 30 cm (12 in.) deep. A thick sand backfill was used and then the area was recapped with sod. The drainage pipe was directed to the storm drainage system. The drainage for the UAB practice field was done by a local engineering firm that chose a fishbone drainage design. A trunk line of 100 mm (4 in.) corrugated pipe is the “spine” of the system with smaller 75 mm (3 in.) pipes stemming off from the main line. All the pipes rest on a gravel base with a sand backfill. This system feeds to a larger basin that collects the stormwater and takes it to the existing storm drainage system. The golf course used the same basic fishbone design noted above, but differed in the sizes of the individual pipes. The drainpipes are 3 m (10 ft.) apart in trenches filled with 75 mm (3 in.) of gravel. The pipes are then covered with 30 cm (12 in.) of sand with the top 50 mm (2 in.) of the sand consisting of a blend of sand and peat moss. This particular mixture is known as the USGA greens sand mix and is readily available because of its popularity in golf course drainage design. If the backfill sand particles are too large, clay is added to the mixture to slow the drainage. However, if the sand particles are too small, the soil will compact too tightly and will not give the desired results. In all of these cases, standing water is rare after rain has stopped, even considering the generally flat playing fields and very high rainfall intensities occurring in the Birmingham area.

Other modifications include amending the soil with other materials. The following discussion summarizes the results of tests of amended soils and the effects on infiltration and groundwater protection.

Water Quality and Quantity Effects of Amending Soils with Compost

Another component of the EPA-sponsored project that included the field infiltration tests was conducted by the College of Forestry Resources at the University of Washington (under the direction of Dr. Rob Harrison) in the Seattle area to measure the benefits of amending urban soils with compost (Pitt, *et al.* 1999a). It was found that compost-amended soils could improve the infiltration characteristics of these soils, along with providing some filtration/sorption benefits to capture stormwater pollutants before they enter the groundwater.

Existing facilities at the University of Washington’s (UW) Center for Urban Horticulture were used for some of the test plot examinations of amended soils. Two additional field sites were also developed, one at Timbercrest High School and one at WoodMoor High School in Northern King County, Washington. Both of these sites are located on poorly-sorted, compacted glacial till soils of the Alderwood soil series. Large plywood bays were used for containing soil and soil-compost mixes.

At the UW test facilities, two different Alderwood glacial till soils were mixed with compost. Two plots each of glacial till-only soil and 2:1 mixtures of soil:compost were studied. The soil-compost mixture rates were also the same for the Timbercrest and Woodmoor sites, using

Cedar Grove compost. The two composts used at the UW sites were Cedar Grove and GroCo. The GroCo compost-amended soil at the UW test site is a sawdust/municipal waste mixture (3:1 ratio, by volume) that is composted in large windrows for at least 1 year. The Cedar Grove compost is a yard waste compost that is also composted in large windrows.

Plots were planted using a commercial turfgrass mixture during the Spring 1994 season for the Urban Horticulture sites and in the fall of 1997 for the Timbercrest and Woodmoor sites. Fertilizer was added to all plots during plot establishment (16-4-8 N-P_2O_5-K_2O) broadcast spread over the study bays at the rate recommended on the product label (0.005 lb fertilizer/ft^2). Due to the poor growth of turf on the control plots, and in order to simulate what would have likely been done anyway on a typical residential lawn, an additional application of 0.005 lb/ft^2 was made to the UW control plots on May 25, 1995. At the new test plots at Timbercrest and Woodmoor , glacial till soil was added to the bays and compacted before adding compost. Cedar Grove compost was added at a 2:1 soil:compost rate and rototilled into the soil surface. Once installed, all bays were cropped with perennial ryegrass.

Sub-surface flows and surface runoff during rains were measured and sampled using special tipping bucket flow monitors (Harrison, *et al.* 1997). The flow amounts and rates were measured by use of tipping bucket type devices attached to an electronic recorder. Each tip of the bucket was calibrated for each site and checked on a regular basis to give rates of surface and subsurface runoff from all plots. Surface runoff decreased by five to ten times after amending the soil with compost (4 inches of compost tilled 8 inches in the soil), compared to unamended sites. However, the concentrations of many pollutants increased in the surface runoff, especially associated with leaching of nutrients from the compost. The surface runoff from the compost-amended soil sites had greater concentrations of almost all constituents, compared to the surface runoff from the soil-only test sites. The only exceptions being some cations (Al, Fe, Mn, Zn, Si), and toxicity, which were all lower in the surface runoff from the compost-amended soil test sites. The concentration increases in the surface runoff and subsurface flows from the compost-amended soil test site were quite large, typically in the range of 5 to 10 times greater. The only exceptions being for Fe, Zn, and toxicity. Toxicity tests indicated reduced toxicity of the water after passing through the soil at both the soil-only and at the compost-amended test sites, compared to surface runoff. This was likely due to the sorption or ion exchange properties of the compost.

Compost-amended soils caused increases in concentrations of many constituents in the surface runoff. However, the compost amendments also significantly decreased the amount of surface runoff leaving the test plots. Table 9 summarizes these expected changes in surface runoff and subsurface flow mass pollutant discharges associated with newly placed compost-amended soils. It is interesting to note that the surface runoff and the subsurface runoff both decreased compared to the soil-only sites. This was due to the increased evapo-transpiration that occurred at the compost-amended soil sites. The shallow soils in the Seattle area overlie low-permeable subsoils, preventing increased deep infiltration, even with enhanced surface infiltration. The original concept of using compost-amended soils in this area was to retain moisture in the surface soils longer than current conditions, making it more susceptible to evaporation. These test results indicate that this concept is correct and that surface runoff and subsurface flows can both be substantially decreased during the low-intensity rains common for this area.

All of the surface runoff mass discharges from the amended soil test plots were reduced from 2 to 50 percent compared to the unamended discharges. However, many of the subsurface flow mass discharges increased, especially for ammonia (340% increase), phosphate (200% increase), plus total phosphorus, nitrates, and total nitrogen (all with 50% increases). Most of the other constituent mass discharges in the subsurface flows decreased. During later field pilot-scale tests, Clark and Pitt (1999) also found that bacteria was reduced by about 50% for every foot of travel through columns having different soils and filtration media.

