

Copyright Robert Pitt @ September 30, 2003

Module 5

Surface and Groundwater Interactions

[Abstract](#)[Introduction](#)[Groundwater Contamination Associated With Stormwater Pollutants](#)[Nutrients](#)[Nutrient Removal Processes in Soil](#)[Pesticides](#)[Pesticide Removal Processes in Soil](#)[Other Organic Compounds](#)[Soil Removal Processes](#)[Pathogens](#)[Soil Removal Processes](#)[Metals](#)[Observed Heavy Metal Groundwater Contamination Associated with Stormwater Infiltration](#)[Dissolved Minerals](#)[Summary and Recommendations](#)[Site Suitability Criteria](#)[Introduction](#)[Critical Areas of Concern when Selecting Stormwater Infiltration Areas](#)[Soil Infiltration Rate/Drawdown Time](#)[Preventing Soil from going Anaerobic between Rain Events](#)[Depth to Bedrock, Water Table, or Impermeable Layer](#)[Soil Physical and Chemical Suitability for Treatment](#)[Cation-Exchange Capacity \(CEC\)](#)[Impact of Major Ions](#)[Comparison of Competing Metals](#)[Cold Climate and Impact of Roadway Deicers](#)[Compacted Urban Soils and their Effects on Infiltration and Bioretention Systems](#)[Introduction and Summary](#)[Infiltration Mechanisms](#)[Prior Infiltration Measurements in Disturbed Urban Soils](#)[Laboratory Controlled Compaction Tests](#)[Laboratory Test Methods](#)[Laboratory Test Results](#)[Summary](#)[Soil Modifications to Enhance Infiltration](#)[Field Studies on Infiltration Capabilities of Compost-Amended Soils](#)[Soil and Compost Analysis](#)[Water Quantity Observations at Test Plots](#)[Water Quality Observations at Test Plots](#)[Visual Appeal of Test Sites and Need for Fertilization](#)[Overall Range of Water Quality Observations in Surface Runoff and Subsurface Flows](#)[Comparison of Water Quality from Amended vs. Unamended Test Plots](#)[Mass Discharges of Nutrients and other Water Quality Constituents](#)[Conclusions](#)

[Groundwater Contamination Potential Associated with Stormwater Infiltration
Compacted Urban Soils and Infiltration
Water Quality and Quantity Effects of Amending Soils with Compost](#)
[References](#)

Abstract

The potential effects of stormwater on groundwater quality can be estimated based on the likely presence of problem constituents in the stormwater, their mobility through soils, the type of treatment received before infiltration, and the infiltration method used. The constituents of most concern include chloride, certain pesticides (lindane and chlordane), organic toxicants (1,3-dichlorobenzene, pyrene and fluoranthene), pathogens, and some heavy metals (nickel and zinc). Reported instances of groundwater contamination associated with stormwater was rare in residential areas where infiltration occurred through surface soils (except for chloride), but was more common (especially for toxicants) in commercial and industrial areas where subsurface infiltration was used.

Introduction

This discussion presents information collected as part of a multi-year research project sponsored by the U.S. EPA (Pitt, *et al.* 1994 and 1996; Pitt, *et al.* 1995; Pitt, *et al.* 1999; Clark and Pitt 1999) and addresses potential groundwater contamination problems associated with stormwater infiltration. Several categories of constituents are discussed that are known to affect groundwater quality: nutrients, pesticides, other organics, pathogens, metals, and dissolved minerals. The intention of this discussion is to identify known stormwater contaminants as to their potential to adversely affect groundwater. This potential is evaluated based on pollutant abundance in stormwater, pollutant mobility in the vadose zone, the treatability of the pollutants, and the infiltration procedure used. Published observations of groundwater contamination in areas of stormwater recharge are also provided in this review paper, along with suggestions to minimize potential contamination problems. Because urban hydrogeology is an active research field, there are many new papers continuously becoming available containing new case studies. The purpose of this discussion is to assemble a collection of information relating to potential groundwater problems that is of great interest to stormwater managers responsible for the design and implementation of infiltration devices and who may be uncertain of these potential problems.

Prior to urbanization, natural groundwater recharge resulted from infiltration of precipitation through pervious surfaces, including grasslands and woods. This infiltrating water was relatively uncontaminated. With urbanization, 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. In addition, the waters available for recharge generally carried increased quantities of pollutants.

There are many types of artificial stormwater infiltration mechanisms that have been used in urbanizing areas in order to decrease discharges of stormwater to surface waters and to help preserve groundwater recharge. These are described in many stormwater design manuals. The following infiltration techniques are most commonly used:

- surface infiltration devices (grass filters and grass-lined drainage swales; infiltration is usually dominant stormwater treatment mechanism; infiltration occurs through turf and surface soils, providing the most opportunities for pollutant trapping before the water reaches groundwater);
- french drains or soak-aways (small source area subsurface infiltration pits, most typically used for infiltrating drainage from roofs; usually simple gravel-filled dug holes, but can be an empty perforated container);
- porous pavements or grid pavers (replace impervious pavements, overlain on a relatively thick storage layer of coarse material; may include drainage pipes to collect excess water that cannot be infiltrated into

underlying soil);

- drainage trenches (collect and infiltrate runoff from adjacent paved areas; generally long, moderately wide, and shallow in dimensions; filled with coarse gravel to provide storage);
- infiltration wells, or dry wells (deep, relatively small diameter holes allowing stormwater to be discharged to deep soil horizons, sometimes directly into saturated zones, commonly located at storm drainage inlet locations serving up to a few hectares of drainage area, with overflows discharged to storm or combined drainage system);
- percolating sewerage (conventional separate storm drainage, but with perforations through pipe or gaps between pipe segments; usually wrapped in geotextile fabric with coarse gravel used as trench backfill material);
- dry (percolating) basins (usually large storage areas typically located at end of drainage system before discharge into receiving water; commonly used as recreation facilities during dry weather; also provides infiltration through turf and surface soils).

All infiltration devices redirect runoff waters from the surface to the sub-surface environments. Therefore, they must be carefully designed using sufficient site specific information to protect the groundwater resources and to achieve the desired water quality management goals.

Groundwater Contamination Associated With Stormwater Pollutants

Nutrients

While nitrate is one of the most frequently encountered contaminants in groundwater (AWWA 1990), groundwater contamination by phosphorus has not been as widespread, or as severe. Nitrogen loadings are usually much greater than phosphorus loadings, especially from nonagricultural sources (Hampson 1986). Nitrogen occurs naturally both in the atmosphere and in the earth's soils. Natural nitrogen can lead to groundwater contamination by nitrate. As an example, in regions with relatively unweathered sedimentary deposits or loess beneath the root zone, residual exchangeable ammonium in the soil can be readily oxidized to nitrate if exposed to the correct conditions. Leaching of this naturally occurring nitrate caused groundwater contamination (with concentrations greater than 30 g/L) in non-populated and non-agricultural areas of Montana and North Dakota (Power and Schepers 1989).

Forms of nitrogen from precipitation may be either nitrate or ammonium. Atmospheric nitrate results from combustion, with the highest ambient air concentrations being downwind of power plants, major industrial areas, and major automobile activity. Atmospheric ammonium results from volatilization of ammonia from soils, fertilizers, animal wastes and vegetation (Power and Schepers 1989).

In the United States, the areas with the greatest nitrate contamination of groundwater include heavily-populated states with large dairy and poultry industries, or states having extensive agricultural irrigation. Extensively irrigated areas of the United States include the corn-growing areas of Delaware, Pennsylvania and Maryland; the vegetable growing areas of New York and the Northeast; the potato growing areas of New Jersey; the tobacco, soybean and corn growing areas of Virginia, Delaware and Maryland (Ritter, *et al.* 1989); the chicken, corn and soybean production areas in New York (Ritter, *et al.* 1991); the western Corn Belt states (Power and Schepers 1989); and the citrus, potato and grape vineyard areas in California (Schmidt and Sherman 1987).

Roadway runoff has been documented as the major source of groundwater nitrogen contamination in urban areas of Florida (Hampson 1986; Schiffer 1989; and German 1989). This occurs from both vehicular exhaust onto road surfaces and onto adjacent soils, and from roadside fertilization of landscaped areas. Roadway runoff also contains phosphorus from motor oil use and from other nutrient sources, such as bird droppings and animal remains, that has contaminated groundwaters (Schiffer 1989). Nitrate has leached from fertilizers and affected groundwaters under

various turf grasses in urban areas, including at golf courses, parks and home lawns (Petrovic 1990; Ku and Simmons 1986; and Robinson and Snyder 1991).

Leakage from sanitary sewers and septic tanks in urban areas can contribute significantly to nitrate-nitrogen contamination of the soil and groundwater (Power and Schepers 1989). Nitrate contamination of groundwater from sanitary sewage and sludge disposal has been documented in New York (Ku and Simmons 1986; and Smith and Myott 1975), California (Schmidt and Sherman 1987), Narbonne, France (Razack, *et al.* 1988), Florida (Waller, *et al.* 1987) and Delaware (Ritter, *et al.* 1989).

Elevated groundwater nitrate concentrations have been found in the heavily industrialized areas of Birmingham, UK, due to industrial area stormwater infiltration (Lloyd, *et al.* 1988; Ford and Tellam 1994). The deep-well injection of organonitrile and nitrate containing industrial wastes in Florida has also increased the groundwater nitrate concentration in parts of the Floridan aquifer (Ehrlich, *et al.* 1979a and 1979b).

Nutrient Removal Processes in Soil

Whenever nitrogen-containing compounds come into contact with soil, a potential for nitrate leaching into groundwater exists, especially in rapid-infiltration wastewater basins, stormwater infiltration devices, and in agricultural areas. Nitrate is highly soluble and will stay in solution in the percolation water, after leaving the root zone, until it reaches the groundwater. Therefore, vadose-zone sampling can be an effective tool in predicting nonpoint sources that may adversely affect groundwater (Spalding and Kitchen 1988).

Nitrogen containing compounds in urban stormwater runoff may be carried long distances before infiltration into soil and subsequent contamination of groundwater (Robinson and Snyder 1991). The amount of nitrogen available for leaching is directly related to the impervious cover in the watershed (Butler 1987). Nitrogen infiltration is controlled by soil texture and the rate and timing of water application (either through irrigation or rainfall) (Petrovic 1990; and Boggess 1975). Landfills, especially those that predate the RCRA Subtitle D Regulations, often produce significant nitrogen contamination in nearby groundwater, as demonstrated in Lee County, Florida (Boggess 1975). Studies in Broward County, Florida, found that nitrogen contamination problems can also occur in areas with older septic tanks and sanitary sewer systems (Waller, *et al.* 1987).

Nutrient leachates usually move vertically through the soil and dilute rapidly downgradient from their source. The primary factors affecting leachate movement are the layering of geologic materials, the hydraulic gradients, and the volume of the leachate discharge (Waller, *et al.* 1987; Wilde 1994).

Once the leachate is in the soil/groundwater system, decomposition by denitrification can occur, with the primary decomposition product being elemental nitrogen (Hickey and Vecchioli 1986). As an example, deep well injection of organonitriles and nitrates in a limestone aquifer acts like an anaerobic filter with nitrate respiring bacteria being the dominant microorganism. These bacteria caused an eighty percent reduction of the waste within one hundred meters of injection in the Floridan aquifer, near Pensacola (Ehrlich, *et al.* 1979b). Gold and Groffman (1993) reported groundwater leaching losses from residential lawns to be low for nitrates (typically <2mg/L), when using application rates recommended for residential lawn care.

During percolation through the soil, some nutrients are removed and the nutrient concentrations affecting the groundwater are significantly reduced. Phosphorus, in the form of soluble orthophosphate, may be either directly precipitated or chemically adsorbed onto soil surfaces through reactions with exposed iron, aluminum or calcium on solid soil surfaces (Crites 1985). Phosphorus fixation is a two-step process, sorption onto the soil solid and then conversion of the sorbed phosphorus into mineraloids or minerals. If the sorption sites are filled either with phosphate anions or another ion, phosphorus sorption will be low. The sorption of phosphorus per unit of percolation liquid decreases with each year of recharge (White and Dornbush 1988).

Downward movement of phosphorus in different soils was found to be directly related to the reactivity index measured for each soil, especially for surface-applied phosphorus fertilizer. In Washington, a difference in depth of penetration was noted, however, between sandy- and clayey-textured soils, with sandy-textured soils showing the

greater depth of penetration. If the fertilizer was surface applied, instead of sprinkler applied, and the soil was not inverted, most of the phosphorus remained within the top 5 to 7.5 cm of the surface (Lauer 1988).

If the nitrogen is not used by the plant, it will leach through the soil toward the groundwater, with some being removed in the soil prior to its reaching the aquifer. Under certain conditions, losses of dissolved nitrate and nitrite can be described by zero-order kinetics (Hampson 1986). In general, however, the process is regulated by so many limiting factors that such a simplified description is not possible. Residual nitrate concentrations were found to be highly variable in soil due to factors such as soil texture, mineralization, rainfall, irrigation, organic matter content, crop yield, nitrogen fertilizer/sludge application rate, denitrification, and soil compaction (Ferguson, *et al.* 1990b). Nitrate's flow to groundwater from stormwater infiltration is controlled by the rate and volume of infiltration, horizontal and vertical groundwater flow, the depth to the water table, and the existence of areas/channels of preferential flow. Once the nitrate has reached the groundwater, its concentration may be reduced by dispersion and diffusion with uncontaminated groundwater (Wilde 1994). The amount of ammonia volatilization is influenced by the position of the nitrogen in the soil/turf grass after application. This position is highly influenced by rainfall and/or irrigation (Bowman, *et al.* 1987, as reported by Petrovic 1990).

Phosphorus concentrations generally decrease with depth in agricultural soils because phosphorus is adsorbed to soil minerals and also precipitates readily with calcium, iron, or aluminum (Lauer 1988 and Ragone 1977). The dominant precipitation reactions are pH dependent, forming mostly iron and aluminum phosphates in acid soils and calcium phosphates under alkaline conditions. In neutral soils, the precipitation reactions are strongly rate-limited, so that the apparent solubility of the phosphate compounds is higher than under either acid or alkaline conditions (Bouwer 1985).

Pesticides

Pesticides are used in urban areas, primarily for weed and insect control in houses, along roadsides, railroad rights-of-way, in parks, on golf courses, and on private lawns (Racke and Leslie 1993). The pesticide loading in runoff water has been correlated to the amount of impervious cover and to the distance the runoff will travel prior to infiltration or decomposition, as demonstrated by Lager (1977) and confirmed in Austin, Texas by Butler, *et al.* (1987). Urban pesticide contamination of groundwater in central Florida likely resulted from municipal and homeowner use of these chemicals for pest control and their subsequent collection in stormwater runoff. Samples from the upper part of the Floridan aquifer have contained detectable amounts of diazinon, malathion, 2,4-D, ethion, methyl trithion, silvex, and 2,4,5-T (German 1989). In California, chlordane groundwater contamination has been traced to its application adjacent to residential foundations where it had been used for termite and ant control (Greene 1992). Atrazine and simazine groundwater contamination was related to their use to control weeds along roadways (Domagalski, *et al.* 1992). In Arizona, diazinon, dacthal, and dioxathion were detected in stormwater runoff entering urban dry wells that recharge the aquifer (Wilson, *et al.* 1990). Diazinon (at 30 g/L) and methyl parathion (at 10 g/L) were detected in groundwater below municipal waste treatment plants in Florida which used land spreading or well injection of wastes (Pruitt, *et al.* 1985). Gold and Groffman (1993) reported groundwater leaching losses from residential lawns to be low for dicamba and 2,4-D (<1 g/L), when using application rates recommended for residential lawn care.

In contrast, groundwater below Fresno, California, stormwater recharge basins contained only one of the organophosphorus pesticides, diazinon. None of the ten chlorinated pesticides (aldrin, chlordane, endosulfan I, endosulfan II, endosulfan sulfate, DDD-mixed isomers, DDT-mixed isomers, DDE-mixed isomers, gamma-BHC, and methoxychlor) and none of the chlorophenoxy herbicides were found (Nightingale 1987b; and Salo, *et al.* 1986).

The Technical University of Denmark (Mikkelsen, *et al.* 1996a and 1996b) has been involved in a series of tests to examine the effects of stormwater infiltration on soil and groundwater quality. They found that heavy metals and PAHs present little groundwater contamination threat, if surface infiltration systems are used. However, they express concern about pesticides which are much more mobile.

Pesticide Removal Processes in Soil

Heavy repetitive use of mobile pesticides, such as EDB, on irrigated and sandy soils likely contaminates groundwater. Fungicides and nematocides must be mobile in order to reach the target pest and hence, they generally have the highest contamination potential. Pesticide leaching depends on patterns of use, soil texture, total organic carbon content of the soil, pesticide persistence, and depth to the water table (Shirmohammadi and Knisel 1989). A pesticide leaches to groundwater when its residence time in the soil is less than the time required to remove it, or transform it to an innocuous form by chemical or biological processes. The residence time is controlled by two factors: water applied and chemical adsorption to stationary solid surfaces. Volatilization losses of soil-applied pesticides can be a significant removal mechanism for compounds having large Henry's constants (K_h), such as DBCP or EPTC (Jury, *et al.* 1983). However, for mobile compounds having low K_h values, such as atrazine, metolachlor, or alachlor, it is a negligible loss pathway compared to the leaching mechanism (Alhajjar, *et al.* 1990).

Pesticide Mobility. Estimates of pesticide mobility can be made based on the three removal mechanisms affecting organic compounds (volatilization, sorption, and solubility), as shown on Tables 1 through 3 (Armstrong and Llana 1992). Application methods and formulation state can also play a significant role in pesticide mobility. Residues of foliar-applied water soluble pesticides appear in high concentrations in runoff (Pierce and Wong 1988). Pesticide movement can be retarded or enhanced depending upon the soil conditions (Alhajjar, *et al.* 1990). Leaching is enhanced in coarse-grained soils (Domagalski and Dubrovsky 1992), but if the vadose zone contains restricting layers, pesticide movement will be slower (Sabol, *et al.* 1987). Leaching is also enhanced by flood-irrigation, in areas needing high recharge rates, and in areas with preferential flow.

Table 1. Mobility Class Definition

Class	K_d	M_i
I - Mobile	<0.1 to 1.0	0.1 to 1.0
II - Intermediate mobility	1.0 to 10.0	0.01 to 0.1
III - Low mobility	10.0 to 100.0	0.001 to 0.01
IV - Very low mobility	>100.0	<0.001

Where: K_d is the soil adsorption coefficient, mL/g
 M_i is the mobility index (ratio of pollutant's migration velocity to migration velocity of water under saturated flow).

Source: modified from Armstrong and Llana 1992.