Table 9. Changes in Pollutant Discharges from Surface Runoff and Subsurface Flows at New Compost-Amended Sites, Compared to Soil-Only Sites

Constituent	Surface Runoff Discharges (mass), Amended-Soil Compared to Unamended Soil	Subsurface Flow Discharges (mass), Amended-Soil Compared to Unamended Soil
Runoff Volume	0.09	0.29
Phosphate	0.62	3.0
Total phosphorus	0.50	1.5
Ammonium nitrogen	0.56	4.4
Nitrate nitrogen	0.28	1.5
Total nitrogen	0.31	1.5
Chloride	0.25	0.67
Sulfate	0.20	0.73
Calcium	0.14	0.61
Potassium	0.50	2.2
Magnesium	0.13	0.58
Manganese	0.042	0.57
Sodium	0.077	0.40
Sulfur	0.21	1.0
Silica	0.014	0.37
Aluminum	0.006	0.40
Copper	0.33	1.2
Iron	0.023	0.27
Zinc	0.061	0.18

Selection of Material for use as Soil Amendments

Additional useful data for soil amendments and the fate of infiltrated stormwater has also been obtained during media filtration tests conducted as part of EPA and WERF-funded projects (Clark and Pitt 1999). A current WERF-funded research at the University of Alabama also includes a test grass swale where amended soil (with peat and sand) is being compared to native conditions. Both surface and subsurface quantity and quality measurements are being made.

The University of Washington and other Seattle amended soil test plots (Pitt, *et al.* 1999a and Harrison 1997) examined GroCo compost-amended soil (a sawdust/municipal waste mixture) and Cedar Grove compost-amended soil (yard waste compost). In addition, an older GroCo compost test plot was also compared to the new installations. These were both used at a 2:1 soil:compost rate. As noted previously, these compost-amended soils produced significant increases in the infiltration rates of the soils, but the new compost test sites showed large increases in nutrient concentrations in surface runoff and the subsurface percolating water. However, most metals showed major concentration and mass reductions and toxicity measurements were also decreased at the amended soil sites. The older compost-amended test plots still indicated significant infiltration benefits, along with much reduced nutrient concentrations. Table 10 shows the measured infiltration rates at the old and new compost-amended test sites in the Seattle area (all Alderwood glacial till soil).

Table 10. Measured Infiltration Rates at Compost-Amended Test Sites in Seattle (Pitt, *et al.* 1999a)

	Average Infiltration Rate (cm/h) (in/h)
UW test plot 1 Alderwood soil alone	1.2 (0.5)
UW test plot 2 Alderwood soil with Ceder Grove compost (old site)	7.5 (3.0)
UW test plot 5 Alderwood soil alone	0.8 (0.3)
UW test plot 6 Alderwood soil with GroCo compost (old site)	8.4 (3.3)
Timbercrest test plot Alderwood soil alone	0.7 (0.3)
Timbercrest test plot Alderwood soil with Cedar Grove compost (new site)	2.3 (0.9)
Woodmoor test plot Alderwood soil alone	2.1 (0.8)
Woodmoor test plot Alderwood soil with Cedar Grove compost (new site)	3.4 (1.3)

The soil that was not amended with either compost had infiltration rates ranging from 0.7 to 2.1 cm/h (0.3 to 0.8 in/h). The old compost amended soil sites had infiltration rates of 7.5 and 8.4 cm/h (3.0 and 3.3 in/h), showing an increase of about 6 to 10 times. The newer test plots of compost-amended soil had infiltration rates of 2.3 and 3.4 cm/h (0.9 to 1.3 in/h), showing increases of about 1.5 to 3.3 times. The older compost-amended soil test sites showed better infiltration rates that the newer test sites. It is likely that the mature and more vigorous vegetation in the older test plots had better developed root structures and were able to maintain good infiltration conditions, compared to the younger plants in the new test plots. The use of amended soils can be expected to significantly increase the infiltration rates of problem soils, even for areas having shallow hard pan layers as in these glacial till soils. There was no significant difference in infiltration for either compost during these tests.

Our earlier work on the performance of different media for use for stormwater filtration is useful for selecting media that may be beneficial as a soil amendment, especially in providing high infiltration rates and pollutant reductions. As reported by Clark and Pitt (1999), the selection of the media needs to be based on the desired pollutant removal performance and the associated conditions, such as land use. The following are the general rankings we found in the pollutant removal capabilities of the different media we tested with stormwater:

- Activated carbon-sand mixture (very good removals with minimal to no degradation of effluent)
- Peat-sand mixture (very good removals, but with some degradation of effluent with higher turbidity, color, and COD)
- Zeolite-sand mixture and sand alone (some removals with minimal degradation of effluent)
- Enretech (a cotton processing mill waste)-sand mixture (some removals with minimal degradation of effluent)
- Compost-sand mixture (some removals but with degradation of effluent with higher color, COD, and solids)

These materials act mostly as ion-exchange materials. This means that when ions are removed from solution by the material, other ions are then released into the solution. In most instances, these exchangeable ions are not a problem in groundwaters. During these tests and for the materials selected, the exchangeable ion for activated carbon was mostly sulfate; while the exchangeable ion for the compost was usually potassium. The zeolite appears to exchange sodium and some divalent cations (increasing hardness) for the ions it sorbs. Of course, ion exchange can't continue indefinitely, as the exchangeable ions do become exhausted. Our tests measured the breakthrough conditions for these media, but in all cases, we found the material to physically clog with particulates long before chemical breakthrough would occur. Soil amendments can be applied with specific objectives to ensure a suitable life for the material and to prevent excessive contamination of the soil-amendment material before it may need to be replaced. Our evaluations indicate that most soil-amendment applications should function for several decades before restoration is needed.

Summary of Soil Infiltration Characteristics

Table 11 compares the infiltration test results from these field and laboratory investigations. The low-head laboratory and field results were similar, except for the higher rates observed for the noncompacted clay field tests. These higher results could reflect actual macro-structure conditions in the natural soils, or the compaction levels obtained in the laboratory were unusually high compared to field conditions. In addition, the high-head laboratory test results produced infiltration rates substantially greater than for the similar low-head results for sandy soil conditions, but not for the other soils. We have scheduled a "final" series of tests over the coming month to examine some of these issues again. Specifically, we anticipate repeating the low-head laboratory infiltration tests, but with higher resolution measurements. In addition, we will conduct a new series of field measurements, and will specifically measure soil density along with soil water and texture. Finally, we will use selected field soil samples for controlled compaction tests in the laboratory. These tests should enable us to specifically investigate alternative conventional infiltration equations, and examine needed modifications for typical compaction conditions; we will confirm a simple method to measure compaction in the field; and we will verify the laboratory measurements for field applications.

Table 11. Comparison of Infiltration Rates from Different Test Series

Group	Field Test Average Infiltration Rates (in/h and COV)	Low-head Laboratory Test Results	High-head Laboratory Test Results
Noncompacted sandy soils	13 (0.4)	8 to 9.5 in/h	30 to 120 in/h
compact sandy soils	1.4 (1.3)	3 to 5 in/h	0.5 to 60 in/h

Noncompacted and dry clayey soils	9.8 (1.5)	0.4 to 0.6 in/h	0 to 0.3 in/h
All other clayey soils (compacted and dry, plus all wetter conditions)	0.2 (2.4)	0 to 0.4 in/h	0 to 0.02 in/h
Noncompacted silty and loamy soils	na	0.25 to 0.6 in/h	0.5 to 3 in/h
Compacted silty and loamy soils	na	0 to 0.02 in/h	0 to 0.04 in/h

Site Suitability Criteria for Stormwater Infiltration

Soil Infiltration Rate/Drawdown Time

For most treatment scenarios, there is a definite tradeoff between storage and treatment rate. There is an indefinite number of combinations of storage and infiltration that can provide treatment of a set condition. At one extreme, high treatment rates (infiltration rates) can be coupled with minimal storage, while at the other extreme, a treatment rate equal to the average long-term flow can be coupled with a suitably-sized storage facility to even out the periods of higher than average flows. In conventional optimization approaches, numerous combinations are examined and the most cost-effective combination is selected. In the treatment of stormwater, these calculations are more complicated because of the wisely varying flow rates and interevent periods.

Continuous, long-term, simulations using a locally calibrated and verified stormwater model is needed in order to determine the drainage times to obtain the desired dry period between events for specific designs. The following subsection presents some recent information pertaining to the need for keeping an infiltration area under aerobic conditions.

Preventing Soil from going Anaerobic between Rain Events

This discussion presents some experimental results that shows the importance of preventing media used to capture stormwater pollutants from going anaerobic. These tests were conducted by Clark (2000) as part of the WERF-sponsored research by Johnson, *et al*. (2003) on a variety of filtration media (activated carbon, peat moss, compost, and sand) and therefore represent a range of soil conditions (with the exception of the activated carbon).

The media were exposed to a concentrated solution made up of spiked tap water (10 mg/L of lead, copper, zinc, iron, nitrate, phosphate, and ammonia) for several hours. The water was then filtered through a 0.45-µm membrane filter. The amount of material sorbed onto the media was calculated using the pre- and post-sorption water concentrations. After rinsing with a buffered distilled water to remove any loosely bound material and to replace any concentrated pore water, the "loaded" media were exposed to water collected from an urban lake for a period of several weeks. One sample of each medium was maintained in an aerobic environment with continuous aeration to keep the lake water saturated in oxygen. The other sample of each medium was exposed to the urban water while in sealed BOD bottles, where the naturally-occurring matter/organisms in the water would consume the oxygen and create an anaerobic environment. At the end of the exposure time, the dissolved oxygen (DO) and the oxidation-reduction potential (ORP) of each aerobic and anaerobic sample were taken. The samples were then filtered through a 0.45-µm gel membrane filter, and the filtrates were analyzed for the ammonia, nitrate, total nitrogen, phosphate, total phosphorus, calcium, magnesium, iron, copper, lead and zinc.

For all three forms of nitrogen measured in this experiment (see Figures 12 and 13, total nitrogen not shown), pollutant retention was equal to or greater under aerobic exposure conditions than under anaerobic exposure conditions. For ammonia, the compost released ammonia during the initial sorption. When exposed to aerobic conditions, additional release did not occur. Additional release/leaching did occur, however, when the compost was exposed to anaerobic conditions. Previously sorbed ammonia was released from the peat moss when the water went anaerobic. Peat moss also released previously-adsorbed nitrate when the exposure water went anaerobic. Within experimental error, no other media were shown to release nitrate when exposed to anaerobic conditions. The behavior of all media for total nitrogen reflected the behavior seen for nitrate (Figure 13).

Phosphorus retention (phosphate: Figure 14; total phosphorus: not shown) on carbon, peat, and sand was excellent under both aerobic and anaerobic exposure conditions, indicating that the phosphate that is sorbed on the media will tend to remain on the media, and, if the sorption capacity is not full, additional phosphorus may be sorbed to the media during the long-term exposure. For compost, retention was better and/or leaching was lesser when the media are held under aerobic conditions than under anaerobic conditions.

Figure 12. Behavior of ammonia-nitrogen under aerobic and anaerobic conditions.

Figure 13. Behavior of nitrate-nitrogen under aerobic and anaerobic conditions.

Figure 14. Behavior of phosphate under aerobic and anaerobic conditions.