Table 2. Organic Compound Mobility For Sandy Loam Soils

Mobile (Class I)

Organic Carbon = 0.01%

2,4-D	Dacthal
Acenaphthylene	Diazinon
Alachlor	Dicamba
Atrazine	Malathion
Cyanazine	Metolachlor

Organic Carbon = 0.1%

2,4-D	Dacthal
Alachlor	Diazinon
Atrazine	Dicamba
Cyanazine	Metolachlor

<u>Organic Carbon = 1.0%</u>	
2,4-D	Diazinon
Alachlor	Dicamba
Dacthal	
Intermediate Mobility (Class II)	
<u>Organic Carbon = 0.01%</u>	
2,3,3',4',6-Pentachlorobiphenyl	Chlordane
2,3',4',5-Tetrachlorobiphenyl	Fluoranthene
2,4'-Dichlorobiphenyl	Fluorene
2,4,4'-Trichlorobiphenyl	Methoxychlor
Anthracene	Pentachlorophenol
Benzo(a) anthracene	Phenanthrene
Bis(2-ethylhexyl) phthalate	Pyrene
<u>Organic Carbon = 0.1%</u>	
2,4'-Dichlorobiphenyl	Malathion
Acenaphthylene	Pentachlorophenol
Fluorene	
<u>Organic Carbon = 1.0%</u>	
Atrazine	Metolachlor
Cyanazine	
Low Mobility (Class III)	
<u>Organic Carbon = 0.01%</u>	
2,2',3,4,4',5,5'- heptachlorobiphenyl	Benzo(ghi) perylene
2,2',4,4',5,5'-Hexachlorobiphenyl	Benzo(k) fluoranthene
Benzo(b) fluoranthene	Chrysene
Benzo(a) pyrene	
<u>Organic Carbon = 0.1%</u>	
2,3,3',4',6-Pentachlorobiphenyl	Chlordane
2,3',4',5-Tetrachlorobiphenyl	Fluoranthene
2,4,4'-Trichlorobiphenyl	Methoxychlor
Anthracene	Phenanthrene
Benzo(a) anthracene	Pyrene
Bis(2-ethylhexyl) phthalate	
<u>Organic Carbon = 1.0%</u>	
2,4'-Dichlorobiphenyl	Malathion
Acenaphthylene	Pentachlorophenol
Fluorene	
Very Low Mobility (Class IV)	
<u>Organic Carbon = 0.01%</u>	
Indeno(1,2,3-cd)Pyrene	
<u>Organic Carbon = 0.1%</u>	
2,2',3,4,4',5,5'-Heptachlorobiphenyl	Benzo(ghi) perylene
2,2',4,4',5,5'-hexachlorobiphenyl	Benzo(k) fluoranthene
2,3,3',4',6-Pentachlorobiphenyl	Chrysene
Benzo(a) pyrene	Indeno(1,2,3-cd) pyrene
Benzo(b) fluoranthene	
<u>Organic Carbon = 1.0%</u>	
2,2',3,4,4',5,5'- heptachlorobiphenyl	Benzo(k) fluoranthene
2,2',4,4',5,5'-Hexachlorobiphenyl	Bis(2-ethylhexyl) phthalate
2,3,3',4',6-Pentachlorobiphenyl	Chlordane
2,3',4',5-Tetrachlorobiphenyl	Chrysene
2,4,4'-Trichlorobiphenyl	Fluoranthene
Anthracene	Indeno(1,2,3-cd) pyrene
Benzo(a) anthracene	Methoxychlor
Benzo(a) pyrene	Phenanthrene
Benzo(b) fluoranthene	Pyrene
Benzo(ghi) perylene	

Source: modified from Armstrong and Llena 1992.

Table 3. Organic Compound Mobility Classes For Silt Loam Soils

Mobile (Class I)

Organic Carbon = 0.01%

2,4-D	Dacthal
2,4'-Dichlorobiphenyl	Diazinon
Acenaphthylene	Dicamba
Alachlor	Malathion
Atrazine	Metolachlor
Cyanazine	

Organic Carbon = 0.1%

2,4-D	Dacthal
Alachlor	Diazinon
Atrazine	Dicamba
Cyanazine	Metolachlor

Organic Carbon = 1.0%

2,4-D	Diazinon
Alachlor	Dicamba
Dacthal	

Intermediate Mobility (Class II)

Organic Carbon = 0.01%

2,3,3',4',6-Pentachlorobiphenyl	Fluoranthene
2,3',4',5-tetrachlorobiphenyl	Fluorene
2,4,4'- trichlorobiphenyl	Methoxychlor
Anthracene	Pentachlorophenol
Benzo(a) anthracene	Phenanthrene
Bis(2-ethylhexyl) phthalate	Pyrene
Chlordane	

Organic Carbon = 0.1%

2,4'-Dichlorobiphenyl	Malathion
Acenaphthylene	Pentachlorophenol
Anthracene	Phenanthrene
Fluorene	

Organic Carbon = 1.0%

Atrazine	Malathion
Cyanazine	Metolachlor

Low Mobility (Class III)

Organic Carbon = 0.01%

2,2',3,4,4',5,5'- heptachlorobiphenyl	Benzo(ghi) perylene
2,2',4,4',5,5'-Hexachlorobiphenyl	Benzo(k) fluoranthene
Benzo(a) pyrene	Chrysene
Benzo(b) fluoranthene	Indeno(1,2,3-cd) pyrene

Organic Carbon = 0.1%

2,2',4,4',5,5'-Hexachlorobiphenyl	Bis(2-ethylhexyl) phthalate
2,3,3',4',6-Pentachlorobiphenyl	Chlordane
2,3',4',5-Tetrachlorobiphenyl	Fluoranthene
2,4,4'-Trichlorobiphenyl	Methoxychlor
Benzo(a)Anthracene	Pyrene

Organic Carbon = 1.0%

2,4'-Dichlorobiphenyl	Fluorene
Acenaphthylene	Pentachlorophenol
Anthracene	Phenanthrene

Very Low Mobility (Class IV)

Organic Carbon = 0.1%

2,2',3,4,4',5,5'-Heptachlorobiphenyl	Benzo(ghi) perylene
2,2',4,4',5,5'-hexachlorobiphenyl	Benzo(k) fluoranthene
Benzo(a) pyrene	Chrysene
Benzo(b) fluoranthene	Indeno(1,2,3-cd) pyrene

Organic Carbon = 1.0%

2,2',3,4,4',5,5'-heptachlorobiphenyl	Benzo(k) fluoranthene
2,2',4,4',5,5'-Hexachlorobiphenyl	Bis(2-ethylhexyl) phthalate
2,3,3',4',6-Pentachlorobiphenyl	Chlordane
2,3',4',5-Tetrachlorobiphenyl	Chrysene
2,4,4'-Trichlorobiphenyl	Fluoranthene
Benzo(a) pyrene	Indeno(1,2,3-cd) pyrene
Benzo(a) anthracene	Methoxychlor
Benzo(b) fluoranthene	Pyrene
Benzo(ghi) perylene	

Reference: modified from Armstrong and Llena 1992.

The greatest pesticide mobility occurs in areas with coarse-grained or sandy soils without a hardpan layer, having low clay and organic matter content and high permeability (Domagalski and Dubrovsky 1992). Structural voids, which are generally found in the surface layer of finer-textured soils rich in clay, can transmit pesticides rapidly when the voids are filled with water and the adsorbing surfaces of the soil matrix are bypassed. This preferential (bypass) flow is demonstrated in some areas where the observed mobility of the pesticide in the soil is greater than the predicted value. This flow occurs in structured, coarse-grained soils, or soils with cracks, root holes, or worm holes. It generally does not occur on continuously flooded loam soil. It likely results from unsaturated zone processes and is controlled by the mobile and immobile fractions, soil heterogeneity, and soil spatial variability. Preferential flow allows the pesticide to flow easily through the soil and bypass the area with the greatest microbial activity and degradation (Rice, *et al.* 1991; and Steenhuis, *et al.* 1988).

Pesticide transport past the root zone through the unsaturated zone depends on the lipophilic nature and other chemical characteristics of the compound, on how the compound is used in relation to climate, and irrigation practices, and on the properties of the soil and aquifer media, including hydraulic conductivity and total organic carbon (Domagalski and Dubrovsky 1992). Basic (high pH) pesticides, such as atrazine, become more mobile in soils having high pH values. Acidic pesticides ionize, depending on their dissociation constants, to form cationic and anionic species, and under neutral soil conditions (pH of 5 to 9), the anionic form predominates. Since these anions are negatively charged, they do not adsorb onto the negatively charged clay mineral surfaces. Similarly, acidic pesticides tend to be more mobile in neutral soils (Pierce and Wong 1988). Pesticide movement will often slow as the depth increases in the vadose zone, allowing more soil contact time. However, the soil still may not be able to completely remove the pesticide from the water due to reduced biodegradation activity deeper in the soil (Bouwer 1987).

Pesticide mobility comparisons have been performed for atrazine, metolachlor and alachlor in the same soil type and it was found that alachlor mobility > metolachlor >> atrazine (Alhajjar, *et al.* 1990), with faster movement generally occurring in sandy loam soils versus loam soils (Krawchuk and Webster 1987).

Restricted pesticide usage on coastal golf courses has been recommended by some U.S. regulatory agencies. The slower moving pesticides were recommended provided they were used in accordance with the approved manufacture's label instructions. These included the fungicides Iprodione and Triadimefon, the insecticides Isofenphos and Chlorpyrifos and the herbicide Glyphosate. Others were recommended against, even when used in accordance with the label's instructions. These included the fungicides Anilazine, Benomyl, Chlorothalonil and Maneb and the herbicides Dicamba and Dacthal. No insecticides were on the "banned list" (Horsley and Moser 1990).

Solubility. Leaching of the less water soluble compounds is determined by the sorption ability of the chemicals to the soil particles, especially the colloids. The sorption ability of the pesticide determines whether it will remain in

solution until it reaches the groundwater (Pierce and Wong 1988). Adsorption of a pesticide to the soil slows, or stops, its travel with the percolating water and possibly prevents its contamination of the groundwater (Bouwer 1987). In general, pesticides with low water solubilities (indicated by high octanol-water partitioning coefficients) are less mobile. Also, in general, basic and nonionic water soluble pesticides are lost in greater amounts in surface runoff than acidic and nonionic, low to moderate water soluble, pesticides with less traveling through the soil toward the groundwater (Pierce and Wong 1988).

Adsorption and desorption control the movement of pesticides in groundwater (Sabatini and Austin 1988). Modeling of pesticide movement using physical non-equilibrium expressions for mass transfer and diffusion most closely mimics the actual movement in soil (Pierce and Wong 1988).

Decomposition. Pesticides decompose in soil and water, but the total decomposition time can range from days to years. Decomposition and dispersion rates in the soil depend upon many factors, including pH, temperature, light, humidity, air movement, compound volatility, soil type, persistence/half-life and microbiological activity (Ku and Simmons 1986). Historically, pesticides were thought to adsorb to the soil during recharge, with decomposition then occurring from the sorbed sites. However, literature half-lives generally apply to surface soils and do not account for the reduced microbial activity found deep in the vadose zone (Bouwer 1987).

Pesticides with long (>30 day) half lives can show considerable leaching. An order of magnitude difference in half-life results in a five to ten-fold difference in percolation loss (Knisel and Leonard 1989). Organophosphate pesticides are less persistent than organochlorine pesticides, but they also are not strongly adsorbed by the sediment and are likely to leach into the vadose zone, and possibly the groundwater (Norberg-King, *et al.* 1991).

As demonstrated in Central Florida and on Long Island, New York, sediment analysis in recharge basins show sediment with significant organic content, indicating that basin storage and recharge may effectively remove a large percentage of the pesticides (Schiffer 1989; and Ku and Simmons 1986). Most organophosphate and carbamate insecticides are regarded as nonpersistent, but they have been found in older, organic soils used for vegetable production and in the surrounding drainage systems (Norberg-King, *et al.* 1991). Studies of recharge basins in Nassau and Suffolk Counties on Long Island, New York, showed that the DDT found in each basin's sediment correlated well with the basin's age and showed that DDT can survive in recharge basins for many years (Seaburn and Aronson 1974).

Other Organic Compounds

Many organic compounds are naturally occurring, although many of concern in groundwater contamination investigations are man-made. Sources of organic contaminants include natural sources, landfills, leaky sewerage systems, highway runoff, agricultural runoff, urban stormwater runoff, and other urban and industrial sources and practices. Organic compounds occur naturally from decomposing animal wastes, leaf litter, vegetation, and soil organisms (Reichenbaugh 1977).

Concentrations of organic compounds in urban runoff are related to land use, geographic location and traffic volume (Hampson 1986). These compounds result from gasoline and oil drippings, tire residuals and vehicular exhaust material (Seaburn and Aronson 1974; Hampson 1986). The primary source is from the use of petroleum products, such as lubrication oils, fuels, and combustion emissions (Schiffer 1989). The organic compounds on many street surfaces consists of: cellulose, tannins, lignins, grease and oil, automobile exhaust hydrocarbons, carbohydrates and animal droppings (Hampson 1986). Toluene and 2,4-dimethyl phenol are also found in urban runoff and are used in making asphalt (German 1992). Polynuclear aromatic hydrocarbons (PAHs) are also commonly found in urban runoff and result from combustion processes, and include fluoranthene, pyrene, anthracene, and chrysene (German 1989; Greene 1992).

In Florida, organic compounds found in runoff were attenuated in the soil, with only one priority pollutant (bis(2-ethylhexyl) phthalate) being detected in the Floridan aquifer as a result of stormwater runoff (German 1989). In Pima County, Arizona, base/neutral compounds appeared in groundwater from residential areas, while phenols in the groundwater were noted only near a commercial site. Groundwater from a commercial site, also in Pima

County, has been contaminated with ethylbenzene and toluene. Perched groundwater samples from residential sites showed the presence of toluene, xylene, and phenol (Wilson, *et al.* 1990). On Long Island, New York, benzene (groundwater concentrations of 2 to 3 g/L); bis(2-ethylhexyl) phthalate (5 to 13 g/L); chloroform (2 to 3 g/L); methylene chloride (stormwater concentration of 230 g/L and groundwater concentrations of 6 to 20 g/L); toluene (groundwater concentrations of 3 to 5 g/L); 1,1,1-trichloroethane (2 to 23 g/L); p-chloro-m-cresol (79 g/L); 2,4-dimethyl phenol (96 g/L); and 4-nitrophenol (58 g/L) were detected in groundwater beneath stormwater recharge basins (Ku and Simmons 1986).

Organic compounds occasionally found in runoff at three stormwater infiltration sites in Maryland included benzene, trichlorofluoromethane, 1,2-dichloroethane, 1,2-dibromoethylene, toluene, and methylene blue active substances (MBAS). Only MBAS's were found consistently and in elevated concentrations beneath the infiltration devices. The other organic compounds found in runoff were removed either in the device or in the vadose zone. Although specific organic compounds were not detected in concentrations above the detection limits in the groundwater beneath and downgradient of the infiltration device, the dissolved organic carbon (DOC) concentration in the groundwater affected by infiltration was greater than that in the native groundwater (Wilde 1994).

Industrial areas contribute heavily to the organic compound load that could potentially leach to the groundwater. Surface impoundments may be used to contain industrial wastes, deep well injection may be used to dispose of water, and stormwater runoff may collect organics as it passes over an industrial site. Phenols and the PAHs benzo(a)anthracene, chrysene, anthracene and benzo(b)fluoranthene, have been found in groundwater near an industrial site in Pima County, Arizona. The phenols are primarily used as disinfectants and as wood preservatives and were present in the stormwater runoff, although they were significantly reduced in concentration by the time they reached the groundwater (generally less than 50 g/L). At an Arizona recharge site, the groundwater has higher concentrations of trichloroethylene, tetrachloroethylene, and pentachloroanisole, than the inflow water, indicating past industrial contamination (Bouwer *et al.* 1984).

In Birmingham, UK, groundwater contamination resulted from hydrocarbon oil and volatile chlorinated solvent use. The metals-related industries have contributed significant amounts of trichloroethylene (groundwater concentrations of up to 4.9 mg/L have been noted) to the groundwater in this area, and since trichloroethylene has been replaced by 1,1,1-tri-chloroethane in industry, 1,1,1-trichloroethane contamination is beginning to occur. The other organic compound to show up in significant concentrations in Birmingham is perchloroethylene, a solvent used primarily in the dry cleaning laundry industry (Lloyd, *et al.* 1988). On the left bank of the Danube, the petrochemical refinery Slovnaft has contributed to groundwater contamination by leaking oil during tanker loading and unloading (Marton and Mohler 1988).

Soil Removal Processes

Most organics are reduced in concentration during percolation through the soil, although they may still be detectable in the groundwater. Groundwater contamination from organics, like from other pollutants, occurs readily in areas with pervious soils, such as sand and gravel, and where the water table is near the land surface (Troutman, *et al.* 1984). Based on septic tank effluent studies, sand seems to be more effective than limestone in filtering the organic material (Schneider, *et al.* 1987). In coastal areas and valleys, direct interaction of groundwater and surface water will result in groundwater contamination if the surface water is contaminated (Troutman, *et al.* 1984). Organic removal from the soil and recharge water can occur by one or more methods: volatilization, sorption, and/or degradation (Crites 1985; and Nellor, *et al.* 1985).

Volatilization. The rate of volatilization is controlled by the compound's physical and chemical properties; its concentration; the soil's sorptive characteristics; the soil-water content; air movement; temperature; and the soil's diffusion ability. Volatilization can occur both during application and from soil sites after infiltration. Volatilization during application is controlled by the compound's physical and chemical characteristics, atmospheric conditions, and application method (Crites 1985) and has been measured by observing the reduction in the organic concentration across an infiltration basin.

Volatilization from sorbed sites of soils is a function of: transfer of the organic compound from the soil's sorbed sites to the solution, movement from the solution to the air trapped in the soil, and advection and diffusion of the compound in the soil air to the atmosphere. The extent of each process depends on the compound's solubility, its concentration gradient in the soil, and proximity of the molecule of interest to the soil surface (Crites 1985).

Volatile organic compounds are rarely found in stormwater recharge basins (<2.4 g/L), dry wells (<175 g/L), or the vadose zone or groundwater below the basins (<4 g/L), as indicated by studies in Fresno, California, by Nightingale (1987b) and in Pima County, Arizona by Wilson, *et al.* (1990).

Sorption. Sorption is affected by many soil-water-compound system characteristics, including: sorbate shape/configuration (including structure and position of functional groups and presence and degree of molecular saturation); sorbate chemical characteristics (including acidity or basicity; water solubility; charge distribution; polarity and polarizability); and sorbent nature (including mineralogical composition, organic matter content and cation exchange capacity (CEC) (Crites 1985) with the clay and particulate organic matter content controlling the sorption (Bouwer *et al.* 1984).

Hydrophobic sorption onto organic matter was found to limit the mobility of less soluble base/neutral and acid extractable compounds through organic soils and the vadose zone in Orlando, Florida (German 1989). The degree of removal in soil of nonhalogenated organic compounds is greater than that of the halogenated organics (Bouwer, *et al.* 1984). Benzene, toluene and xylene were found in the soils and in the perched groundwater in Arizona and, as these compounds are relatively soluble, they may percolate easily through the vadose zone. The toluene concentration in the perched groundwater was 54 g/L, while 3.7 g/L of toluene was found in the main groundwater body at depth (Wilson, *et al.* 1990).

Sorption is typically only a temporary removal mechanism. A study in Florida has shown that organic solubilization can occur for several storms following dry periods. However, extended periods of complete aeration of bottom sediments may be counter-productive when trying to reduce organic compound concentrations (Hampson 1986).

Degradation and Decomposition. The third process for organic compound attenuation is chemical or biological degradation. Examples of chemical degradation processes include hydrolysis and photodegradation. However, most of the trace organic compound removal is the result of biological degradation (Smith and Myott 1975). Many organics can be degraded by microorganisms, at least partially, but others cannot. Temperature, pH, moisture content, cation exchange capacity, and air availability may limit the microbial degradation potential for even the most degradable organics (Crites 1985).

Conditions in a thick, aerobic, unsaturated zone provide a good environment for detergent concentration reductions through biochemical degradation and adsorption (Smith and Myott 1975). Halogenated one- and two-carbon aliphatic compounds are biotransformed under methanogenic, but not aerobic, conditions (Bouwer, *et al.* 1984). The rate of breakdown of chlorinated hydrocarbons in the soil increases with temperature, water content, and organic matter content (Bouwer 1987). Nonhalogenated hydrocarbons decreased fifty to ninety percent during percolation through the soil, with concentrations in the renovated water being detectable, but near, or below, the detection limit. The halogenated organic compounds generally decreased to a lesser extent during percolation.

The chlorinated aromatics are relatively refractory and mobile in the ground and have lesser concentration decreases than nonchlorinated aromatic hydrocarbons. Significant reductions of TOC concentrations occurred during the first several meters of soil percolation and a gradual decrease in TOC concentration occurred with longer underground travel times at the Phoenix, Arizona, 23rd Ave. recharge project site (Bouwer, *et al.* 1984).

Pretreatment of the recharge water in Arizona by chlorination resulted in higher chloroform concentrations and in the formation of three brominated trihalomethanes (Bouwer, *et al.* 1984). Ponds and lakes affected by stormwater runoff have a high potential for the formation of trihalomethanes (THMs) with chlorination because of the precursor organics existing in the stormwater. Factors which affect THM formation include the chlorine contact

time, pH, and temperature. THM formation in stormwater is enhanced by the presence of algae, bacteria, and humic substances (Wanielista, *et al.* 1991).

In Arizona, partial degradation of the chlorinated benzenes occurs during percolation through the aerobic zone, but the poor overall removal efficiency of chlorinated aromatics probably results from their lack of degradation under anoxic conditions. In general, infiltration and percolation through the soil has the effect of dampening concentration fluctuations and eliminating occasional extreme values (Bouwer, *et al.* 1984).

Pathogens

Viruses were detected in groundwater on Long Island at sites where stormwater recharge basins were located less than thirty-five feet above the water table (Vaughn, *et al.* 1978). At other locations, viruses are likely removed from the percolation water by either inactivation and/or adsorption. As expected, the highest bacteria and viruses groundwater concentrations occur when the water table is near the land surface (Boggess 1975).