The results for the heavy metals are shown in Figures 15, 16, 17, and 18 (copper, iron, lead, and zinc, respectively). As expected, once the metals were adsorbed onto the media, only negligible removal occurred during rinsing and exposure, except for the iron-compost combination. For copper, lead, and zinc, the sorption onto the peat and compost appeared to be permanent, likely due to the formation of complexes with the organic compounds on the surface of these materials. Retention by the sand and the carbon also appears to be permanent under the conditions of this experiment. Iron (Figure 16) was adsorbed to all four media. However, when the initial sorption pH is closer to neutral (as in these experiments), the bonding between the compost and the iron was not as strong, and pollutant release occurred during anaerobic conditions. When the test was repeated with a lower initial sorption pH, pollutant release was not seen for any of the metals under either aerobic or anaerobic conditions.

Calcium and magnesium (data not shown) were leached from the compost and peat media (loss greater than total amount sorbed), likely due to competition between these ions and the other ions in solution (especially the heavy metals) for sorption sites on these media. The leaching is significantly greater under anaerobic conditions, where a reducing environment has been developed. Minimal sorption of calcium and magnesium was seen on the carbon and the sand.

Figure 15. Behavior of copper under aerobic and anaerobic conditions.

Figure 16. Behavior of iron under aerobic and anaerobic conditions.

Figure 17. Behavior of lead under aerobic and anaerobic conditions.

Figure 18. Behavior of zinc under aerobic and anaerobic conditions.

These results indicate that permanent retention by the media for the heavy metals may occur even when the filter goes anaerobic. However, retention of the nutrients may not occur under anaerobic conditions, especially for compost. This indicates that in situations where nutrient releases will cause problems for the receiving water, the soil needs to stay aerobic.

Depth to Bedrock, Water Table, or Impermeable Layer

The infiltrating water requires a long flow path through the soil for deep water tables, or deep impermeable layers. This longer flow path, and greater contact time, allows more pollutants to be removed from the water and offers greater protection to the groundwater. Table 12 is from Clark (2000) and Johnson, et al. (2003) and describes the level of removal for several pollutants after an 18 inch depth of flow. The amended sand using peat had quite large removals for these compounds, and for the bacteria also shown in the following figure.

Table 12. Median Removal Efficiencies Based on Pilot-Scale Testing

	Loam	Peat-Sand	Compost-Sand	Sand
Turbidity (unfiltered)		68 (0.04)	65 (0.01)	75 (0.01)
Total Solids		35 (0.05)		4 (0.01)
Dissolved Solids		40 (0.02)		
Hardness	13 (0.04)	68 (0.01)		
Calcium (total)	20 (0.01)	96 (0.02)		
Iron (total)		42 (0.05)	44 (0.05)	

*Note: the p values for the statistically significant removals, based on the Wilcoxon sign-rank test, are shown in parentheses.

Additional experimental data was collected by Pitt, *et al.* (1997) in the Seattle area during tests to examine the benefits of amending local soils when infiltrating stormwater. As part of these tests, surface runoff was compared to subsurface flows. Table 13 lists the average surface and subsurface flow concentrations from these test plots, plus the calculated reductions in the concentrations (based on the average values). These test plots had 12 inches of soil as the flow depth, and this data are for unamended soils. The amended soils (50% compost with 50% soil) showed fewer concentration reductions during the first year of the tests due to leaching of pollutants from the compost additions. However, older test plots from the University of Washington showed that the compost no longer released pollutants, but had significant pollutant removals, plus enhanced infiltration and evapotranspiration losses of the runoff water.

Table 13. Average (and COV) Values for all Runoff and Subsurface Flow Samples

	Soil-only plots		**Percent reductions in average concentrations after infiltration**
Constituent (mg/L, unless noted)	**Surface Runoff**	**Subsurface Flows**	
PO_4-P	0.27 (1.4)	0.17 (2.0)	37%
TP	0.49 (1.0)	0.48 (2.2)	10
NH_4-N	0.65 (1.7)	0.23 (1.3)	65
NO_3-N	0.96 (1.4)	1.2 (2.5)	-125
TN	2.5 (0.9)	1.9 (0.7)	24
Cl	2.4 (1.0)	2.1 (0.9)	13
SO_4-S	0.68 (1.1)	0.95 (2.0)	-140
Al	11 (1.8)	1.7 (2.1)	85
Ca	12 (1.5)	17 (0.7)	-140
Cu	0.01 (0.8)	0.01 (1.6)	n/a
Fe	4.6 (1.4)	2.8 (1.6)	39
K	5.4 (1.0)	4.6 (0.8)	15
Mg	3.9 (0.8)	5.0 (0.6)	-128
Mn	0.75 (2.9)	0.41 (2.8)	45
Na	3.8 (0.9)	3.4 (0.5)	11
S	1.1 (0.8)	1.3 (1.5)	-120
Zn	0.2 (1.2)	0.05 (2.2)	75
Si	26 (1.7)	8.9 (0.5)	66

The following lists the categories of pollutants associated with each range of concentration reduction:

Large reductions (≥75%):
Al, Zn

Moderate reductions (25 to 74%):
NH_4, Si, PO_4, Fe, Mn

Minimal reductions, or increases (<24%):
TP, TN, Cl, K, Na, NO_3, SO_4, Ca, Mg, S

These data indicate that natural soils certainly can reduce concentrations for a wide range of pollutants. The tests were for relatively shallow soils (1 to 1-1/2 feet in depth) and imply that deeper soils will provide greater benefits. Metals are removed much better than nutrients, and some major ions actually increase (due to ion exchange or leaching).

Soil Physical and Chemical Suitability for Treatment

Cation-Exchange Capacity (CEC)

Much of the groundwater protection offered by soils is associated with its' cation-exchange capacity. The cation-exchange capacity (CEC) of a material is defined as the sum of the exchangeable cations it can adsorb at a given pH. Alternatively, the CEC is a measure of the negative charge present at the sorbent surface. The CEC is generally measured to evaluate the ability of certain soils to sorb K^+ (from fertilizers), heavy metals, and various other target ions whose mobility in the soil is an issue of concern. The CEC is a function of available surface charge per unit area of material, the pH at which exchange occurs, and the relative affinities of the ions to be exchanged for the material surface. The CEC is measured at a specific pH. If the actual pH is less, the CEC also is less.

Sands have low CEC values, typically ranging from about 1 to 3 meq/100g of material. As the organic content of the soil increases, so does its' CEC. Compost, for example, can have a CEC of between 15 and 20 meq/100 grams, while clays can have CEC values of 5 and 60 meq/100 grams. Natural soils can therefore vary widely in the CEC depending on their components. Silt loam soils can have a CEC between 10 and 30 meq per 100 gram for example. Soil amendments (usually organic material, such as compost) can greatly increase the CEC of a soil that is naturally low in organic material, or clays.

Johnson, *et al.* (2003) conducted CEC measurements using standard methods, and also calculated the actual CEC based on the removal and exchange of all cations from a stormwater solution in a variety of filtration media. The capacity calculations confirmed the literature that indicated that peat moss, since it is often formed in calcium-poor conditions, had a high exchange/sorption capacity for calcium and for hardness. For peat, the quantity of cations exchanged was much greater than the standard CEC tests indicated. This likely was a result of the relatively large size of the test molecule for the CEC measurements (a copper trielthylenetetramine complex), which may not have been able to penetrate some of the micropores that the ionic forms of the metals and major ions could penetrate.

	Sand	Peat	Compost
CATION EXCHANGE CAPACITY (calculated from batch tests)	1.41	292	13.5
CATION EXCHANGE CAPACITY (CEC analysis)	3.49	21.47	18.83

The total cation content of a water can be easily calculated knowing the major ion content of the water and the associated equivalent weights. The sum of the cations must equal the sum of the anions (expressed in equivalent weight). Assume the following typical stormwater characteristics:

Component	mg/L	Equivalent weight	meq/L
Ca^{2+}	13.3	20.0	0.67
Mg^{2+}	3.3	12.2	0.27
Na^+	3.9	23.0	0.17
K^+	2.3	39.1	0.06
		Total cations:	**1.17**
HCO_3^{3-}	36.7	61.0	0.60
SO_4^{2-}	22.4	48.0	0.47
Cl^-	3.7	35.5	0.10
		Total anions:	**1.17**

The above example only lists the major ions in the water, although we may be most interested in the heavy metals that are also cations. However, the concentrations of the dissolved heavy metals in stormwater are rarely more than about 0.10 mg/L and therefore contribute little to the total cation content of the water. The total (unfiltered) heavy metal concentrations of some metals can be much higher, but only the ionic forms affect the CEC. The total hardness of the above sample (the sum of the divalent cations) is 0.94 meq/L. With an equivalent weight of 50 meq/L per mg/L as $CaCO_3$, the resulting hardness concentration is about 47 mg/L.

The consumption of the CEC in the soil can be calculated by dividing the soil total CEC by the total cation content of the water. If the soil is ½ meter thick, and the soil density is about 1.5 grams/cc, the total CEC of a soil having a CEC of 10 meq/100 grams, per m^2, is approximately 75,000 meq. If the stormwater has a total cation content of about 1.17 meq/L, then the total water treatment capacity of the soil, per m^2, is about 64,000 L, or a column of water about 64 m high. If the soil is only receiving rain water (having this cation content), and 1 m of rain falls per year, then the CEC content of the soil would be exhausted in about 60 to 70 years. The natural soil building process, and accumulating layers of organic material, would continue to "recharge" the soil CEC in an undeveloped setting, with very slow changes in the soil CEC with time. In an urban area infiltration device, the CEC of a soil could be exceeded much sooner, unless soil amendments are periodically added.

• Problem: Determine the approximate "life" of the CEC of a soil in an infiltration device having the following characteristics:
- the soil in an urban infiltration device has a CEC of 200 meq/L (averaged for ½ m in depth and soil had a dry density of 1.6 g/cm^3),
- receives the runoff from a paved area 30 times the area of the infiltration device,
- 1 m of rainfall a year, and paved area Rv is 0.85, and
- the total cation content of the runoff water is 1.0 meq/L

• Solution:

- total CEC content of soil (per m^2):

- total cation content of a years worth of runoff (per 30 m^2):

- therefore, the unit's CEC would be able to protect the groundwater for about 63 years, a suitable design period. However, if the soil CEC was only 5 meq/100 grams, then the facility would only protect the groundwater for about 3 years. In this case, either the infiltration device should be made larger, the contributing paved area made smaller, or the soil will have to be replaced every several years.

Impact of Major Ions
Most of the soil treatment processes affect major constituents in the water in addition to the targeted pollutant. As noted above, the major cations in the water (such as Ca, Mg, Na, and K) would all be affected by the CEC capacity of the soil, not just the heavy metals of most concern. The following illustrates the potential effects of the major cations on heavy metal exchange.

Johnson, *et al.* (2003) examined the ions Ca, Mg, K, and Na during uptake tests to measure any correlation between metal sorption and ion desorption on different materials. The following summaries are for composites and peat mixtures. Figures 19 and 20 show zinc sorbed and major ions desorbed from peat-sand and compost. For comparison, a few batch equilibrium isotherm tests were also performed with copper. Figure 21 shows copper sorbed and ions desorbed for compost.

Figure 19. Zinc Sorbed and Major Ions Desorbed during Batch Equilibrium Tests for Zinc onto Peat-Sand

Figure 20. Zinc Sorbed and Major Ions Desorbed during Batch Equilibrium Tests for Zinc onto Compost

Figure 21. Copper Sorbed and Major Ions Desorbed during Batch Equilibrium Tests for Copper onto Compost

The amount of Ca desorbed from the peat-sand appeared to increase as the quantity of zinc sorbed increased, indicating the possibility that Ca participates in ion exchange with the metals. The amount of Mg, K, and Na that desorbed were comparatively small and any correlation with zinc sorption was uncertain.

When examining tests with compost, the Ca and Mg desorption (mg/g) increased as zinc and copper sorption (mg/g) increased, an indication that ion exchange was occurring. Na and K were also desorbed from the compost, but the amount of Na and K desorbed appeared to hold roughly constant and did not appear to be related to zinc and copper uptake.