Soil Removal Processes

During land application or burial of sewage sludge, pathogens may leach from the sludge and contaminate groundwater, although most are removed in the soil during percolation (Gerba and Haas 1988). The factors that affect the survival of enteric bacteria and viruses in the soil include pH, antagonism from soil microflora, moisture content, temperature, sunlight, and organic matter (Crites 1985). In general, drying of the soil will kill both bacteria and viruses.

Virus Removal Processes. Viral adsorption is promoted by increasing cation concentration, decreasing pH and decreasing soluble organics (EPA 1992) and is controlled by both the efficiency of short-term virus retention and the long-term behavior of viruses in the soil (EPA 1992 and Crites 1985).

The downward movement and distribution of viruses are controlled by convection and sorption and hydraulic dispersion mechanisms. Since the movement of viruses through soil to groundwater occurs in the liquid phase and involves water movement and associated suspended virus particles, the distribution of viruses between the adsorbed and liquid phases determines the viral mass available for movement.

The distribution (or sorption) of virus particles in the soil matrix is largely due to electrostatic double layer interactions and Van der Waals forces. Adsorption of viruses in soil is rapid and reversible and can be adequately described by an equilibrium (linear Freundlich isotherm) expression (Tim and Mostaghim 1991). Viral adsorption by the soil does not necessarily result in virus inactivation and has been shown to be reversible after a change in soil conditions, such as the ionic environment (Jansons, *et al.* 1989b; and Vaughn, *et al.* 1978). Also, once the virus reaches the groundwater, it can travel laterally through the aquifer until it is either adsorbed or inactivated (Vaughn, *et al.* 1978).

Enterovirus survival in groundwater is highly variable and is influenced by a number of factors, including: virus type, pH, temperature, dissolved oxygen concentration, and microbial antagonism (Jansons, *et al.* 1989b; and Tim and Mostaghim 1991). The two most important attributes of viruses that permit their long-term survival in the environment are their structure and very small size. These characteristics permit virus occlusion and protection within colloid-size particles. Viruses in wastewater applied to the soil are integral parts of submicron particles and are small enough to move with the applied water to the groundwater (Wellings 1988).

Dissolved oxygen is a significant factor in loss of virus activity in groundwater, possibly because direct oxidation of components of the virus capsid inactivates the virus or, as with temperature, the level of dissolved oxygen influences the activity of antagonistic microorganisms. At temperatures below 4°C, microorganisms can survive for months or even years, whereas at higher temperatures, inactivation or dieoff occurs rapidly. Decreasing pH promotes virus adsorption and results in shorter survival times, both of the virus and of the antagonistic bacteria. High levels of organic matter appear to shield viruses from adsorption (Trewick 1985). Virus inactivation in the subsurface environment can be described by a first order decay reaction (Tim and Mostaghim 1991). It is difficult

to describe a soil that will remove all viruses effectively, as different soil types affect each virus differently (Jansons, *et al.* 1989a).

Enteric viruses are more resistant to environmental factors than enteric bacteria and they exhibit longer survival times in natural waters. They can occur in potable and marine waters in the absence of fecal coliforms. Enteroviruses are more resistant to commonly used disinfectants than are indicator bacteria, and can occur in groundwater in the absence of indicator bacteria (Marzouk, *et al.* 1979).

Removal of Bacterial Pathogens. The major bacterial removal mechanisms in soil are straining at the soil surface and at intergrain contacts, sedimentation, and sorption by soil particles (Crites 1985). Potable water use requires continuous disinfection to prevent disease outbreaks in the population served (Craun 1979).

Factors such as temperature, pH, metal concentration, nutrient availability and other environmental characteristics affect the ability of a bacterial colony to survive in the water or soil (Ku and Simmons 1986). Once the microorganisms are retained in the soil, their survival depends upon the sunlight exposure, oxidation rates, desiccation and antagonism from the established soil microbial population (Crites 1985).

Bacteria survive longer in acid soils and when large amounts of organic matter are present. Bacteria and larger organisms in wastewater are usually removed during percolation through a short distance of soil (EPA 1992). Ehrlich, *et al.* (1979a) found that the logarithm of the coliform bacteria density decreases linearly with distance from the recharge well when recharging using deep well injection.

In general, enteric bacteria survive in soil between two and three months, although survival times up to five years have been documented (Crites 1985). *E. coli* can survive and multiply on trapped organic matter and *Salmonella* and *Shigella* have survived for forty-four days and twenty-four days, respectively, at a recharge site in Israel (Goldshmid 1974).

Metals

Metals may be a problem when infiltrating stormwater when using a rapid infiltration system (Crites 1985), such as a dry well. Most metals have very low solubilities at the pHs found in most natural waters and they are readily removed by either sedimentation or sorption removal processes (Hampson 1986). Many are also filtered, or otherwise sorbed, in the surface layers of soils in infiltrating devices when using surface infiltration.

Nickel, chromium, and zinc concentrations exceeded the regulatory limits in the soil below a recharge area at an Arizona commercial site. However, only manganese was present at an elevated concentration in the groundwater at a residential site (Wilson, *et al.* 1990). At a site in Lee County, Florida, groundwater near an unlined landfill had elevated iron concentrations due to landfill leachate. The leachate also may have increased the groundwater manganese concentrations (Boggess 1975). In New York, cesspool leachates have elevated the concentrations of boron and barium in the shallow groundwaters of the Magothy Aquifer (Smith and Myott 1975).

Boron concentrations were found to be high in groundwaters below industrialized areas in Birmingham, UK, as it is used in many metal-related industries. The presence of high concentrations of aluminum, cadmium, manganese, and titanium was also noted in the groundwaters near metals industries in the Birmingham industrial area (Ford and Tellam 1994).

Hütter and Remmler (1996) described a groundwater monitoring plan, including monitoring wells that were established during the construction of an infiltration trench for stormwater disposal in Dortmund, Germany. The worst case problem expected is with zinc, if the infiltration water has a pH value of 4. Mull (1996) stated that traffic areas are the third most important source of groundwater contamination in Germany (after abandoned industrial sites and leaky sewers). The most important contaminants are chlorinated hydrocarbons, sulfate, organic compounds, and nitrates. However, heavy metals are generally not an important groundwater contaminant because of their affinity for soils.

Metal Removal Processes in Soils. The interaction of surface water and groundwater has resulted in selenium contamination of groundwater in Wyoming (Peterson 1988). Sandy soils provided minimal removal of boron and nickel, while the percolate water had no cobalt (Crites 1985). In general, studies of recharge basins receiving large metal loads show that most of the heavy metals are removed either in the basin sediment or in the vadose zone (Ku and Simmons 1986; and Hampson 1986).

Removal of metals by soil may be accomplished through one of several processes, including: (1) soil surface association, (2) precipitation, (3) occlusion with other precipitates, (4) solid-state diffusion into soil minerals, and (5) biologic system or residue incorporation (Crites 1985). Most of these removal processes are pH-dependent, as is the solubility of most metals. In general, the solubility of a metal increases as the solution's pH decreases (Wilde 1994).

Although the concentrations of many trace metals were reduced through sorption/ion exchange to bottom materials and/or native sediments, elevated concentrations (greater than background) of these metals were found in groundwater beneath and downgradient of three infiltration devices in Maryland. The greater-than-expected metal concentrations could have resulted from fluctuations in pH and/or dissolved oxygen; unfavorable conditions for sorption, oxidation, ion exchange, or precipitation in the pond and/or unsaturated zone; or significantly greater flow rates through the unsaturated zone (Wilde 1994).

Adsorption, Cation Exchange, Organic Complexation, and Chelation. Dissolved metal ions are removed from stormwater during infiltration mostly by adsorption onto the near-surface particles in the vadose zone, while the particulate metals are filtered out at the soil surface (Ku and Simmons 1986). Studies of dissolved lead in recharge ponds in Jacksonville, Florida, found that allowing the ponds to go dry between storms was counterproductive to the removal of lead from the water during recharge (Hampson 1986). Apparently, the adsorption bonds were weakened during the drying period. Studies in Fresno, California, recharge basins found that lead, zinc, cadmium, and copper accumulated at the soil surface with little downward movement over five years. However, the microtopographic features, such as small depressions and basin inlet and outlet locations, influenced the metal's distribution in the soil (Nightingale 1987a).

Similarities in water quality between the runoff water and the groundwater show that there is downward movement of copper and iron in sandy and loamy soils. However, the other metals of concern (arsenic, nickel, and lead) did not significantly move downward through the soil to the groundwater. The exception to this was some downward movement of lead with the percolation water in the sandy soils of Fresno stormwater recharge basins (Nightingale 1987b).

In soils, heavy metals enter into general cation exchange reactions with clay and organic matter and into chelation reactions with organic molecules. As the organic molecules are decomposed, the metals become free to react with iron and aluminum hydroxides, calcium, and other compounds. These new compounds are immobilized in the soil profile. The immobilization reactions are more pronounced at high pH and in an aerobic environment. Boron is adsorbed to iron and aluminum hydroxide coatings on clay minerals, to iron and aluminum oxides, to micaceous clay minerals, and to magnesium hydroxide coatings on weathering surfaces of ferro-magnesium minerals. In sandy soils and quartz, boron is not significantly immobilized (Bouwer 1985). Interactions of certain metals with phosphorus can form either soluble or insoluble complexes. The type of clay mineral also affects heavy metal adsorption. The higher the cation exchange capacity (CEC) of the soil, more metallic cations will be bound to the soil (Nightingale 1987a). A soil's CEC is pH-dependent; therefore, the ion exchange ability of a soil to remove metals from solution is pH-dependent.

Organic complexation of a metal may enhance the metal's ability to move freely to the groundwater. Organic complexes often are stable and uncharged or negatively charged. Because of their negative or neutral charge, they are not attracted to negatively-charged adsorption/ion exchange sites and are not easily removed from solution (Wilde 1994).

Most of the heavy metals in stormwater are associated with particulates and can be readily strained-out through filtration as the water infiltrates into the soil (Pitt, *et al.* 1995). Therefore, these direct physical removal mechanisms are likely more important than chemical removal mechanisms for most heavy metals.

Precipitation. In central Florida, the dissolved iron concentrations in recharge water are much greater than in the groundwater, indicating that dissolved iron ions are being removed from the recharge water during percolation. The decrease of dissolved iron concentrations resulted from precipitation of iron or complexation into other nonsoluble species (Schiffer 1989). Iron and manganese transformations in groundwater are controlled by both the oxidation-reduction conditions and the acid-alkali balance in the water. The migration and diffusion of iron (Fe^{+2}) and oxygen is rather slow in water, but is accelerated by iron bacteria. The amount of iron oxidized by iron bacteria is ten, or even hundreds of times, greater than that oxidized by the chemical reaction alone (Bao-rui 1988). In central Florida, zinc, which is more soluble than iron, was commonly found in higher concentrations in groundwater than iron (Schiffer 1989). Iron and manganese oxidation may lower the pH of a water because the oxidation reactions add acidity (as the H^+ ion) to the water. A lower pH may cause an increase in the dissolved metal concentrations in the water. Concurrent decreases in pH and Eh in the unsaturated zone may have increased the mobility of manganese beneath one stormwater impoundment in Maryland (Wilde 1994).

Observed Heavy Metal Groundwater Contamination Associated with Stormwater Infiltration

In dry recharge wells in Arizona, manganese was the only metal that was mobile in the vadose zone and was the only metal to show up in the groundwater at elevated concentrations (Wilson, *et al.* 1990).

Aluminum mobility is governed by pH, amount of stormwater infiltration, horizontal and vertical groundwater flow, depth to water table, and existence of channels for preferential flow. Aluminum is soluble above pH 9.0. Cadmium solubility/mobility is governed by pH, redox potential, biological uptake, and solubilities of carbonates and sulfides. If sulfide is present, cadmium is nearly immobile.

Hydrophilic and negatively-charged, or neutrally-charged, complexes are not likely to be retarded in the vadose zone. Copper is soluble at high pH values (greater pH than zinc or nickel) and its solubility is affected by complexation with iron and other ligands and by coprecipitation by oxides. If chromium entering an infiltration device is negatively charged, or is bound in a stable, negatively-charged compound, it may move easily through the vadose zone (Wilde 1994).

Chromium was detected in groundwater in Maryland beneath stormwater infiltration devices and was not removed from solution by sorption to the sediments in the device, or to the vadose zone sediments. Lead forms compounds with hydroxides, carbonates, sulfides, and sulfates, all of which have a low solubility. Lead also is removed from solution by binding with organic matter, coprecipitating with manganese oxides, and sorbing to organic and inorganic substrates. Zinc mobility is limited by high pH, the partial pressure of carbon dioxide in the solution, and the presence of sulfide. Aqueous zinc can be reduced by coprecipitation with other minerals, cation-exchange, biochemical activities, and complexation and sorption to organic and inorganic substrates (Wilde 1994).

In Maryland, groundwater pH below and downgradient of stormwater infiltration devices was less than 5.0 and tended to keep the metals in solution or solubilize metals attached to particulates. Concentrations of cadmium, chromium, and lead exceeded U.S. Environmental Protection Agency's Maximum Contaminant Levels in some groundwater samples and concentrations of barium, copper, nickel, strontium, and zinc in downgradient groundwater were often greater than their concentrations in native groundwater (Wilde 1994).

Experiments in Orlando, Florida (Harper 1988), concerning metal mobility in soil and its resultant stability have led to the ranking of metals in order of attenuation from recharge water (dependent on water and soil chemistry):

zinc (most mobile) > lead > cadmium > manganese > copper > iron > chromium > nickel > aluminum (least mobile).

Other studies of metal pollutant mobility in soil have led to the generation of mobility classes, as shown in Tables 4 and 5 (Armstrong and Llena 1992).

Table 4. Metal Mobility

Inorganic Pollutant	Mobility Class* for:	
	Sandy Loam	Silt Loam
Arsenic	III and IV	III and IV
Cadmium	III	III and IV
Chromium	III and IV	II and III
Copper	IV	IV
Lead	IV	IV
Nickel	III	III
Zinc	III	III and IV

* I: mobile
 II: Intermediate mobility
 III: low mobility
 IV: very low mobility

Source: modified from Armstrong and Llena 1992.

Table 5. Metal Removal Mechanisms In Soil

Element	Principal Forms in Soil Solution	Principal Removal Mechanisms
Arsenic	AsO_4^{-3}	Strong associations with clay fractions of soil.
Barium	Ba^{+2}	Precipitation and sorption onto metal oxides and hydroxides.
Cadmium	Cd^{+2} complexes chelates	Ion exchange, sorption, and precipitation.
Chromium	Cr^{+3} Cr^{+6} $\text{Cr}_2\text{O}_9^{-2}$ CrO_4^{-2}	Sorption, precipitation, and ion exchange.
Cobalt	Co^{+2} Co^{+3}	Surface sorption, surface complex ion formation, lattice penetration, ion exchange, chelation, and precipitation.
Copper	Cu^{+2} $\text{Cu}(\text{OH})^+$ anionic chelates	Surface sorption, surface complex ion formation, ion exchange, and chelation.
Iron	Fe^{+2} Fe^{+3}	Surface sorption and surface complex ion.

	polymeric forms	
Lead	Pb ⁺²	Surface sorption, ion exchange, chelation, and precipitation.
Manganese	Mn ⁺²	Surface sorption, surface complex ion formation, ion exchange, and chelation, precipitation.
Mercury	Hg ⁺ HgS HgCl ₃ ⁻ HgCl ₄ ⁻² CH ₃ Hg ⁺ Hg ⁺²	Volatilization, sorption, and chemical and microbial degradation.
Nickel	Ni ⁺²	Surface sorption, ion exchange, and chelation.
Selenium	SeO ₃ ⁻² SeO ₄ ⁻²	Ferric-oxide selenite complexation.
Silver	Ag ⁺	Precipitation
Zinc	Zn ⁺² complexes chelates	Surface sorption, surface complex ion formation, lattice penetration, ion exchange, chelation, and precipitation.

Source: modified from Crites 1985.

Mobility Classes for Heavy Metals. Table 5 summarizes the principal removal mechanisms in the soil for each metal (Crites 1985). The surface water heavy metal concentrations were the most significant variables in predicting the concentrations of the heavy metals in the groundwater (Harper 1988).

Dissolved Minerals

Some dissolved minerals are of concern in groundwater contamination. Increasing chloride concentrations in groundwater have been used as an indicator of early groundwater contamination in Great Britain (Lloyd, *et al.* 1988). When using rapid infiltration for recharge, inorganic dissolved solids are of concern and include chloride, sulfate, and sodium (Crites 1985).

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. In Arizona, stormwater infiltration in dry wells dissolves natural salts in the vadose zone which are then carried to the groundwater (Wilson, *et al.* 1990).

Salt Removal Processes in Soils. 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. Use of lower salinity water as recharge water at the Leaky Acres stormwater recharge facility in Fresno, California, was shown to decrease the salt concentrations in the groundwater (Nightingale and Bianchi 1977). Reduction in the pH of groundwater, such as would result from nitrification and the biodegradation of carbonaceous substances, resulted in the dissolution of soil minerals and subsequent increases in the total dissolved solids concentrations and the hardness of groundwater at the Whittier Narrows site in Los Angeles County, California (Nellor, *et al.* 1985). This effect was noted in Florida during the deep-well injection of acidic, high-oxygen demanding industrial waste. Initially, neutralization of the waste

occurred through solution of the calcium carbonate in the limestone. Later, the calcium concentration in the groundwater was found to be elevated and the pH decreased, but the effects have still been confined to the lower strata of the Floridan aquifer (Goolsby 1972). The higher the salt concentration of the soil solution, the higher the soil hydraulic conductivity will be for a given sodium adsorption ratio (SAR) (Bouwer and Idelovitch 1987). Schmidt and Sherman (1987) found a direct relationship between concentrations of groundwater nitrates and salts.

Solubility Equilibrium of Salts. For chloride, sodium, and sulfate, reductions in concentrations entering the recharge system are likely accounted for by differences in seasonal precipitation, with a higher loading in the summer than in the winter. Changes in the groundwater concentration reflect these loading differences (Hampson 1986). Potassium exchanges with hydrogen ions on the clay during percolation. Other exchanges cause the calcium and magnesium concentrations to be much greater than had been predicted (Ragone 1977). Deep-well injection waters have shown an increase in alkalinity and bicarbonate concentrations, reflecting the mineralization of the organic compounds. Dissolved calcium and bicarbonate are the primary products of limestone dissolution. Many parameters in natural groundwater systems are controlled, or are influenced, by the calcium carbonate equilibrium system.

Removal. Soil is not very effective at removing most salts. Depth of dissolved mineral penetration in soil has been studied at a site with a shallow, unconfined aquifer (Close 1987). This study found that sulfate and potassium concentrations decreased with depth, while sodium, calcium, bicarbonate and chloride concentrations increased with depth in the soil. The dissolution of the aquifer material may be the source of many of the chloride, bicarbonate, calcium, and sodium ions.

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. The cations sodium, potassium, calcium, and magnesium appeared in a shallow aquifer three to six months after the source water was applied to the soil (Higgins 1984).

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 the 3-year study, indicating that groundwater systems are not easily purged of conservative contaminants, even if the groundwater flow rate is relatively high. Sodium and/or calcium concentrations also were constantly elevated in the groundwater beneath and downgradient of the infiltration devices (Wilde 1994).

Summary and Recommendations

Table 6 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. However, the Groundwater Recharge Committee of the National Academy of Science (Andelman, *et al.* 1994) examined risks associated with consuming contaminated groundwater.

Table 6. Groundwater Contamination Potential for Stormwater Pollutants

	Compounds	Mobility (worst case: sandy/low organic soils)	Abundance in storm- water	Fraction filterable	Contamination potential for surface infiltr. and no pretreatment	Contamination potential for surface infiltr. with sediment- ation	Contamination potential for sub-surface injection 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
	-BHC	intermediate	moderate	likely low	low/moderate	low	low/moderate
	(lindane)						
	malathion	mobile	low	likely low	low	low	low
	atrazine	mobile	low	likely low	low	low	low
Other organics	chlordane	intermediate	moderate	very low	moderate	low	moderate
	diazinon	mobile	low	likely low	low	low	low
	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	high	very low	moderate	low	moderate
	pyrene	intermediate	high	high	moderate	moderate	high
	Pathogens	enteroviruses	mobile	likely present	high	high	high
<i>Shigella</i>		low/inter.	likely present	moderate	low/moderate	low/moderate	high
<i>Pseudomonas aeruginosa</i>		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 zinc	very low/low/very low	moderate/high	very low/high	low/low	low/low	moderate/high
Salts	chloride	mobile	seasonally high	high	high	high	high

Source: Pitt, *et al.* 1994

General causes of concern indicating probable groundwater contamination potential are:

- high mobility (low sorption potential) in the vadose zone,
- high abundance (high concentrations and high detection frequencies) in stormwater, and
- high soluble fractions (small fraction associated with particulates which would have little removal potential using conventional stormwater sedimentation controls) in the stormwater.

The contamination potential is defined as the most critical rating of the influencing factors. As an example, if no pretreatment was to be used before percolation through surface soils, the mobility and abundance criteria are most important. The filterable fraction is not as important as no treatment is being used, based on the assumption that physical removal of particulates is the most important removal process for stormwater. If a compound was mobile, but was in low abundance (such as for VOCs), then the groundwater contamination potential would be low because the concentrations are low to begin with. However, if the compound was mobile and was also in high abundance (such as for sodium chloride, in certain conditions), then the groundwater contamination potential would be high.

If sedimentation pretreatment is to be used before surface infiltration, then some of the 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.

If subsurface injection (with minimal pretreatment) is to be used, then only the abundance factor is significant. If the pollutant is present in adverse concentrations, it will likely have an adverse effect on the groundwater. Attenuation through the vadose zone (as reflected in the mobility factor) may be insignificant as the water would bypass the vadose zone for a deep injection well. Similarly, the filterable fraction of the pollutant would be less important as no treatment is conducted before disposal. However, pollutants that are mostly in filterable forms would likely have a greater effect on the groundwater quality than those mostly associated with particulates.

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. However, if subsurface infiltration/injection was used instead of surface percolation, both the mobility and the abundance factors would be important, with some regard given to the filterable fraction information for operational considerations.

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 had a high organic content, then most of the organic compounds 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 would likely be greater for critical source areas (especially vehicle service areas) and critical land uses (especially manufacturing industrial areas). Other, more detailed methods are possible to assess the potential problems caused by stormwater infiltration, such as proposed by Martinelli and Alfakih (1998).

The stormwater pollutants of most concern (those that may have the greatest potential 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. If the stormwater nitrate concentration was high, then the groundwater contamination potential would likely also 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.

other organics: 1,3-dichlorobenzene 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, but unlikely for most other 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, especially 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.

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.

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.

The control of these compounds will require a varied approach, including source area controls, end-of-pipe controls, and pollution prevention. All dry-weather flows should be diverted from infiltration devices because of their potentially high concentrations of soluble heavy metals, pesticides, and pathogens. Similarly, all runoff from manufacturing industrial areas should also be diverted from infiltration devices because of their relatively high concentrations of soluble toxicants. Combined sewer overflows should also be diverted because of sanitary sewage contamination. In areas of extensive snow and ice, winter snowmelt and early spring runoff should also be diverted from infiltration devices.

All other runoff should include pretreatment using sedimentation processes before infiltration, to both minimize groundwater contamination and to prolong the life of the infiltration device (if needed). This pretreatment can take the form of grass filters, sediment sumps, wet detention ponds, etc., depending on the runoff volume to be treated and other site specific factors. Pollution prevention can also play an important role in minimizing groundwater contamination problems, including reducing the use of galvanized metals, pesticides, and fertilizers in critical areas. The use of specialized treatment devices can also play an important role in treating runoff from critical source areas before these more contaminated flows commingle with cleaner runoff from other areas. Sophisticated treatment schemes, especially the use of chemical processes or disinfection, may not be warranted, except in special cases, especially considering the potential of forming harmful treatment by-products (such as THMs and soluble aluminum).

The use of surface percolation devices (such as grass swales and percolation ponds) that have a substantial depth of underlying soils above the groundwater, is preferable to using subsurface infiltration devices (such as dry wells, trenches or French drains, and especially injection wells), unless the runoff water is known to be relatively free of pollutants. Surface devices are able to take greater advantage of natural soil pollutant removal processes. However, unless all percolation devices are carefully designed and maintained, they may not function properly and may lead to premature hydraulic failure or contamination of the groundwater.

Site Suitability Criteria

Introduction

There are many removal processes and mechanisms in soils that reduce the likelihood of stormwater pollutants from reaching and contaminating groundwaters. However, not all pollutants can be reduced sufficiently (such as chlorides from de-icing operations during the winter, or nitrates, if present in high concentrations). The prior discussion on potential groundwater impacts associated with stormwater infiltration reviews many of the issues involved. The following list shows some of the principal removal mechanisms for some ionic forms of heavy metals:

Cadmium and chromium: Ion exchange, sorption, and precipitation

Copper: Surface sorption, surface complex ion formation, ion exchange, and chelation

Lead: Surface sorption, ion exchange, chelation, and precipitation

Mercury: Volatilization, sorption, and other chemical and microbial processes

Zinc: Surface sorption, surface complex ion formation, lattice penetration, ion exchange, chelation, and

precipitation

Other pollutants, including organics, nutrients, bacteria, and others, all “rely” on a variety of removal mechanisms in the soil.

Particulate forms of the pollutants are mostly removed by filtration processes in the soil, although colloidal forms do behave in a complex manner. Generally, it is assumed that for a well graded filter, where all grains are close to the same diameter, and generally spherical in shape, particulates in the water about 1/3 of the diameter of the filter grains, and larger, will be trapped. For poorly graded natural soils, the interstitial spaces between the larger material is mostly filled with smaller soil particles, making it difficult to identify the critical size. In addition, as filtration proceeds, the trapped particulates form their own reduced interstitial spaces, resulting in improved filtration (but at slower rates). Table 7 lists some of the standard soil texture classifications along with the expected particulates that could be physically trapped initially.

Table 7. Soil Grain Sizes and Approximate Sizes of Filtered Particulates Trapped

Name	Size Range (mm)	Particulates Trapped (μm)
sand	0.050-2	15 to 650
silt	0.002-0.049	1 to 15
clay	< 0.002	<1

Although clay could trap the smallest stormwater particulates, very little water would be able to be infiltrated in these soils. Sand, loamy sand, sandy loam, and loam soils are the preferred soils for infiltration devices. These soils have from 40 to 100% sand, 50 to 100% silt, and no more than 20% clay. With some silt and minimal clay in the soil (the loams), soils would be able to trap much of the stormwater particulates. Indeed, this has been observed at long-term stormwater infiltration sites (see prior discussion on groundwater impacts associated with stormwater infiltration).

Critical Areas of Concern when Selecting Stormwater Infiltration Areas

Pitt, *et al.* (1994, 1996, and 1999) summarized many information sources on the groundwater contamination potential of stormwater infiltration. Much of the following discussion is from these prior sources.

The most critical source areas are those that have more contaminated flows than other areas. Besides the obvious CSOs and construction site runoff sources, these mostly include manufacturing industrial areas, plus areas having large amounts of automobile activity. Tables 8 and 9 show some source area runoff quality data indicating the variety of important source areas. Adequate pretreatment options were previously described and could be suitable for most commercial areas having large automobile activity. However, the other source areas, including manufacturing industrial areas, would require much more careful evaluation before infiltration should be allowed.

Table 8. Heavy Metal Source Area Observations (Pitt, *et al.* 1995)

Toxicant	Highest median conc. (g/L)	Source	Highest observed conc. (g/L)	Source
Cadmium	8	vehicle service area runoff	220	street runoff
Chromium	100	landscaped area runoff	510	roof runoff
Copper	160	urban receiving water	1250	street runoff
Lead	75	CSO	330	storage area runoff
Nickel	40	parking area runoff	130	landscaped area runoff
Zinc	100	roof runoff	1580	roof runoff

Table 9. Toxic Organic Source Area Observations (Pitt, *et al.* 1995)

Toxicant	Maximum (g/L)	Detection Frequency (%)	Significant Sources
----------	---------------	-------------------------	---------------------

Benzo (a) anthracene	60	12	gasoline, wood preservative
benzo (b) fluoranthene	226	17	gasoline, motor oils
benzo (k) fluoranthene	221	17	gasoline, bitumen, oils
Benzo (a) pyrene	300	17	asphalt, gasoline, oils
Fluoranthene	128	23	oils, gasoline, wood preservative
Naphthalene	296	13	coal tar, gasoline, insecticides
Phenanthrene	69	10	oils, gasoline, coal tar
Pyrene	102	19	oils, gasoline, bitumen, coal tar, wood preservative
Chlordane	2.2	13	insecticide
Butyl benzyl phthalate	128	12	plasticizer
Bis (2-chloroethyl) ether	204	14	fumigant, solvents, insecticides, paints, lacquers, varnishes
Bis (2-chloroisopropyl) ether	217	14	pesticides
1,3-Dichlorobenzene	120	23	pesticides

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 widely 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 1 and 2, 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 2).

Phosphorus retention (phosphate: Figure 3; 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.

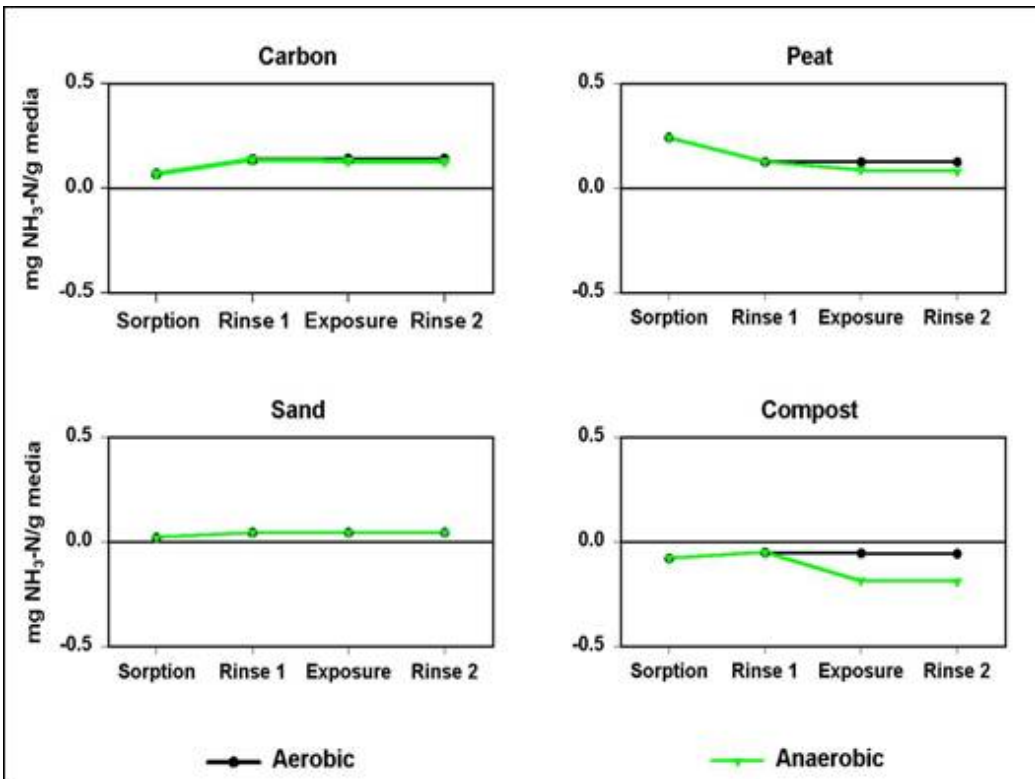


Figure 1. Behavior of ammonia-nitrogen under aerobic and anaerobic conditions.

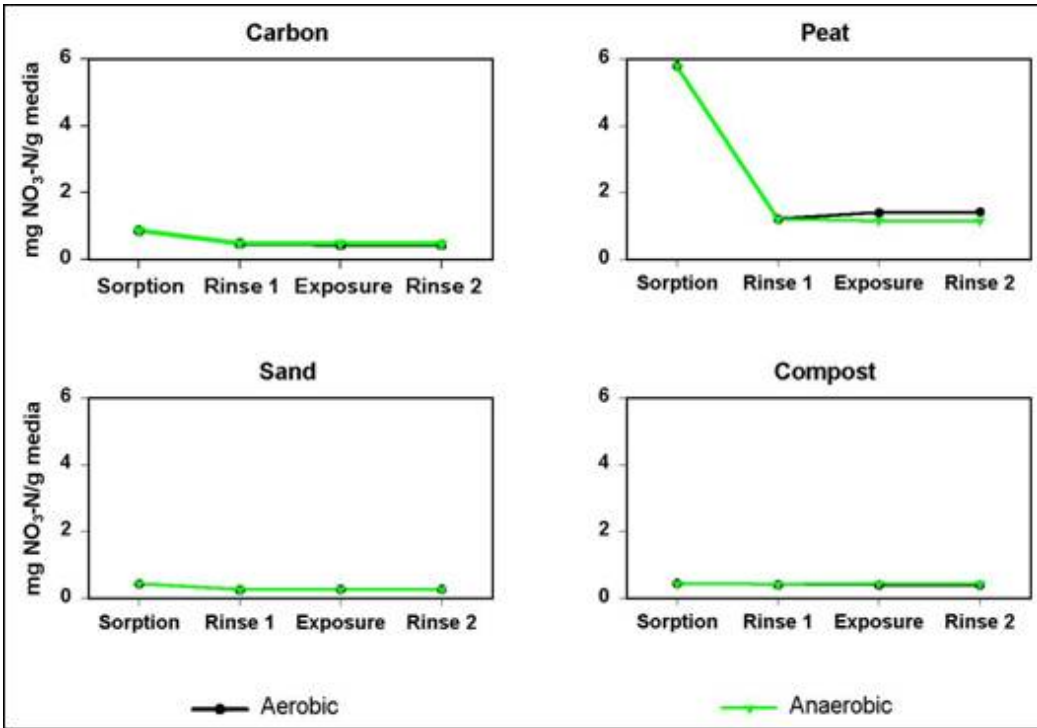


Figure 2. Behavior of nitrate-nitrogen under aerobic and anaerobic conditions.

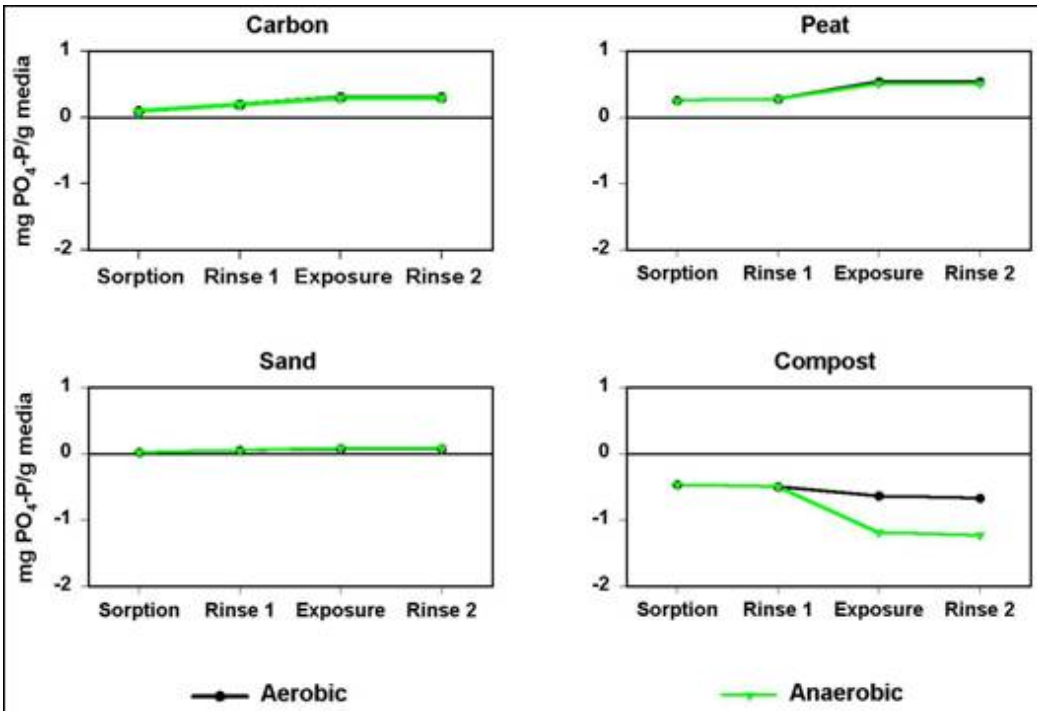


Figure 3. Behavior of phosphate under aerobic and anaerobic conditions.

The results for the heavy metals are shown in Figures 4, 5, 6, and 7 (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 5) 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.

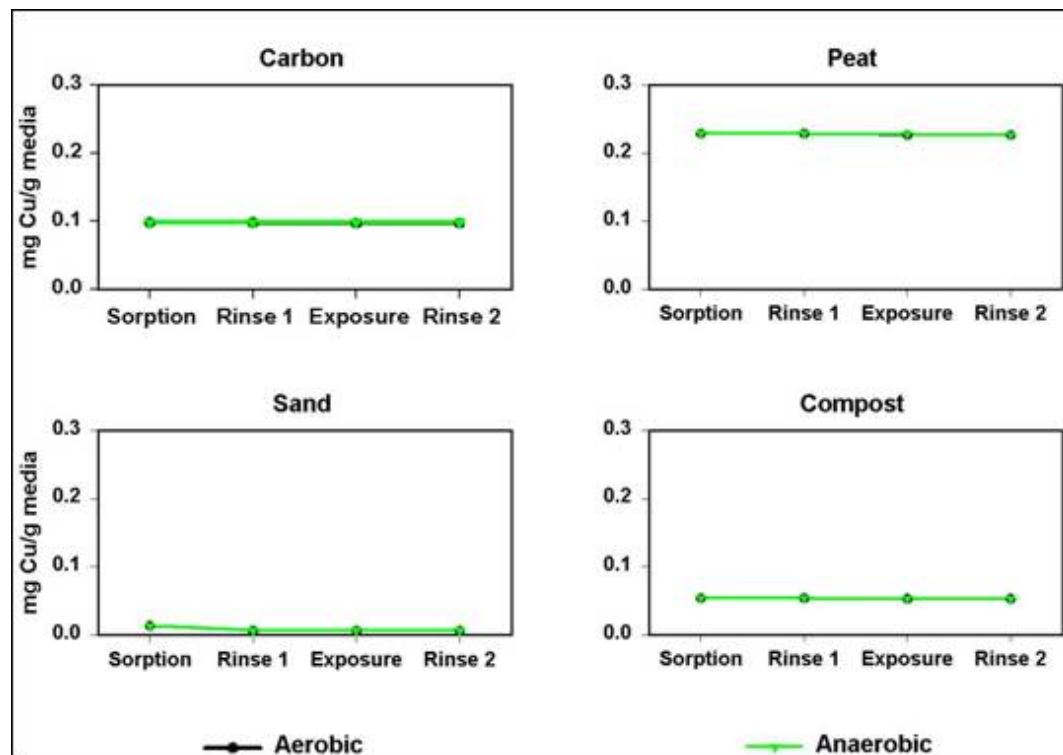


Figure 4. Behavior of copper under aerobic and anaerobic conditions.

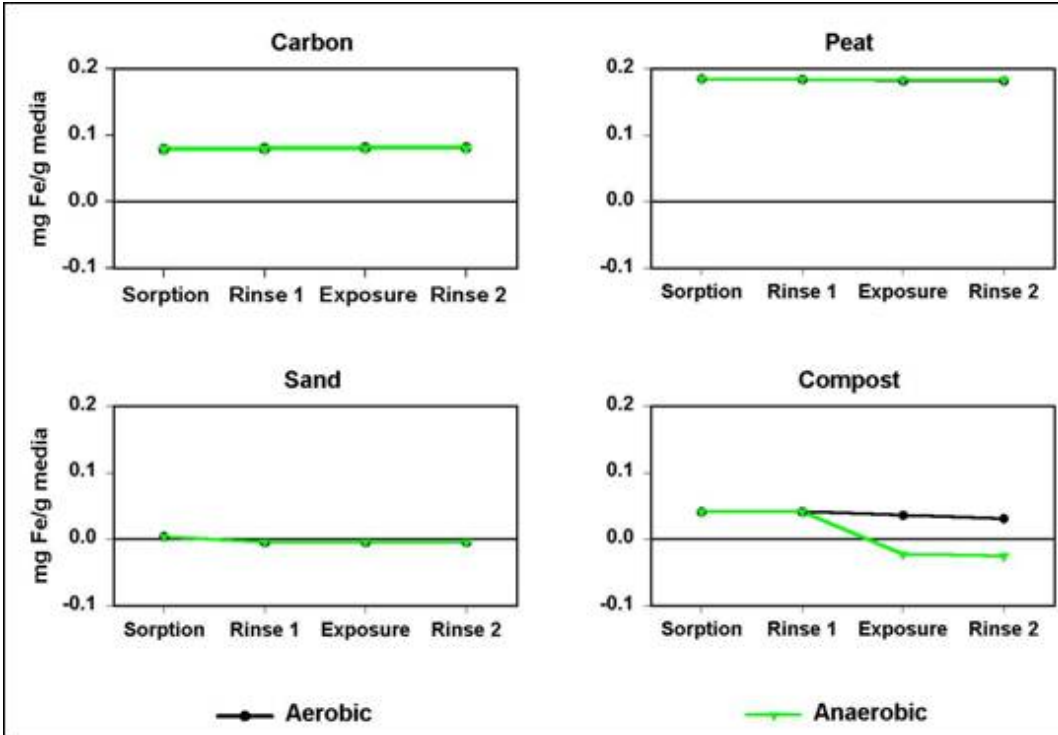


Figure 5. Behavior of iron under aerobic and anaerobic conditions.