Comparison of Competing Metals

The results from kinetic uptake experiments were also used by Johnson, *et al.* (2003) to examine which metals were removed the fastest and to the greatest degree under the given test conditions. Figures 22 and 23 show the fraction of the initial metal concentrations (Ct/Co) remaining in solution verses time for the three final media.

Figure 22. Fraction of Initial Metal Concentration Remaining verses Time for Metals onto Peat-Sand for a Mixed Metal Solution.

Figure 23. Fraction of Initial Metal Concentration Remaining verses Time for Metals onto Compost for a Mixed Metal Solution.

The order of preference for removal, using low metal concentrations, was Cd, Pb>Zn,Cu>Cr>Fe for peat-sand and Cd>Zn>Pb>Cu>Cr>Fe for compost. Subsequent column tests, which used stormwater runoff, were able to replicate the test results noted above.

Sodium Adsorption Ratio (SAR)
The sodium adsorption ratio can radically affect the performance of an infiltration device. According to Swift (2003), soils with an excess of sodium ions, compared to calcium and magnesium ions, remain in a dispersed condition, almost impermeable to rain or applied water. A "dispersed" soil is extremely sticky when wet, tends to crust, and becomes very hard and cloddy when dry. Water infiltration is usually severely restricted. Dispersion caused by sodium may result in poor physical soil conditions and water and air do not readily move through the soil. An SAR value of 15, or greater, indicates that an excess of sodium will be adsorbed by the soil clay particles. This can cause the soil to be hard and cloddy when dry, to crust badly, and to take water very slowly. SAR values near 5 can also cause problems, depending on the type of clay present. Montmorillonite, vermiculite, illite and mica-derived clays are more sensitive to sodium than other clays. Additions of gypsum (calcium sulfate) to the soil can be used to free the sodium and allow it to be leached from the soil.

The SAR is calculated by using the concentrations of sodium, calcium, and magnesium (in meq) in the following formula:

Swift (2003) presented the following example to show how the SAR is calculated:

A soils lab reported the following chemical analyses:

100 pounds/acre of sodium (Na^{+})
5000 pounds/acre of calcium (Ca^{+2})
1500 pounds/acre of magnesium (Mg^{+2})

These concentrations need to be first converted to parts per million (ppm), and then to meq/L. An acre of soil (43,560 square feet, or 4047 square meters), 6 inches deep (15 cm), weighs about 2,000,000 pounds (910,000 kg) and contains 22,000 cubic feet of soil (620 cubic meters). The pounds reported per acre are divided by 2 to produce ppm:

100 pounds/acre of Na divided by 2 = 50 ppm of Sodium
5000 pounds/acre of Ca divided by 2 = 2500 ppm of Calcium
1500 pounds/acre of Mg divided by 2= 750 ppm of Magnesium

The ppm values are divided by the equivalent weight of the element (given previously in the CEC discussion) to obtain the milliequivalent (meq) values. The milliequivalent weights of Na, Ca, and Mg in this example are:

50 ppm of Na divided by 23 = 2.17
2500 ppm of Ca divided by 20 = 125
750 ppm of Mg divided by 12.2 = 61.5

The SAR is therefore:

This value is well under the critical SAR value of 15, or even the critical value of 5 applicable for some clays. This soil is therefore not expected to be a problem. However, if the runoff water contained high levels of sodium in relationship to calcium and magnesium, a SAR problem may occur in the future, necessitating the addition of gypsum to the infiltration area. The amount of gypsum (calcium sulfate) needed to be added can be determined from an analysis of the soil in the infiltration area.

Cold Climate and Impact of Roadway Deicers
As discussed by Pitt, *et al*. (1994; 1994; 1999), some dissolved minerals are of concern when infiltrating stormwater. Salt applications for winter traffic safety is a common practice in many northern areas and the sodium and chloride, which are collected in the snowmelt, travel down through the vadose zone to the groundwater with little attenuation. Most salts are not attenuated during movement through soil. In fact, salt concentrations typically increase due to leaching of salts out of soils. Groundwater salt concentration decreases may occur with dilution by less saline recharging waters.

Soil is not very effective at removing most salts. On Long Island, New York, it was noted that the heavy metals load was significantly reduced during passage through the soil, while chloride was not reduced significantly. Once contamination with salts begin, the movement of salts into the groundwater can be rapid. The salt concentration may not lessen until the source of the salts is removed. At three stormwater infiltration locations in Maryland, the nearby use of deicing salts and their subsequent infiltration to the groundwater shifted the major-ion chemistry of the groundwater to a chloride-dominated solution. Although deicing occurred only three to eight times a year, increasing chloride concentrations were noted in the groundwater throughout a 3-year USGS study, indicating that groundwater systems are not easily purged of conservative contaminants, even if the groundwater flow rate is relatively high.

Because of the unlikely mitigation of salts from deicing operations, the chloride content of groundwaters will increase in these areas. There have been many EPA reports describing the effects of deicers and alternatives that can be used. There are no pretreatment options available to remove the chlorides from snowmelt before it enters an infiltration area, and there is no soil process that will attenuate the salt movement to the groundwater.

Conclusions

Very large errors in soil infiltration rates can easily be made if published soil maps are used in conjunction with most available models for typically disturbed urban soils, as these tools ignore compaction. Knowledge of compaction (which can be measured using a cone penetrometer, or estimated based on expected activity on grassed areas, or directly measured) can be used to more accurately predict stormwater runoff quantity, and to better design biofiltration stormwater control devices. In most cases, the mapped soil textures were similar to what was actually measured in the field. However, important differences were found during many of the 153 tests. Although the COV values are generally high (0.5 to 2), they are much less than if compaction was ignored. These data can be fitted to conventional infiltration models, but the high variations within each of these categories makes it difficult to identify legitimate patterns, implying that average infiltration rates within each event may be most suitable for predictive purposes. The remaining uncertainty can probably best be described using Monte Carlo components in runoff models.