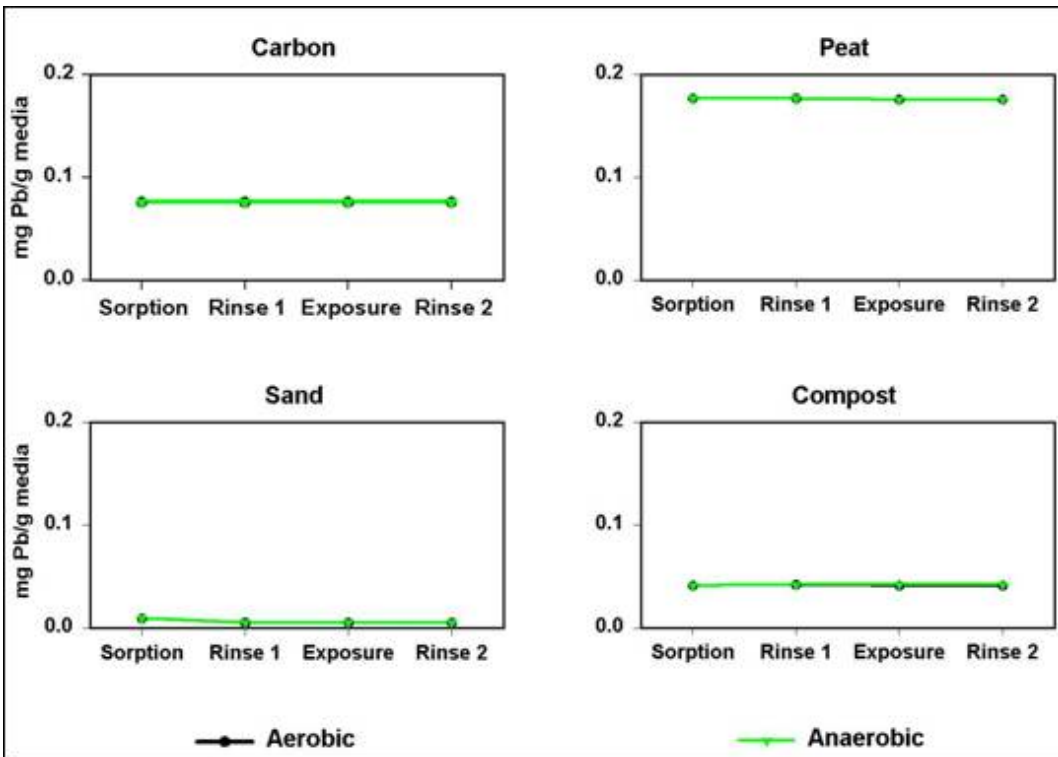


Figure 6. Behavior of lead under aerobic and anaerobic conditions.

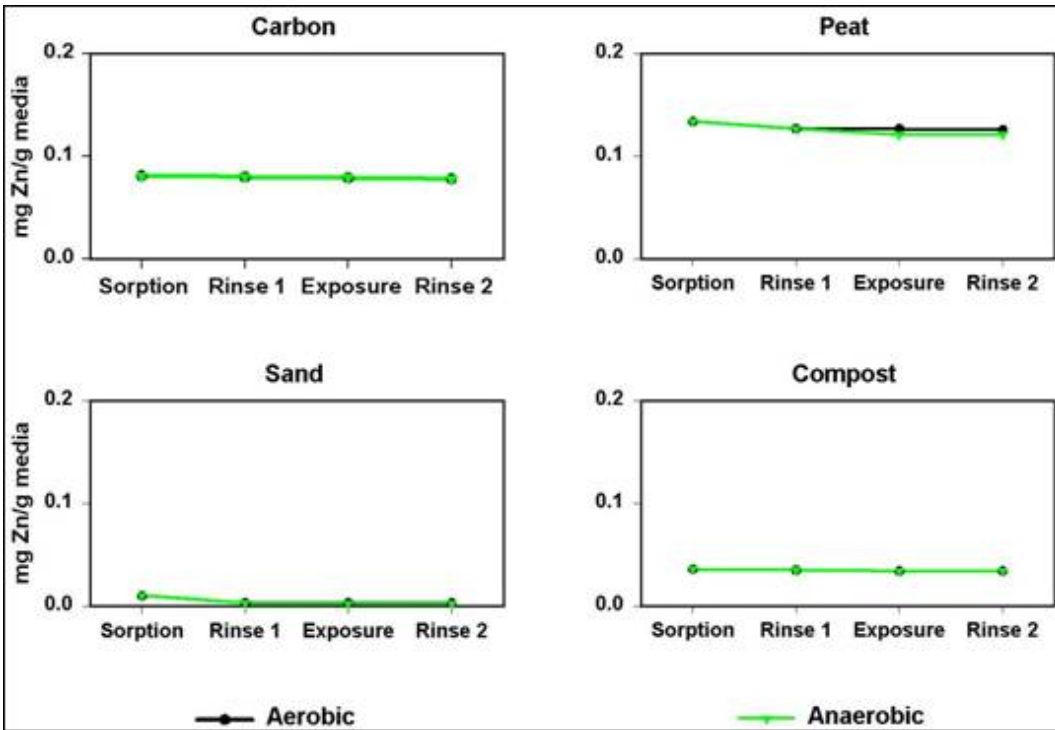


Figure 7. Behavior of zinc under aerobic and anaerobic conditions.

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 10 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 Figure 8.

Table 10. 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.

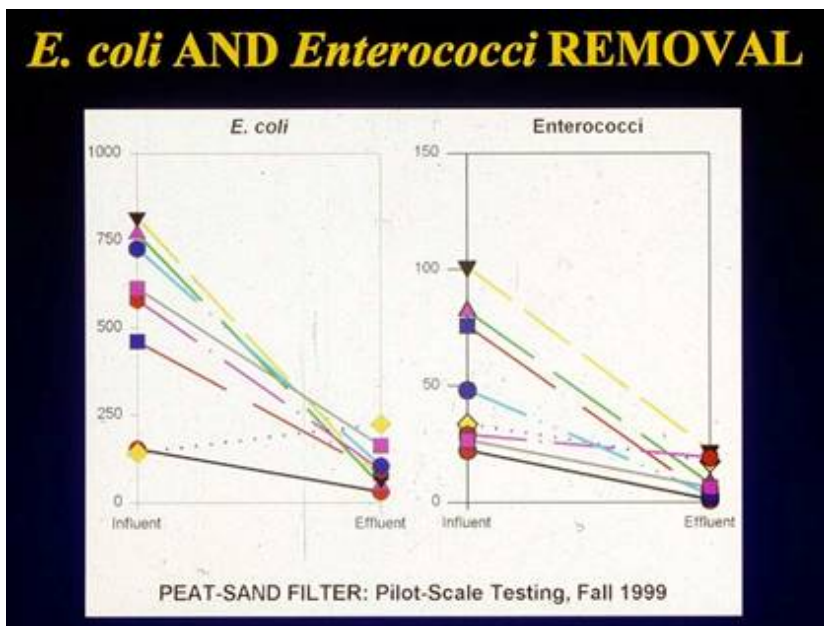


Figure 8. Bacteria control in a test soil column.

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

Constituent (mg/L, unless noted)	Soil-only plots		Percent reductions in average concentrations after infiltration
	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 ($\geq 75\%$):

Al, Zn

Moderate reductions (25 to 74%):

NH₄, Si, PO₄, Fe, Mn

Minimal reductions, or increases (<24%):

TP, TN, Cl, K, Na, NO₃, SO₄, 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.

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 triethylenetetramine 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 CaCO₃, 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 m², 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 m², 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 little decrease 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/cm³),
- 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 m²):

$$0.5 \text{ m}^3 \times \frac{1.6 \text{ g}}{\text{cm}^3} \times \frac{(100 \text{ cm})^3}{\text{m}^3} \times \frac{200 \text{ meq}}{100 \text{ g}} = 1,600,000 \text{ meq}$$

- total cation content of a years worth of runoff (per 30 m²):

$$30m^2 \times \frac{0.85m}{year} \times \frac{(1000L)}{m^3} \times \frac{1meq}{L} = \frac{25,500meq}{year}$$

- 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 9 and 10 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 11 show copper sorbed and ions desorbed for compost.

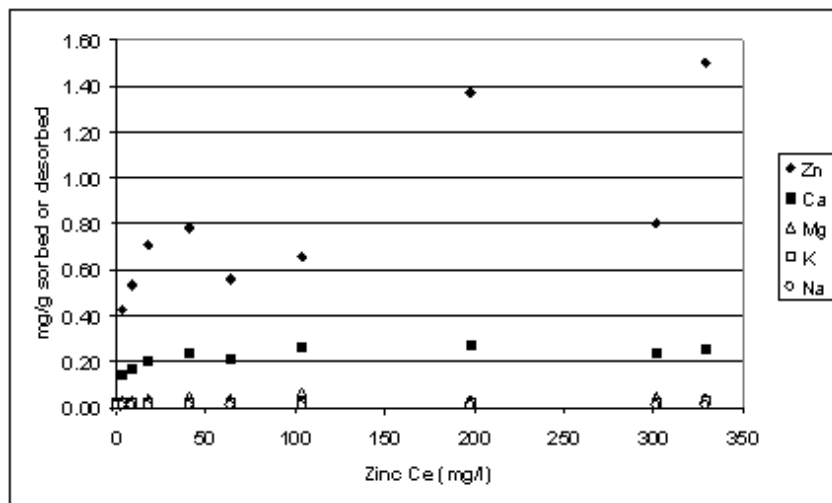


Figure 9. Zinc Sorbed and Major Ions Desorbed during Batch Equilibrium Tests for Zinc onto Peat-Sand

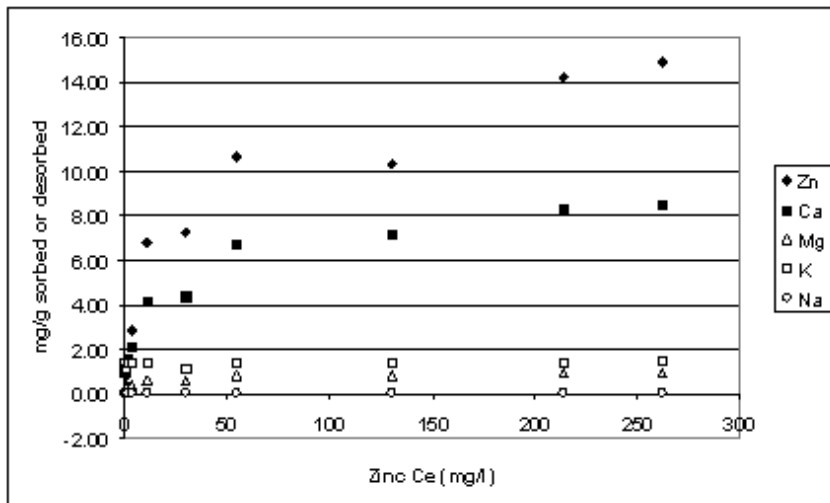


Figure 10. Zinc Sorbed and Major Ions Desorbed during Batch Equilibrium Tests for Zinc onto Compost

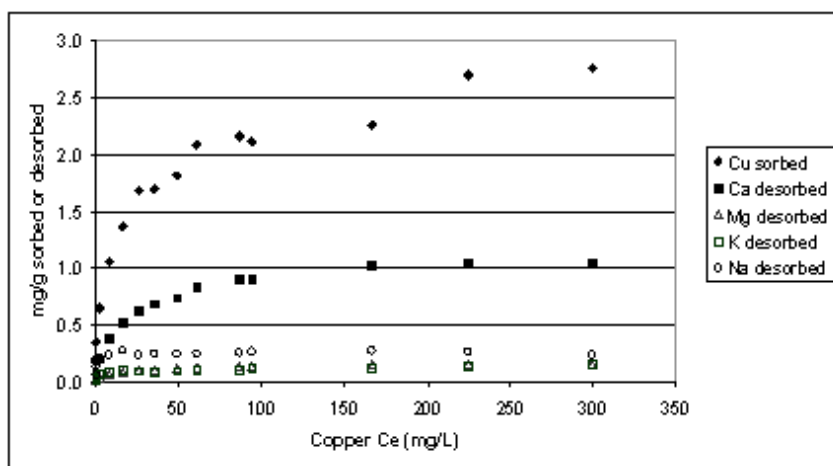


Figure 11. Copper Sorbed and Major Ions Desorbed during Batch Equilibrium Tests for Copper onto Compost

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 12 and 13 show the fraction of the initial metal concentrations (Ct/Co) remaining in solution verses time for the three final media.

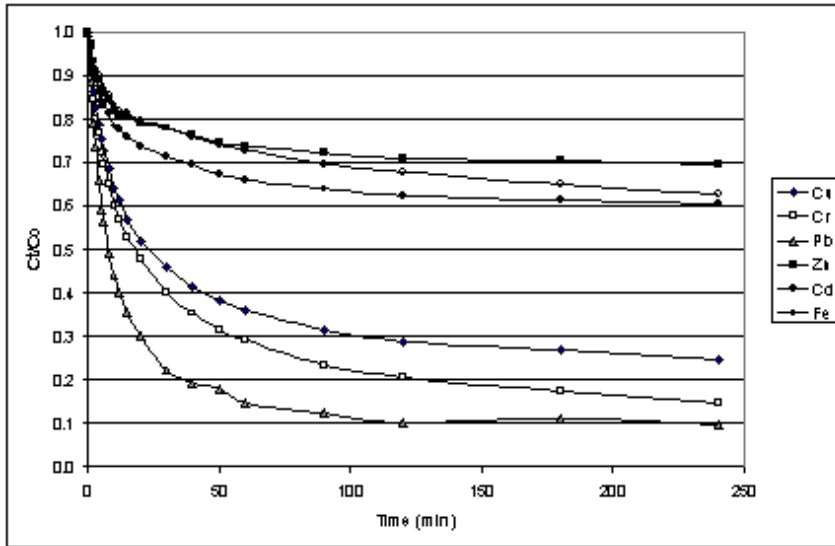


Figure 12. Fraction of Initial Metal Concentration Remaining verses Time for Metals onto Peat-Sand for a Mixed Metal Solution.

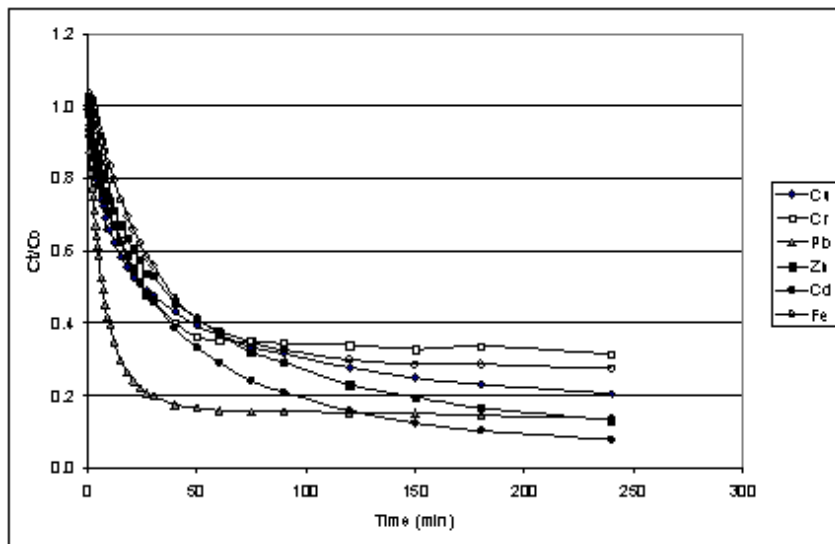


Figure 13. Fraction of Initial Metal Concentration Remaining verses Time for Metals onto Compost for a Mixed Metal Solution.

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 repeat the test results noted above.

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 begins, 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.

Compacted Urban Soils and their Effects on Infiltration and Bioretention Systems

Introduction and Summary

Prior research by Pitt (1987) examined runoff losses from paved and roofed surfaces in urban areas and showed significant losses at these surfaces during the small and moderate sized events of most interest for water quality evaluations. However, Pitt and Durrans (1995) also examined runoff and pavement seepage on highway pavements and found that very little surface runoff entered typical highway pavement. During earlier research, it was also found that disturbed urban soils do not behave as indicated by most stormwater models. Additional tests were therefore conducted to investigate detailed infiltration behavior of disturbed urban soils.

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 bioretention devices can be jeopardized. Basic infiltration measurements in disturbed urban soils were conducted during an EPA-sponsored project by Pitt, *et al* (1999a), along with examining 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 conducted 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. The following discussion is a summary of this recently collected information and it is hoped that it will prove useful to both stormwater practice designers and to modelers.

The role of urban soils in stormwater management cannot be under-estimated. 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, potential groundwater contamination questions arise. Prior EPA-funded research, an updated book, and a more recent review paper (Pitt, *et al.* 1994, 1996 and 1999b) discuss the potential for this problem. This material shows that it is possible to incorporate many stormwater infiltration options in urban areas, as long as suitable care is taken. Infiltration controls should especially be considered in residential areas where the runoff is relatively uncontaminated and surface infiltration can typically be applied. Manufacturing industrial areas and subsurface injection should normally be excluded from stormwater infiltration consideration, in contrast.

Over the past few years, we have conducted several sets of tests, 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, as expected. Sandy soils are better able to withstand compaction, although their infiltration rates are still significantly reduced.

A previous EPA report (Pitt 1999a) describes the results from a series of tests that have examined how the infiltration capacity of compacted soils can be recovered through the use of soil amendments (such as composts). This 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 research 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.

The NRCS (2001), especially in New Jersey, have also been active in investigating problems associated with urban soils during land development.

Alternative stormwater management options can be examined using the Source Loading and Management Model (WinSLAMM) and this soil information. The use of bioretention controls, such as roof gardens for example, can result in almost complete removal of roof runoff from the surface runoff component. It must be recognized that matching pre-development runoff characteristics through stormwater controls at the time of development may not be possible. Certainly, the careful use of different types of infiltration and bioretention controls, especially in low and medium density developments, are more likely to meet pre-development conditions than if these controls are not used. Accurate hydrologic modeling and correct design of these practices that consider the unique features of urban soils will help in minimizing many types of urban receiving water problems.

Areas have increased runoff after development due to a number of reasons. The most important cause is usually the increased amount of pavement and roof areas. However, as noted in this paper, urban soils also undergo major modifications that also result in increased runoff. These soil modifications may mostly affect infiltration (as described in the following paper sections), but other soil changes also occur. Specifically, reductions in the organic content of the surface soil layers and removal of plants will reduce the evapotranspiration (ET) losses and contribute to increases in runoff. This is especially important in areas where surface soils are relatively shallow and located above impermeable layers (such as the glacial till in the Seattle area, the location of our research on amended soils that was conducted to increase the ET rates of urban soils, Harrison, *et al.* 1997 and Pitt, *et al.* 1999a).

The soil compaction during construction and use likely causes most of the reduced infiltration capacity of urban soils. In addition, many more subtle changes will also reduce infiltration, such as the replacement of native plants which typically have much deeper root systems with shallow-rooted grasses. Many of these subtle changes contribute to the variations in the measured infiltration rates noted during these experiments reported in this paper. The removal of the native surface soils results in the removal of organic matter, mature and deep-rooted plants, and the soils themselves, often exposing a deeper soil material that is much less able to allow infiltration or evapotranspiration.