The field measurements of infiltration rates during these tests were all substantially larger than expected, but comparable to previous standard double-ring infiltrometer tests in urban soils. Other researchers have noted the general over-predictions of ponding infiltrometers compared to actual observations during natural rains. In all cases, these measurements are suitable to indicate the relative effects of soil texture, compaction, and soil-water on infiltration rates. However, the measured values can be directly used to predict the infiltration rates that may be expected from stormwater infiltration controls that utilize ponding (most infiltration and biofiltration devices).

The use of soil amendments, or other methods to modify soil structure and chemical characteristics, is becoming an increasingly popular stormwater control practice. However, little information is available to reasonably quantify benefits and problems associated with these changes. An example examination of appropriate soil chemical characteristics, along with surface and subsurface runoff quantity and quality, was shown during the Seattle tests. It is recommended that researchers considering soil modifications as a stormwater management option conduct similar local tests in order to understand the effects these soil changes may have on runoff quality and quantity. During these Seattle tests, the compost was found to have significant sorption and ion exchange capacity that was responsible for pollutant reductions in the infiltrating water. However, the newly placed compost also leached large amounts of nutrients to the surface and subsurface waters. Related tests with older test plots in the Seattle area found much less pronounced degradation of surface and subsurface flows with aging of the compost amendments. In addition, it is likely that the use of a smaller fraction of compost would have resulted in fewer negative problems, while providing most of the benefits. Again, local studies using locally available compost and soils, would be needed to examine this emerging stormwater management option more thoroughly.

This information can be effectively used in the modeling of small-scale stormwater controls, such as biofiltration devices located near buildings and grass swales. As an example of the benefits these devices may provide in typical urban areas, WinSLAMM, the Source Loading and Management Model (www.winslamm.com) (Pitt and Voorhees 1995) was used to calculate the expected reductions in annual runoff volumes for several different controls. Table 14 illustrates these example reductions for Phoenix (9.3 in/year of rainfall), Seattle (33.4 in/yr), and Birmingham, AL (52.5 in/yr). The reductions are only for roof runoff control, but illustrate the magnitude of the reductions possible. The calculations are based on long-term continuous simulations (about 5 years of historical rain records were used). The test site is a single-family residential area with silty soils and directly connected roofs. In this type of area, directly connected residential roofs produce about 30 to 35% of the annual runoff volume for the rain conditions in these three cities.

Table 14. Example Calculations of Benefits of On-Site Stormwater Controls (% reduction of annual roof runoff volumes).

	Phoenix, AZ	Seattle, WA	Birmingham, AL
Roof garden (1in/h amended soils, 60ft^2 per house)	96%	100%	87%
Cistern for stormwater storage and reuse of roof water (375ft^3 per house)	88	67	66
Disconnect roof runoff to allow drainage onto silty soils	91	87	84
Green roof (vegetated roof surface)	84	77	75

The roof garden option using amended soils provides large reductions, even for a relatively small treatment area. This is especially useful for sites with extremely poor soils or small landscaped areas. Biofiltration options can be sized to provide specifically desired runoff reductions, considering actual, or improved, soil conditions. This table also shows potential runoff reductions associated with storage of roof runoff for later reuse for on-site irrigation, and an option for a green roof, where the roof surface is actually vegetated allowing increased evapotranspiration.

This table shows that even for a wide range of rainfall conditions, these options can provide substantial reductions in runoff volume from residential roofs. An estimated 20 to 35% reductions in annual runoff volumes for the complete drainage areas would be expected for these alternatives. Obviously, these controls can be applied to the runoff from other areas, in addition to the roofs, for additional runoff reductions.

Acknowledgements

Much of the infiltration measurements were carried out as class projects by University of Alabama at Birmingham hydrology, experimental design, and soil mechanics students. Choo Keong Ong directed the laboratory compaction tests, while Janice Lantrip directed the field infiltration tests. Specific thanks are given to the following students who assisted with the projects summarized here: Robecca Martin, Stacey Sprayberry, Muhammad Salman, Wade Burcham, Brian Adkins, Sarah Braswell, Scott Lee, and Jennifer Harper. Partial support was provided by the Urban Watershed Management Branch, U.S. Environmental Protection Agency, Edison, NJ for portions of the field measurements, including the amended soil tests. EPA support was also provided for the research on potential groundwater contamination associated with stormwater infiltration. Thomas O'Connor was the EPA project officer and provided valuable project guidance.

References

Akan, A. O., *Urban Stormwater Hydrology: A Guide to Engineering Calculations*. Lancaster. PA: Technomic Publishing Co., Inc., 1993.

American Society of Testing and Materials. *Annual Book of ASTM Standards.* West Conshohocken. PA: ASTM vol. 04.08, 1996.

Bedient, P.B., and Huber, W.C. *Hydrology and Floodplain Analysis.* New York: Addison-Wesley Publishing Co., 1992.

Box, G.E.P., Hunter, W.G., and Hunter, J.S.. *Statistics for Experimenters*. John Wiley and Sons, New York, 1978.

Clark, S. and R. Pitt. *Stormwater Treatment at Critical Areas, Vol. 3: Evaluation of Filtration Media for Stormwater Treatment*. U.S. Environmental Protection Agency, Water Supply and Water Resources Division, National Risk Management Research Laboratory. EPA/600/R-00/016, Cincinnati, Ohio. 442 pgs. October 1999.

Clark, S. *Urban Stormwater Filtration: Optimization of Design Parameters and a Pilot-Scale Evaluation*. Ph.D. Dissertation. University of Alabama at Birmingham, Birmingham, Alabama. 450 pages. 2000.

Das, B.M. *Principals of Geotechnical Engineering* Boston: PWS Publishing Co., 1994.

Dickey-John Corporation. *Installation Instructions Soil Compaction Tester.* Auburn, Illinois:, 1987.

Gilbert, R.O., *Statistical Methods for Environmental Pollution Monitoring* New York: Van Nostrand Reinhold Publishing Co., 1987.