Infiltration Mechanisms

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 greater than the infiltration rate. The infiltration rate typically decreases during periods of rainfall excess. Storage capacity is recovered when the drainage from the vadose zone is faster than 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 transmitted away, so this transmission rate affects the infiltration rate during longer events. The depletion of available storage capacity in the soil affects the transmission and drainage rates. The storage capacity of soils depends on the soil thickness, porosity, and the soil-water content. Many factors, such as 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 was much 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 several factors: the decreased area of exposed soils, removal of surface soils and exposing subsurface soils, 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 devices, although compaction from heavy traffic is also a recognized problem. More recently, "bioretention" 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 bioretention devices also allow modifications of the soil with amendments.

Prior Infiltration Measurements in Disturbed Urban Soils

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	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 Infiltrimeters 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 moisture meter and in the laboratory using standard soil-moisture content methods. Saturated conditions occurred for most soils when the soil-moisture content exceeded about 20%.

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 (DICKY-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 14 and 15 are 3D plots of the field infiltration data, illustrating the effects of soil-moisture and compaction, for both sands and clays. Four general conditions were observed to be statistically unique, as listed on Table 12. 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.

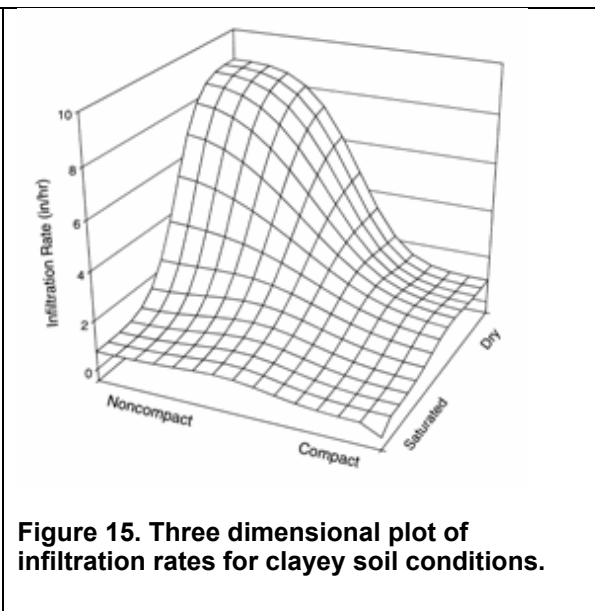
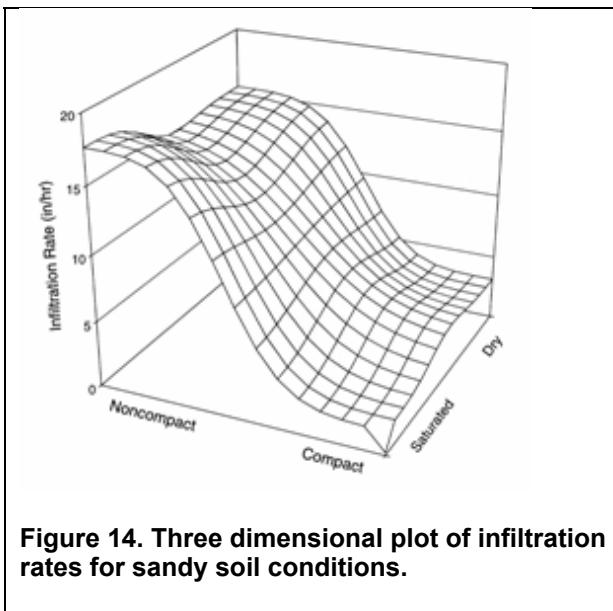


Table 12. Infiltration Rates for Significant Groupings of Soil Texture, Soil-Water Content, and Compaction Conditions

Group	Number of tests	Average infiltration rate (in/hr)	COV
noncompacted sandy soils	36	13	0.4
compact sandy soils	39	1.4	1.3
noncompacted and dry clayey soils	18	9.8	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 13 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.

Figures 16 through 19 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.

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:

Table 13. Soil Infiltration Rates for Different Categories and Storm Durations (all rate values are in inches per hour)

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

- 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 14 (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.

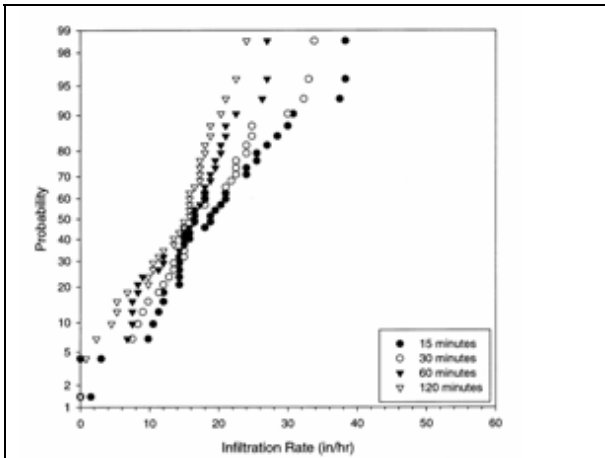


Figure 16. Probability plots for infiltration measurements for noncompacted, sandy soil, conditions.

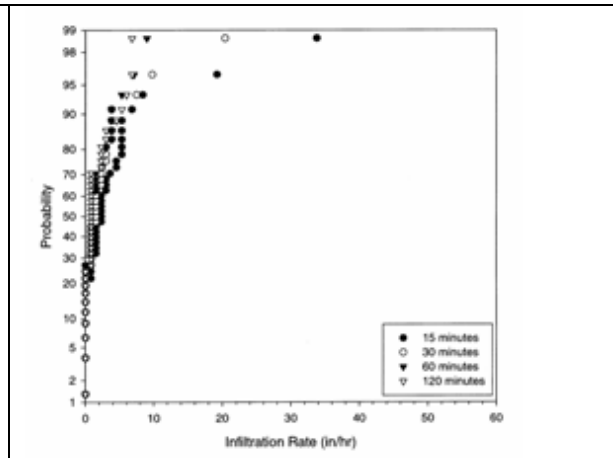


Figure 17. Probability plots for infiltration measurements for compacted, sandy soil, conditions.

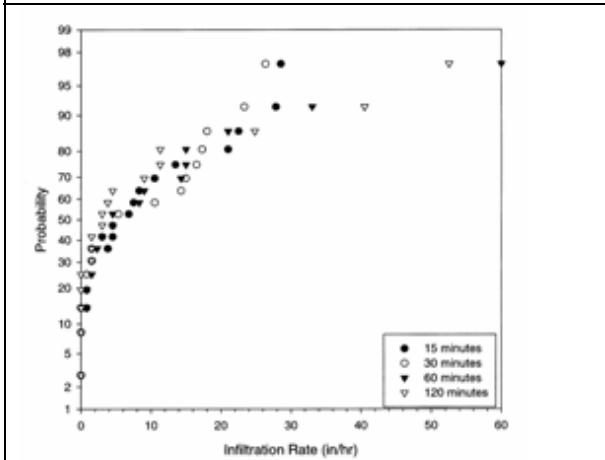


Figure 18. Probability plots for infiltration measurements for dry-noncompacted, clayey soil,

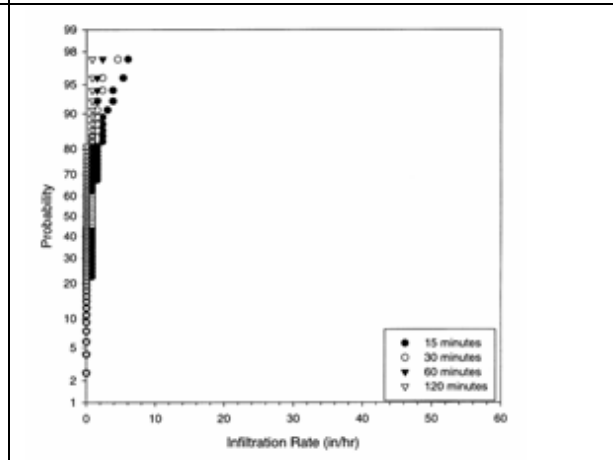


Figure 19. Probability plots for infiltration measurements for wet-noncompacted, dry-

conditions.	compacted, and wet-compacted, clayey soil conditions.
--------------------	--

Table 14. 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

Laboratory Controlled Compaction Tests

Laboratory Test Methods

The above summarized research (Pitt, *et al.* 1999a) has identified significant reductions in infiltration rates in disturbed urban soils. The tests reported in the following discussion were recently 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 15. The initial sample moistures were determined and water was added to bring the initial soil moistures to about 8%, per standard procedures (ASTM D1140-54), reflecting typical “dry” soil conditions and to allow water movement through the soil columns. Table 16 lists the actual soil moisture levels at the beginning of the tests, along with the actual dry bulk soil densities and indications of root growth problems.

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 cm³). 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°F.

Table 15. 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 16. Soil Moisture and Density Values during Laboratory Tests

Soil Types	Compaction Method	Dry Bulk Density Before Test (g/cc)	Root Growth Potential Problems (NRCS 2001)		Before Test Moisture Content (%)	After Test Moisture Content (%)	
			Ideal Bulk Density	Bulk Densities that may Affect Root Growth			Bulk Densities that Restrict Root Growth
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

As shown on Table 16, 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” moisture levels generally corresponded to the “saturated soil” conditions of the earlier field measurements.

Also shown on Table 16 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 20 through 24 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 bioretention 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.

Most recently, another series of controlled laboratory tests were conducted to better simulate field conditions and standard double-ring infiltration tests, as shown in Table 17. 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.

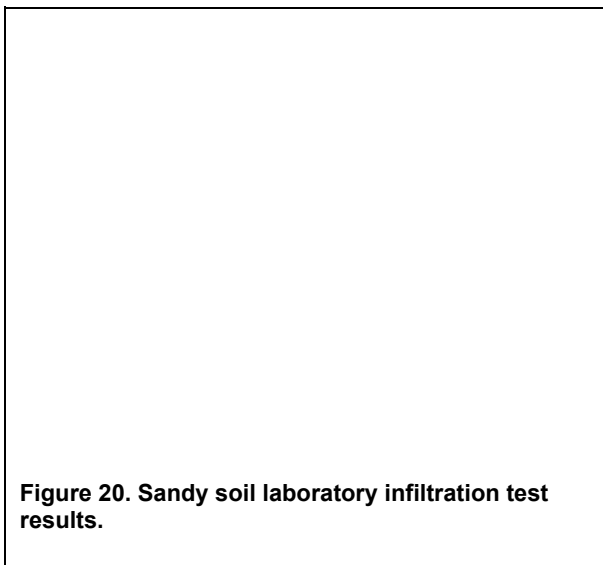


Figure 20. Sandy soil laboratory infiltration test results.

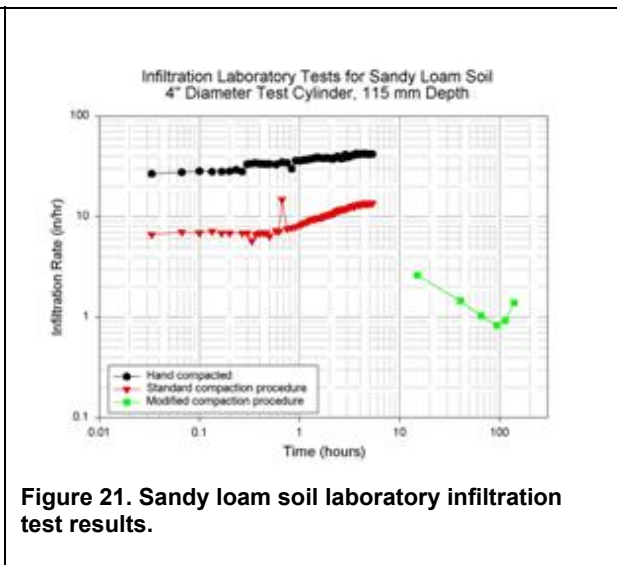


Figure 21. Sandy loam soil laboratory infiltration test results.

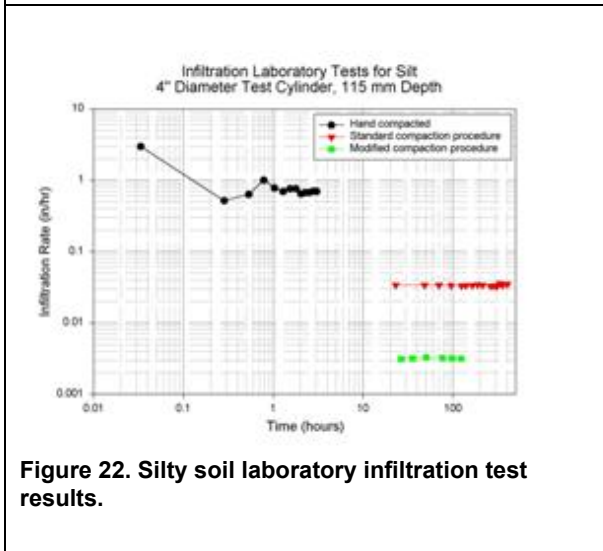


Figure 22. Silty soil laboratory infiltration test results.

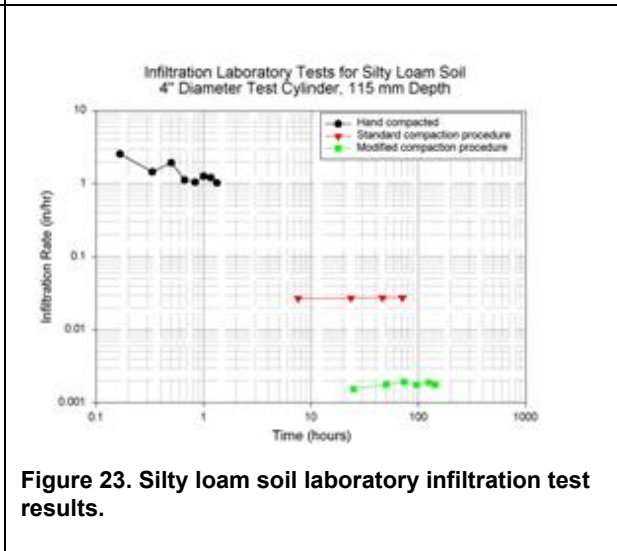


Figure 23. Silty loam soil laboratory infiltration test results.

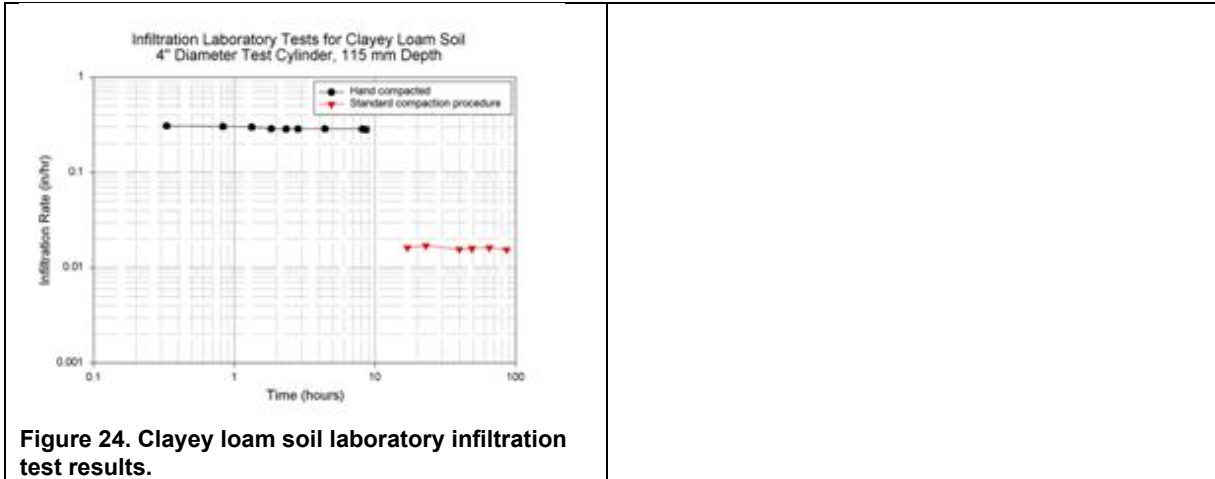


Table 17. Low-Head Laboratory Infiltration Tests for Various Soil Textures and Densities (densities and observed infiltration rates)

	Hand Compaction	Standard Compaction	Modified Compaction
Sand (100% sand)	Density: 1.36 g/cc (ideal for roots) 0 to 0.48 hrs: 9.35 in/hr 0.48 to 1.05 hrs: 7.87 in/hr 1.05 to 1.58 hrs: 8.46 in/hr	Density: 1.71 g/cc (may affect roots) 0 to 1.33 hrs: 3.37 in/hr 1.33 to 2.71 hrs: 3.26 in/hr	Density: 1.70 g/cc (may affect roots) 0 to 0.90 hrs: 4.98 in/hr 0.90 to 1.83 hrs: 4.86 in/hr 1.83 to 2.7 hrs: 5.16 in/hr
Silt (100% silt)	Density: 1.36 g/cc (close to ideal for roots) 0 to 8.33 hrs: 0.26 in/hr 8.33 to 17.78 hrs: 0.24 in/hr 17.78 to 35.08 hrs: 0.25 in/hr	Density: 1.52 g/cc (may affect roots) 0 to 24.22 hrs: 0.015 in/hr 24.22 to 48.09: 0.015 in/hr	Density: 1.75 g/cc (will likely restrict roots) 0 to 24.20 hrs: 0.0098 in/hr 24.20 to 48.07: 0.0099 in/hr
Clay (100% clay)	Density: 1.45 g/cc (may affect roots) 0 to 22.58 hrs: 0.019 in/hr 22.58 to 47.51 hrs: 0.016 in/hr	Density: 1.62 g/cc (will likely restrict roots) 0 to 100 hrs: <2X10 ⁻³ in/hr	Density: 1.88 g/cc (will likely restrict roots) 0 to 100 hrs: <2X10 ⁻³ in/hr
Sandy Loam (70% sand, 20% silt, 10% clay)	Density: 1.44 g/cc (close to ideal for roots) 0 to 1.17 hrs: 1.08 in/hr 1.17 to 4.37 hrs: 1.40 in/hr 4.37 to 7.45 hrs: 1.45 in/hr	Density: 1.88 g/cc (will likely restrict roots) 0 to 3.82 hrs: 0.41 in/hr 3.82 to 24.32 hrs: 0.22 in/hr	Density: 2.04 g/cc (will likely restrict roots) 0 to 23.50 hrs: 0.013 in/hr 23.50 to 175.05 hrs: 0.011 in/hr
Silty Loam (70% silt, 20% sand, 10% clay)	Density: 1.40 g/cc (may affect roots) 0 to 7.22 hrs: 0.17 in/hr 7.22 to 24.82 hrs: 0.12 in/hr 24.82 to 47.09 hrs: 0.11 in/hr	Density: 1.64 g/cc (will likely restrict roots) 0 to 24.62 hrs: 0.014 in/hr 24.62 to 143.52 hrs: 0.0046 in/hr	Density: 1.98 g/cc (will likely restrict roots) 0 to 24.62 hrs: 0.013 in/hr 24.62 to 143.52 hrs: 0.0030 in/hr
Clay Loam (40% silt, 30% sand, 30% clay)	Density: 1.48 g/cc (may affect roots) 0 to 2.33 hrs: 0.61 in/hr 2.33 to 6.13 hrs: 0.39 in/hr	Density: 1.66 g/cc (will likely restrict roots) 0 to 20.83 hrs: 0.016 in/hr 20.83 to 92.83 hrs: 0.0066 in/hr	Density: 1.95 g/cc (will likely restrict roots) 0 to 20.83 hrs: <0.0095 in/hr 20.83 to 92.83 hrs: 0.0038 in/hr

Summary

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. Also, 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 bioretention devices).

The initial exploratory analyses of the data showed that sand was mostly affected by compaction, with little change due to moisture levels. However, the clay sites were affected by a strong interaction of compaction and moisture. The variations of the observed infiltration rates in each category were relatively large, but four soil conditions were found to be distinct. The data from each individual test were fitted to the Horton equation, but the resulting equation coefficients were relatively imprecise (Table 18) and it may not matter much which infiltration model is used, as long as the uncertainty is considered in the evaluation. Therefore, when modeling runoff from urban soils, it may be best to assume relatively constant infiltration rates throughout an event, and to utilize Monte Carlo procedures to describe the observed random variations about the predicted mean value, possibly using the time-averaged infiltration rates and COV values shown in Table 19.

Table 18. Observed Horton Equation Parameter Values for Sandy and Clayey Soils

	f_o (in/hr)		f_c (in/hr)		k (1/min)	
	mean	range	mean	range	mean	range
Observed noncompacted sandy soils	39	4.2 to 146	15	0.4 to 25	9.6	1.0 to 33
Observed compact sandy soils	15	0.1 to 86	1.8	0.1 to 9.5	11	1.8 to 37
Observed dry noncompacted clayey soils	18	2.5 to 58	6.6	0.1 to 24	8.8	-6.2 to 19
Observed for all other clayey soils (compacted and dry, plus all saturated conditions)	3.4	0 to 48	0.4	-0.6 to 6.7	5.6	0 to 46

Very large errors in soil infiltration rates can easily be made if published soil maps and most available models are used for typically disturbed urban soils, as these tools ignore compaction. Knowledge of compaction (which can be mapped using a cone penetrometer, or estimated based on expected activity on grassed areas) can be used to much more accurately predict stormwater runoff quantity.

Table 19. Soil Infiltration Rates (in/hr) for Different Categories and Storm Durations, mean (COV)

	15 minutes	30 minutes	60minutes	120 minutes
Sand, Non-compacted	22.9 (0.4)	19.5 (0.4)	16.9 (0.4)	16.3 (0.4)
Sand, Compacted	6.7 (0.2)	4.9 (0.2)	3.8 (0.2)	2.5 (0.2)
Clay, Dry Non-compacted	12.7 (1.0)	10.8 (1.0)	9.6 (1.0)	8.8 (1.0)
All other clayey soils (compacted and dry, plus all saturated conditions)	1.8 (1.5)	1.3 (1.5)	1.0 (1.5)	0.7 (1.5)

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 variations in the infiltration rates in each category can be generally high (up to 1.5), they are much less than if compaction was ignored. The results of the factorial analysis indicated that the best models were separated by the soil texture. For more accurate modeling, it is recommended that site specific data be obtained. 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.

Table 20 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 summer to examine some of these issues again. We expect to report these results during the conference presentation. 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 moisture 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.

The use of soil amendments, or otherwise modifying 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, is summarized in the following section (Pitt, *et al.* 1999a). 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 the 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.

Table 20. Comparison of Infiltration Rates from Different Test Series

Group	Field Test Average Infiltration Rates (in/hr and COV)	Low-head Laboratory Test Results	High-head Laboratory Test Results
Noncompacted sandy soils	13 (0.4)	8 to 9.5 in/hr	30 to 120 in/hr
compact sandy soils	1.4 (1.3)	3 to 5 in/hr	0.5 to 60 in/hr
Noncompacted and dry clayey soils	9.8 (1.5)	0.4 to 0.6 in/hr	0 to 0.3 in/hr
All other clayey soils (compacted and dry, plus all wetter conditions)	0.2 (2.4)	0 to 0.4 in/hr	0 to 0.02 in/hr
Noncompacted silty and loamy soils	na	0.25 to 0.6 in/hr	0.5 to 3 in/hr
Compacted silty and loamy soils	na	0 to 0.02 in/hr	0 to 0.04 in/hr

This information can be effectively used in the modeling of small-scale stormwater controls, such as bioretention 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 21 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 21. Example Calculations of Benefits of On-Site Stormwater Controls (% reduction of annual roof runoff volumes)

	Phoenix, AZ	Seattle, WA	Birmingham, AL
Roof garden (1in/hr 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. Bioretention 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.

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. The test results presented in this section (Pitt, *et al.* 1999a), conducted by the College of Forestry Resources at the University of Washington in the Seattle area, showed the benefits of amending urban soils with compost. It was anticipated that compost-amended soils would improve infiltration characteristics of these soils, along with providing some filtration/sorption benefits to capture stormwater pollutants before they enter the groundwater.

Several golf course and athletic field test sites were also examined in Alabama during this research to document how turf areas can be constructed to enhance infiltration. These areas were designed to rapidly dry-off following a rain to minimize downtime due to excessive soil moisture 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 multiple 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. It

is likely that similar soil construction (without subsurface drainage in most cases) could be used in high density urban areas to enhance stormwater infiltration.

Field Studies on Infiltration Capabilities of Compost-Amended Soils

This research project examined the benefits of using compost as a soil amendment to improve the infiltration capacity and pollutant retention capacity of disturbed urban soils. Currently, due to their wide distribution and inherent stability, most residential housing developments in the Seattle area are sited on the Alderwood soil series, which is characterized by a compacted subsurface layer that restricts vertical water flow. When disturbed (and particularly when disturbed with cut and fill techniques as with residential or commercial development), uneven water flow patterns develop due to restricted permeability. This contributes to excessive overland flow (especially during storm events) and transport of dissolved and suspended particulates to surface waters.

Research has demonstrated compost's effectiveness in improving the soil physical properties of porosity and continuity of macropores which influence soil-water relationships. Compost's chemical properties can also be valuable in some cases, such as in complexing potentially harmful trace metals including copper, lead, and zinc. Under this premise, the effectiveness of using compost to increase stormwater infiltration and water holding capacity of glacial till soils was examined during special tests in the Seattle area. Additionally, the project also examined whether or not increasing the infiltrative and retentive capacity of glacial till soils (Alderwood series) can increase the contact with and retention of nutrients and metals by soil absorptive mechanisms.

The College of Forest Resources (CFR) has examined the effectiveness of using compost as a soil amendment to increase surface water infiltration to reduce the quantity and/or intensity of surface runoff and subsurface flow from land development projects. In addition, runoff and subsurface flow was evaluated for dissolved nutrients and other constituents.

The CFR utilized the existing Urban Water Resource Center (UWRC) project site at the University of Washington's Center for Urban Horticulture (CUH) for conducting the study. The CFR utilized the UWRC design of large plywood beds for containing soil and soil-compost mixes. Additional sites of a similar design were also constructed at Timbercrest and Woodmoor public schools in cooperation with the Woodinville Water District.

The test plots at the UWRC were developed and tested previously during a study conducted for the city of Redmond, WA (Harrison, *et al.* 1997). The following paragraphs summarize some of the findings and conclusions from that earlier study, conducted when the test plots were newly constructed. The earlier project specifically examined the use of compost as an amendment to Alderwood series soil to increase water-holding capacity, reduce peak flow runoff, and decrease phosphorus in both surface runoff and subsurface flows. Seven 8 ft. x 32 ft. beds were constructed out of plywood lined with plastic and filled with Alderwood subsoil or mixtures of soil and compost. Surface and subsurface flow samples were obtained over the period from March 7 to June 9, 1995, during a series of seven simulated rainfall events. To create different antecedent soil moisture conditions, some storm events were quickly followed by another event. Simulated rainfall was applied at total amounts ranging from 0.76 to 2.46 inches per storm, with rainfall intensities ranging from 0.29 to 0.63 in/hour. Compost amendments had the following effects on physical water properties:

- Water-holding capacity of the soil was about doubled with a 2:1 compost:soil amendment.
- Water runoff rates were moderated with the compost amendment, with the compost-amended soil showing greater lag time to peak flow at the initiation of a rainfall event and greater base flow in the interval following a rainfall event.

The following comments are examples of the beneficial flow characteristics associated with composted amended soil. At the start of the rainfall events, there was an increased lag time before significant runoff occurred. The compost-amended plots continued to store higher rates and total amounts of water for a longer period of time. Total storage increased by about 65%, and the field capacity increased by about 60%, with compost amendment. During one test with a rainfall intensity of about 0.3 in/hour, for example, the control (unamended) plot required about 30 minutes to respond with total surface runoff and subsurface flow >0.01 in/hour. The compost-amended site,

however, required nearly twice as long to respond with a similar flow. It required 0.75 hours from the start of the rainfall simulation for the total flow to become >0.1 in/hour in the unamended soil, while it required 1.75 hours for the compost-amended soil to increase to that rate. In order for the total runoff (surface plus subsurface flows) to reach 90% of the input rainfall intensity, it required nearly 2.0 hours for the unamended site, compared to 5.25 hours for the compost-amended site. Following the cessation of rainfall, it required 0.75 hours for total runoff in the unamended site to drop to $<10\%$ of the rainfall intensity, where it required 1.5 hours for the compost-amended site. Similar results occurred during the other tests using smaller rainfall intensities and total amounts, including one series of natural rainfall events. Compost-amended soils consistently had longer lag times to response, longer times to peak flows, higher base flows, higher total storage, and smaller total runoff than unamended soils. This indicates that compost-amended soils have better water-holding and runoff characteristics than unamended Alderwood soils.

The total (surface plus subsurface) runoff effects of using compost amendments during the wettest parts of the winter would likely be minimal on these Alderwood soils, since there is very little transpiration during this time. However, during the early fall and late spring seasons, the additional water-holding capacity of the compost-amended soils would result in additional transpiration from the plots and possibly lowered need for irrigation. Despite the lack of probable effects on total runoff during the winter season, the effect on storm peak surface flows would clearly be significant.

Nutrient concentrations (total P, soluble-reactive P and nitrate-N) in the surface runoff and subsurface flows were also measured for a series of artificial and natural rainfall events during this earlier study. For the overall study, which included fertilizer treatments, the following results were observed:

- Runoff from the compost-amended soil had 24% lower average total P concentration (2.05 vs. 2.54 mg/L) compared to the Alderwood soil that did not receive compost.
- Soluble-reactive P was 9% lower in the compost-amended soil (1.09 vs. 1.19 mg/L) compared to the Alderwood soil that did not receive compost amendment.
- Nitrate-N was 17% higher in the compost-amended soil (1.68 vs. 1.39 mg/L) compared to the Alderwood soil that did not receive compost amendment.

Overall, the amended sites had somewhat higher $\text{NO}_3\text{-N}$ concentrations. A possible reason may be associated with the behavior of the semi-wild Canada geese living in the area. The geese were noted to prefer eating the much more lush grass on the compost-amended plots, while ignoring the unamended plots. During these feedings, they also left a considerable amount of droppings, which typically are high in phosphorus and nitrogen compounds. In any case, these nutrient concentrations in the runoff collected from compost-amended plots vs. the unamended plots do not show large differences. The water flow data from several storm events was coupled with the nutrient concentration data to generate fluxes of nutrients from the plots. When these fluxes were summed, the compost-amended soils showed the following, compared to the unamended soils:

- 70% less total P,
- 58% less soluble-reactive P, and
- 7% less nitrate in runoff compared to runoff from the glacial till-only soil.

These differences in fluxes were attributed more to the changes in water flux rates than to water chemistry, but both accounted for the lowered P with compost amendment.

The artificial storms utilized in these studies represent intense rains having 25 to 100 year return intervals. It would be expected that the differences between the glacial till-only soil and the compost-amended glacial till soil would be greater at less-intense rainfall events, though the peak rates of runoff of both are likely to be reduced. The results of this earlier study pointed out the promise of the use of organic amendments for improving water-holding

capacity, runoff properties and runoff water quality of Alderwood soils converted to turf grass during urban development.

These earlier study results were the basis for this current study. This study examined some of these same test plots at the UWRC several years after their initial establishment, and during natural rains, to see if their behavior is substantially different with time. In addition, new test sites were also established at two school additional locations for comparison.

Soil and Compost Analysis

The terminology used in industry and science for compost and soil properties is somewhat inconsistent. In this report, percent by weight uses an oven-dried basis for calculation. Volumes can change depending on handling, storage, moisture content and other factors. Also, the density (weight per unit volume) for compost is usually much lower (i.e. 0.2-0.3 g/cm³) than for soil (i.e. 1.0-1.4 g/cm³), so a weight percent change from compost amendment will usually be much lower than a volume unit change, and moisture capacity based on volume may be much different than moisture capacity based on weight. There were also large changes in the chemical and physical properties of the soil/compost mixtures due to the compost amendment.

Total C and organic matter was enhanced by adding compost, increasing from 0.1-0.4% C to about 1.1-5.2% C by weight. Total N was also enhanced, increasing from 0.02-0.08% to about 0.06-0.35% with the compost amendment. Gravimetric field moisture capacity increased significantly from 24% to 35% with the compost amendment. Volumetric field moisture capacity was also increased from 46 to 50% by the addition of compost.

Total porosity was increased from 41 to 48%. Bulk density was decreased from about 1.7 to 1.1 g/cm³. Particle density was decreased from about 2.5 to 2.1 g/cm³. Particle size analysis was not greatly affected by the compost amendment. Soil structure, which is not a quantitative property, was also not greatly affected by compost amendment.

Thus, there was a generally beneficial effect of the compost amendment in regards to nutrient content as well as soil physical properties known to affect water relations in soils.

Water Quantity Observations at Test Plots

This study utilized a dual tipping bucket system to measure surface runoff and subsurface flows vs. time. As pointed out earlier, the tipping buckets did not accurately record all surface runoff or subsurface seepage at the test sites due to unexpected leakage or faulty operation of the tipping buckets. However, most of the surface runoff and subsurface flow information was obtained.

Infiltration rate measurements were also made at the test plots using the ASTM D3385-94 double ring method. Table 22 shows the results of these tests, contrasting the measured infiltration rates at the compost-amended test plots with the rates measured at the test plots that only contained soil. The use of compost-amended soil resulted in significantly increased infiltration rates compared to soil alone. These infiltration rate increases (from 1.5 to 10 times the untreated rates) would substantially decrease the runoff volumes and flow rates during rain storms from turf areas. These lower runoff volumes and flow rates would decrease many detrimental stormwater effects, including reduced mass discharges of pollutants, reduced downstream flooding, and improved in-stream habitat conditions for aquatic life. These benefits would be much more pronounced in areas having high rainfall intensities and where much of the urban soils could be amended with compost. The additional infiltrating water would be more slowly released to the surface waters after the initial runoff flows have subsided, or would recharge deeper groundwaters, depending on subsurface soil conditions. The soil structure at the Alderwood soil sites would likely prevent much of this increased infiltrating water from reaching deeper groundwaters, but the compost amendments would still improve surface water flow characteristics, as extensively evaluated by Harrison, *et al.* (1997) during the initial tests at the Urban Horticulture test plots.

Table 22. Infiltration Rate Measurements at Field Test Plots

Location	Test Plot Treatment	Average Infiltration Rate (cm/hr) (in/hr)	Improvement with Compost (ratio)
UW Urban Horticulture plot 1	Alderwood soil A	1.2 (0.5)	
UW Urban Horticulture plot 2	Alderwood soil A with Cedar Grove compost	7.5 (3.0)	6.3
UW Urban Horticulture plot 5	Alderwood soil B	0.8 (0.3)	
UW Urban Horticulture plot 6	Alderwood soil B with GroCo compost	8.4 (3.3)	10.5
Timbercrest	Alderwood soil C	0.7 (0.3)	
Timbercrest	Alderwood soil C with Cedar Grove compost	2.3 (0.9)	3.3
Woodmoor	Alderwood soil D	2.1 (0.8)	
Woodmoor	Alderwood soil D with Cedar Grove compost	3.4 (1.3)	1.5

Surface runoff and subsurface flows were also monitored over several extended periods at the test plot sites. Table 23 summarizes the surface runoff and subsurface flow data. This table shows the fractions of the total rainfall that resulted in surface runoff, subsurface flow, and other losses (assumed to be mostly evapotranspiration). The surface runoff fraction is the volumetric runoff coefficient (R_v) and is the simple ratio of runoff depth to rainfall depth. The four soil-only Alderwood test plots were quite different, with average R_v values ranging from about 0.01 to 0.25, reflecting a large amount of variability of infiltration conditions for this single soil type. The age of construction of the test plots does not explain this variation.

The soil-only and compost-amended soil test plots for each of the soils at the UW Urban Horticulture site were quite similar, with both test plots in each pair having very similar R_v values (even though the infiltration measurements report previously indicated large differences). In contrast, the newer test plots at Timbercrest and Woodmoor showed significant decreases in surface runoff for the compost-amended test plots, compared to the soil-only test plots. In fact, very little surface runoff was observed at the Timbercrest test plot having compost-amended soils. The soil-only plot at Timbercrest had a R_v of only about 0.04, so the improved infiltration improvement is not very important from a flow perspective (but could be from a mass pollutant runoff perspective). However, the Woodmoor site showed large and important improvements in infiltration conditions, with the R_v being reduced from about 0.25 (relatively large for a soil), to a much smaller R_v of about 0.05.

In addition, the evapotranspiration rates all increased with compost-amended soils, although by only a very small amount at one of the urban horticulture test plot pairs. The increase in evapotranspiration ranged from about 33 to 100 percent at the newer sites at Timbercrest and Woodmoor.

Table 23. Water Flow Fractions (range and flow-weighted averages)

Location	Treatment	Surface runoff	Subsurface flow	Evapotranspiration
UW plot 1	Alderwood soil A	0.004 – 0.011 (0.009)	0.50 – 1.00 (0.74)	0.00 – 0.49 (0.25)
UW plot 2	Alderwood soil A and Cedar Grove compost	0.005 – 0.010 (0.009)	0.45 – 0.89 (0.74)	0.11 – 0.54 (0.25)
	Ratio of compost to soil average fraction	0.98	1.00	1.01
UW plot 5	Alderwood soil B	0.15 – 0.26 (0.22)	0.39 – 0.83 (0.59)	0.02 – 0.44 (0.19)
UW plot 6	Alderwood soil B and GroCo compost	0.001 – 0.42 (0.25)	0.00 – 0.77 (0.46)	0.13 – 1.00 (0.29)
	Ratio of compost to soil average fraction	1.10	0.78	1.57
Timbercrest	Alderwood soil C	0.006 – 0.13 (0.040)	0.32 – 0.39 (0.35)	0.54 – 0.68 (0.61)
	Alderwood soil C and Cedar Grove compost	0.00 – 0.00 (0.00)	0.02 – 0.43 (0.19)	0.57 – 0.98 (0.81)
	Ratio of compost to soil average fraction	0.00	0.54	1.33
Woodmoor	Alderwood soil D	0.022 – 0.38 (0.25)	0.13 – 0.74 (0.59)	0.00 – 0.84 (0.16)
	Alderwood soil D and Cedar Grove compost	0.00 – 0.092 (0.045)	0.03 – 0.79 (0.64)	0.15 – 0.97 (0.31)

Ratio of compost to soil average fraction	0.18	1.08	1.97
---	------	------	------

Water Quality Observations at Test Plots

Visual Appeal of Test Sites and Need for Fertilization

It was apparent, from the visual appeal of the sites, that more inorganic fertilizer will be applied to an unamended area, compared to an area that has been amended with compost, in order to achieve the same visual appeal. Following planting, compost-amended plots developed a dark green color quickly, and achieved 100% coverage much more rapidly than unamended plots. At the end of the study, the compost-amended sites were much better aesthetically, with a darker green color and no bare spots. No soil can be seen through the grass. There are many bare spots with exposed soil in the plots that did not receive compost amendment. The rates of growth of turf were also greater, even after a considerable period of time. The visual appeal of the compost-amended sites was much greater during the duration of the study, although all sites did grow grass.

Inorganic fertilizer was applied initially in equal rates at all test sites during these tests since there was nothing but bare ground initially. Over time, however, it was apparent that it would be very difficult to achieve the same visual appeal with inorganic fertilizer applied to unamended soil, in comparison to compost-amended soil.

Overall Range of Water Quality Observations in Surface Runoff and Subsurface Flows

There was a very large variation in water quality in the surface runoff and subsurface flow samples. For instance, the average total P (TP) concentration for all samples analyzed was 2.76 mg/L, while the minimum P was 0.00 and the maximum 125 mg/L. This high degree of variation in concentration is not unexpected, considering the variety of sampling conditions: test plots with treatments ranging from surface runoff with high water flow in a very infertile, unfertilized glacial till soil to surface runoff and subsurface flows in soils freshly fertilized with soluble NPK fertilizers.

The sampling scheme was organized with a complete block design in order to recognize significant differences between the test plots and between surface runoff and subsurface flows. The following subsection presents the statistical analyses for these comparisons. Before those results are presented, it is worthwhile to examine patterns between the water quality constituents. The following discussion therefore presents the results of simple Pearson correlation analyses, cluster analyses, and principal component analyses that were conducted using the complete data set (except for those analyses resulting in mostly non detected observations). SYSTAT, version 8, was used to conduct these statistical tests.