Harrison, R.B., M.A. Grey, C.L. Henry, and D. Xue. *Field Test of Compost Amendment to Reduce Nutrient Runoff*. Prepared for the City of Redmond. College of Forestry Resources, University of Washington, Seattle. May 1997.

Horton, R.E. "An approach toward a physical interpretation of infiltration capacity." *Transactions of the American Geophysical Union*. Vol. 20, pp. 693 – 711. 1939.

Johnson, P.D., R. Pitt, S.R. Durrans, M. Urrutia, and S. Clark. *Innovative Metals Removal Technologies for Urban Stormwater*. Water Environment Research Foundation. WERF 97-IRM-2. Alexandria, VA. forthcoming in 2003.

McCuen, R. *Hydrologic Analysis and Design, 2nd edition*. Prentice Hall. 1998.

Morel-Seytoux, H.J. "Derivation of Equations for Variable Rainfall Infiltration." *Water Resources Research. pp.* 1012-1020. August 1978.

NRC (National Research Council), Groundwater Recharge Committee, National Academy of Science*. Ground Water Recharge using Waters of Impaired Quality*. ISBN 0-309-05142-8. National Academy Press, Washington, D.C. 1994. 284 pages.

NRCS. Soil Quality Institute 2000, Urban Technical Note 2, as reported by Ocean County Soil Conservation District, Forked River, NJ. 2001.

Pitt, R. "Small storm hydrology and why it is important for the design of stormwater control practices." In: *Advances in Modeling the Management of Stormwater Impacts, Chapter 4, Volume 7*. (Edited by W. James). Computational Hydraulics International, Guelph, Ontario and Lewis Publishers/CRC Press. 1999.

Pitt, R. and J. Lantrip. "Infiltration through disturbed urban soils." In: *Advances in Modeling the Management of Stormwater Impacts, Chapter 1,Volume 8*. (Edited by W. James). Computational Hydraulics International, Guelph, Ontario. 2000.

Pitt, R. and J. Voorhees. "Source loading and management model (SLAMM)." *Seminar Publication: National Conference on Urban Runoff Management: Enhancing Urban Watershed Management at the Local, County, and State Levels*. March 30 – April 2, 1993. Center for Environmental Research Information, U.S. Environmental Protection Agency. EPA/625/R-95/003. Cincinnati. Ohio. pp. 225-243. April 1995.

Pitt, R. and S.R. Durrans. *Drainage of Water from Pavement Structures*. Alabama Dept. of Transportation. 253 pgs. September 1995.

Pitt, R. *Small Storm Urban Flow and Particulate Washoff Contributions to Outfall Discharges*, Ph.D. Dissertation, Civil and Environmental Engineering Department, University of Wisconsin, Madison, WI, November 1987.

Pitt, R., B. Robertson, P. Barron, A. Ayyoubi, and S. Clark. *Stormwater Treatment at Critical Areas: The Multi-Chambered Treatment Train (MCTT)*. U.S. Environmental Protection Agency, Wet Weather Flow Management Program, National Risk Management Research Laboratory. EPA/600/R-99/017. Cincinnati, Ohio. 505 pgs. March 1999.

Pitt, R., J. Lantrip, R. Harrison, C. Henry, and D. Hue. *Infiltration through Disturbed Urban Soils and Compost-Amended Soil Effects on Runoff Quality and Quantity*. U.S. Environmental Protection Agency, Water Supply and Water Resources Division, National Risk Management Research Laboratory. EPA 600/R-00/016. Cincinnati, Ohio. 231 pgs. 1999a. Available at: http://www.eng.ua.edu/~rpitt/Publications/BooksandReports/Compacted%20and%20compost%20amended%20soil%20EPA%20report.pdf

Pitt, R., S. Clark, and K. Parmer*. Protection of Groundwater from Intentional and Nonintentional Stormwater Infiltration*. U.S. Environmental Protection Agency, EPA/600/SR-94/051. PB94-165354AS, Storm and Combined Sewer Program, Cincinnati, Ohio. 187 pgs. May 1994. Available at: http://www.eng.ua.edu/~rpitt/Publications/BooksandReports/Groundwater%20EPA%20report.pdf

Pitt, R., S. Clark, and K. Parmer*. Protection of Groundwater from Intentional and Nonintentional Stormwater Infiltration*. U.S. Environmental Protection Agency, EPA/600/SR-94/051. PB94-165354AS, Storm and Combined Sewer Program, Cincinnati, Ohio. 187 pgs. May 1994.

Pitt, R., S. Clark, and R. Field. "Groundwater contamination potential from stormwater infiltration practices." *Urban Water.* Vol. 1, no. 3, pp. 217-236. 1999b.

Pitt, R., S. Clark, K. Parmer, and R. Field. *Groundwater Contamination from Stormwater Infiltration*. Ann Arbor Press. Chelsea, Michigan. 218 pages. 1996.

Pitt, R., with contributions from S. Clark, R. Field, and K. Parmer. *Groundwater Contamination from Stormwater.* ISBN 1-57504-015-8. Ann Arbor Press, Inc. Chelsea, Michigan. 1996*.* 219 pages.

Swift, C.E., Colorado State Cooperative Extension, Grand Junction, CO. updated:01/26/2003. http://www.coopext.colostate.edu/TRA/PLANTS/index.html#http://www.colostate.edu/Depts/CoopExt/TRA/PLANTS/sar.html

Turf Tec International. *Turf Tec Instructions.* Oakland Park, Florida. 1989.

United States Department of Agriculture, Soil Conservation Service (now Natural Resources Conservation Service). *Soil Survey of Chilton County,* Alabama. October 1972.

United States Department of Agriculture, Soil Conservation Service (now Natural Resources Conservation Service), *Soil Survey of Jefferson County,* Alabama. August 1982.

United States Department of Agriculture, Soil Conservation Service (now Natural Resources Conservation Service), *Soil Survey of Shelby County,* Alabama. July 1984.

Willeke, G.E., "Time in Urban Hydrology." *Journal of the Hydraulics Division Proceedings of the American Society of Civil Engineers.* pp. 13-29. January 1966.