A Pearson correlation matrix was prepared comparing all of the observed data. High correlations imply close and simple relationships between the contrasted parameters. As an example, it is expected that many of the nutrients would be highly correlated with each other because of their common source (chemical fertilizer), and toxicity may be highly correlated with toxic constituents. The following list shows the correlation pairings that had high correlation coefficients (>0.7), when all of the water quality data were compared:

Phosphate (PO₄) with:

- TP (0.998)
- NH₄ (0.975)
- TN (0.955)
- SO₄ (0.949)
- S (0.981)

Total Phosphorus (TP) with:

- PO₄ (0.998)
- NH₄ (0.976)
- TN (0.958)

- SO₄ (0.945)
- S (0.979)

Ammonium (NH₄) with:

- PO₄ (0.975)
- TP (0.976)
- TN (0.995)
- SO₄ (0.977)
- K (0.773)
- S (0.994)

Nitrate (NO₃) with:

- none (the largest correlation with NO₃ was with Ca at 0.34)

Total Nitrogen (TN) with:

- PO₄ (0.955)
- TP (0.958)
- NH₄ (0.995)
- SO₄ (0.978)
- K (0.828)
- S (0.987)

Chloride (Cl) with:

- Mg (0.699)
- Na (0.723)

Sulfate (SO₄) with:

- PO₄ (0.949)
- TP (0.945)
- NH₄ (0.977)
- TN (0.978)
- K (0.774)
- S (0.988)

Aluminum (AL) with:

- Si (0.964)

Calcium (Ca) with:

- Mg (0.901)

Copper (Cu) with:

- none (many non-detected Cu values, the largest correlation with Cu was for toxicity at 0.34)

Iron (Fe) with:

- none (the largest correlations with Fe were with Si at 0.532 and Al at 0.53)

Potassium (K) with:

- NH₄ (0.773)
- TN (0.828)
- SO₄ (0.774)

Magnesium (Mg) with:

- Cl (0.699)
- Ca (0.901)
- Na (0.810)

Manganese (Mn) with:

- Ca (0.758)

Sodium (Na) with:

- Cl (0.723)
- Ca (0.739)

Sulfur (S) with:

- PO₄ (0.981)
- TP (0.979)
- NH₄ (0.994)
- TN (0.987)
- SO₄ (0.988)

Zinc (Zn) with:

- none (the largest correlations with Zn were with Al at 0.42, Si at 0.39, and with toxicity at 0.35)

Silica (Si) with:

- Al (0.964)

Tenth percentile particle size with:

- Fiftieth percentile particle size (0.791)

Fiftieth percentile particle size with:

- Tenth percentile particle size (0.791)
- Ninetieth percentile particle size (0.721)

Ninetieth percentile particle size with:

- Fiftieth percentile particle size (0.721)

Toxicity with:

- none (the largest correlations with toxicity were with S at 0.55, Na at 0.54, SO₄ at 0.59, and Cl at 0.55)

These correlation coefficients show the expected strong correlations between the nutrient parameters and between other obviously related parameters (such as SO₄ and S, major cations and major anions, and particle sizes). It is surprisingly to note the poor correlation between NO₃ and TN (0.011) and between NO₃ and NH₄ (0.002). The strongest correlations with toxicity were noted for salinity parameters (NaCl and SO₄), pointing out the sensitivity of the test organism (a marine phytobacterium) with salinity.

More complex inter-relationships between the chemical parameters can be identified through cluster analyses. Figure 25 is a dendrogram showing the close relationships between the nutrients, and less clear relationships for many of the other parameters. Phosphate and total phosphorus, along with ammonium and total nitrogen, are the

closest and simplest relationships, while nitrate is poorly related to any other parameter. The major cations and major anions have a somewhat more complex inter-relationship, while toxicity is affected by all of the major ions, plus the nutrients.

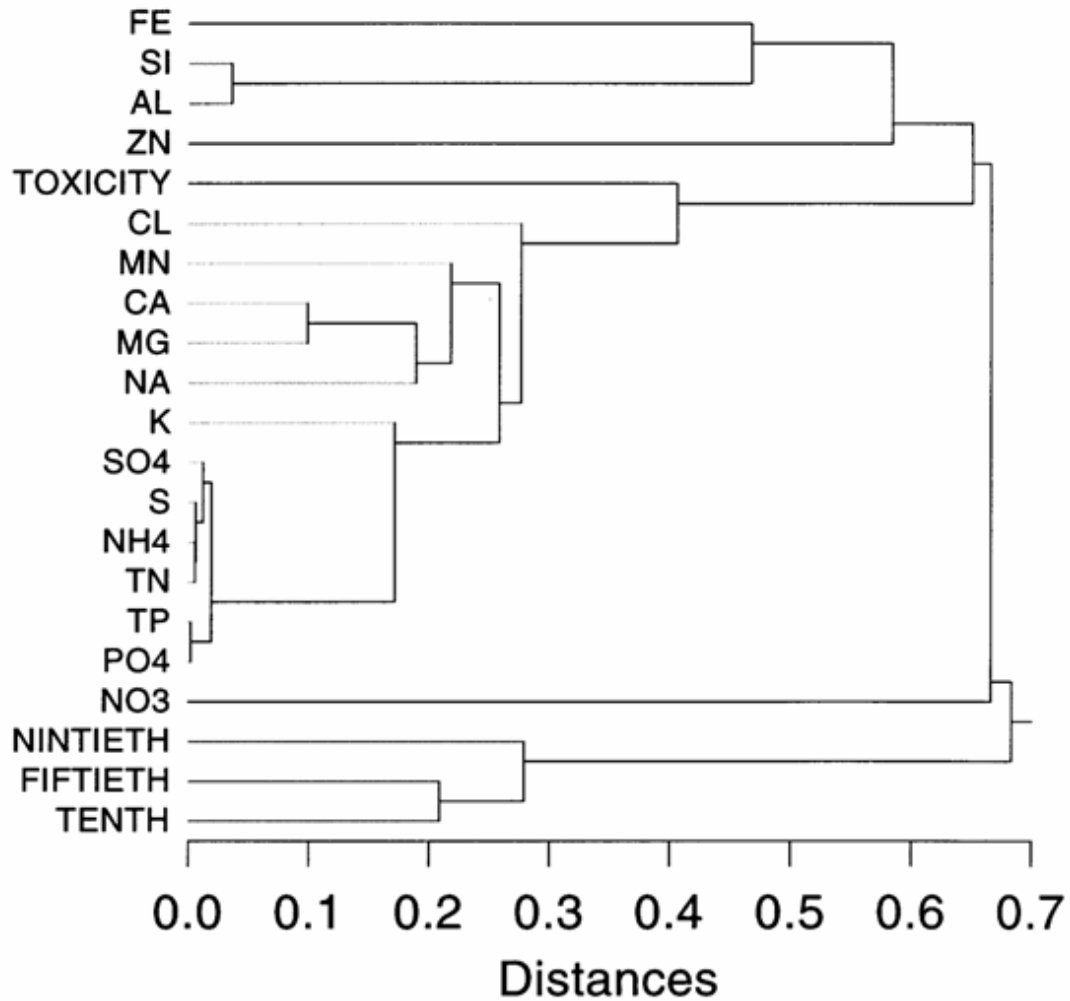


Figure 25. Dendrogram showing complex relationships of monitored chemical parameters at soil and amended soil test sites.

The final overall analysis conducted was a principal component evaluation of all of the water quality parameters. This analysis also groups the parameters into components that are closely related. In this case, three components were determined to account for about 75% of the total variance of the data. The first component accounted for about 45% of the variance and was mostly associated with the following 11 parameters: NH₄, TN, Cl, SO₄, Ca, K, Mg, Mn, Na, S, and toxicity. This component was mostly made of the major cations and anions, plus the nitrogen compounds, and toxicity. The second most important component explained a further 18% of the variance and was

mostly associated with the following six parameters: Al, Fe, Si, and the three particle size parameters. The final principal component explained about 12% of the total variance and was comprised of the following four parameters: PO₄, TP, NO₃, and Zn. Less important components accounted for the remaining 25% of the total variance and were comprised of combinations of all of the water quality parameters.

The following briefly lists some of the water quality criteria and goals for the constituents measured during this study:

Phosphate	0.1 mg/L goal to prevent eutrophication in flowing waters
Ammonia	as low as 0.11 mg/L for warm water and pH of 9, to 2.5 mg/L for cold water and pH of 6.5
Nitrate	10 mg/L for human health
Chloride	250 mg/L for human health
Zinc	5 mg/L (human health, through consumption of fish)
	33 at 25 mg/L hardness to 140 mg/L at 140 mg/L hardness for chronic exposure to fish

Many of the observed phosphate and ammonia concentrations exceeded the above water quality goals during all test conditions. However, only the maximum observed nitrate values exceeded the nitrate standard, and no chloride or zinc observations exceeded any of the listed criteria. Even though temperature was not monitored during this study, the use of landscaped areas is known to be an important moderating factor in controlling elevated runoff temperatures in stormwater from urban areas, compared to many other urban surfaces. A healthier turf stand will provide a greater temperature-moderating influence than bare soil, or a poor turf stand.

The soluble-reactive P (PO₄-P) concentration for all samples analyzed was 2.3 mg/L, while the minimum P was below detection, and the maximum was 125 mg/L. The average PO₄-P concentration measured is considerably above the State of Washington Water Quality recommendations for freshwater, according to WAC 173-201 (1992), which is 0.1 mg/L for flowing water not discharging directly into a lake or impoundment. The ammonium-N concentration averaged 6.6 mg/L, while the minimum ammonium-N was below detection, and the maximum was 360 mg/L. The NO₃-N concentration averaged 2.6 mg/L, while the minimum NO₃-N was below detection, and the maximum was 74 mg/L.

Overall, 72% of the 63 samples analyzed were determined to be not toxic (<20% light reductions), 25% were considered moderately toxic (light reductions of 20 to 60%), while 3% (2 samples) were considered to be highly toxic (>60% light reductions). Both of the highly toxic samples were from the Woodmoor test sites, one being a surface runoff sample from the soil-only plot (2/20/98), and the other being a subsurface flow sample from the compost-amended soil plot (1/5/98).

A few samples had significantly larger concentrations than most of the others, as listed below. These noted constituent concentrations were all much larger than for the other samples (typically at least 10 times greater):

- Woodmoor, Cedar Grove compost-amended test plot:

1/5/98, the first sample collected from this test plot, subsurface flow sample only (no surface runoff sample was available for analysis): NH₄ (59.4 mg/L), TN (118 mg/L), Cl (181 mg/L), Ca (190 mg/L), K (283 mg/L), Mg (70 mg/L), Mn (13 mg/L), Na (36 mg/L), and S (65 mg/L).

2/20/98, the next sample after the above analyses (surface runoff, subsurface flow concentrations): NH₄ (27, 43.9 mg/L), TN (48, 90 mg/L), SO₄ (4.8, 11 mg/L), Ca (52, 132 mg/L), and K (158, 241 mg/L).

3/15/98, the next sample after the above analyses (surface runoff only, as no subsurface flow sample was available for analysis): NH₄ (19 mg/L), TN (34 mg/L), and K (117 mg/L).

- Timbercrest, Cedar Grove compost-amended test plot:

6/26/98, surface runoff sample only (the subsurface sample was not available for analysis): PO₄ (125 mg/L), TP (125 mg/L), NH₄ (360 mg/L), TN (479 mg/L), SO₄ (223 mg/L), K (361 mg/L), and S (356 mg/L).

The Woodmoor compost-amended site was strongly influenced by initial conditions, likely by the initial Cedar Grove compost amendment leaching nutrients and other minerals. At this site, only the compost-amended plot showed dramatic decreases in concentrations with time, as shown on Figures 26 and 27. These figures dramatically show decreasing concentrations for phosphorus and nitrogen compounds in the subsurface flows with time for the compost-amended test plot at Woodmoor. No noticeable concentration trends are seen for the soil-only test plots. The nitrogen compounds in the subsurface flow from the compost-amended plot approached the subsurface flow concentrations from the soil-only plot after about six months. However, the phosphorus compounds remained high at the end of this period, although the concentrations decreased substantially since the beginning of the test period. As shown in the following subsections, the phosphorus concentrations in the runoff from the compost-amended test plots at the UW Urban Horticulture test plots remained two to three times higher than from the soil-only test plots, even after several years.

Both surface runoff and subsurface flows are very high on 2/20/98 at the Woodmoor Cedar Grove compost-amended test plot. That set of analyses showed large increases (about doubling the concentrations) in constituent concentrations after infiltrating through the compost-amended soil. The one very high value at Timbercrest (6/26/98) was also at the compost-amended test plot, but data was only available for the surface runoff. Therefore, it could not be confirmed if the surface runoff was also high, or if earlier samples were even higher (expected).

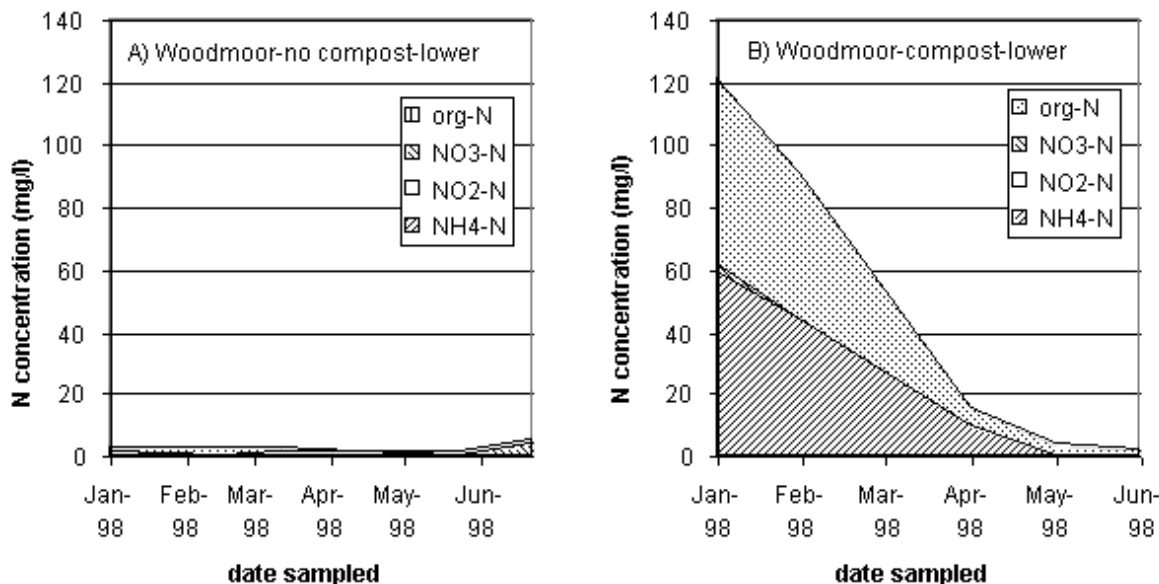


Figure 26. Nitrogen compounds and elemental concentration averages.

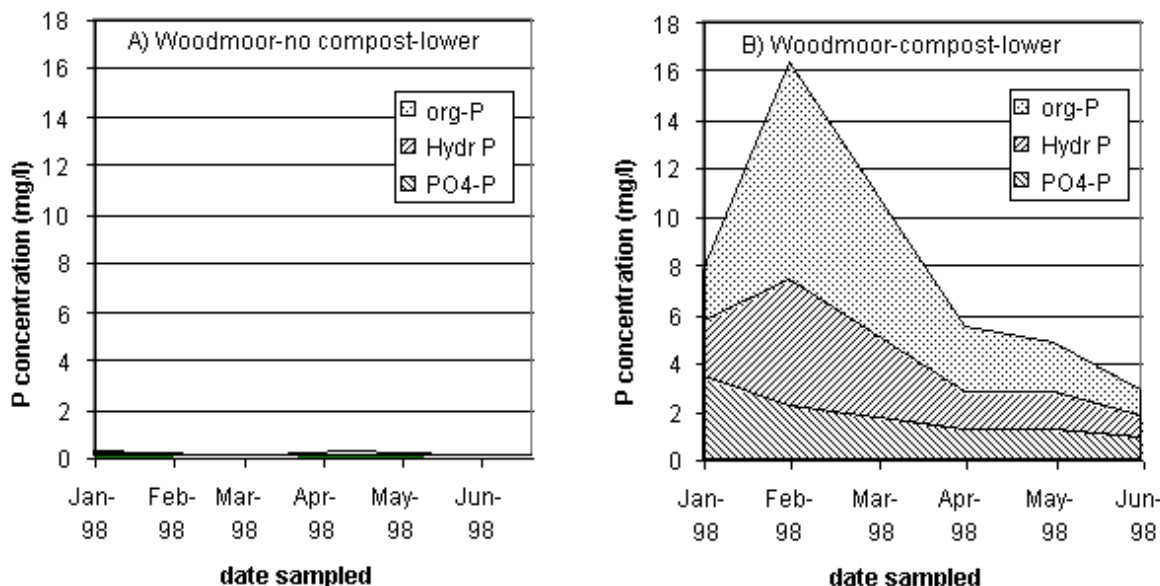


Figure 27. Phosphorus compounds and concentration averages.

Comparison of Water Quality from Amended vs. Unamended Test Plots

Table 24 summarizes the average concentrations for surface runoff and subsurface flow samples separated by “soil-only” test plots and “soil plus compost” test plots. This table shows the average observations along with the coefficient of variations (standard deviation divided by the average value). The table only shows data for tests having both surface runoff and subsurface flow samples. The subsurface flows in the soil-only test plots mostly had lower concentrations than the associated surface runoff. The only exceptions were NO₃, SO₄, Ca, Mg, and S which had slightly elevated concentrations (increases of about 10 to 30%) in the subsurface flows compared to the surface runoff. However, there were more constituents that were in higher concentrations in subsurface flows, compared to surface runoff, for the compost-amended soil test plots. In addition, the increases were generally larger (as much as 2.5 times greater) than for the increases observed at the soil-only test plots. The constituents with elevated concentrations in the subsurface flows compared to surface runoff at the compost-amended test plots were NO₃, TN, SO₄, Al, Ca, Fe, K, Mg, Mn, Na, and S.

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. Interestingly, the only exceptions were for the cations Al, Fe, Mn, Zn, Si, plus 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. Subsurface flow concentration increases for the compost-amended soil test sites were also common and about as large. The only exceptions being for Fe, Zn, and toxicity. Toxicity tests indicated reduced toxicity with filtration at both the soil-only and at the compost-amended test sites.

Table 24. Average (and COV) Values for all Runoff and Subsurface Flow Samples

Constituent (mg/L, unless noted)	Soil-only plots		Soil plus Compost Plots	
	Surface Runoff	Subsurface Flows	Surface Runoff	Subsurface Flows

PO ₄ -P	0.27 (1.4)	0.17 (2.0)	1.9 (1.0)	1.8 (1.2)
TP	0.49 (1.0)	0.48 (2.2)	2.7 (0.9)	2.5 (1.1)
NH ₄ -N	0.65 (1.7)	0.23 (1.3)	4.1 (1.8)	3.5 (3.0)
NO ₃ -N	0.96 (1.4)	1.2 (2.5)	3.0 (1.6)	6.2 (2.8)
TN	2.5 (0.9)	1.9 (0.7)	8.4 (1.5)	10 (2.1)
Cl	2.4 (1.0)	2.1 (0.9)	6.7 (1.1)	5.0 (1.6)
SO ₄ -S	0.68 (1.1)	0.95 (2.0)	1.5 (0.9)	2.4 (1.4)
Al	11 (1.8)	1.7 (2.1)	0.7 (1.6)	2.4 (1.6)
Ca	12 (1.5)	17 (0.7)	18 (1.1)	35 (1.1)
Cu	0.01 (0.8)	0.01 (1.6)	0.02 (1.2)	0.02 (0.9)
Fe	4.6 (1.4)	2.8 (1.6)	1.2 (1.5)	2.6 (0.9)
K	5.4 (1.0)	4.6 (0.8)	30 (1.3)	34 (1.6)
Mg	3.9 (0.8)	5.0 (0.6)	5.8 (1.2)	10 (1.1)
Mn	0.75 (2.9)	0.41 (2.8)	0.36 (1.9)	0.80 (2.4)
Na	3.8 (0.9)	3.4 (0.5)	3.2 (0.8)	4.6 (1.2)
S	1.1 (0.8)	1.3 (1.5)	2.5 (0.8)	4.7 (1.6)
Zn	0.2 (1.2)	0.05 (2.2)	0.14 (1.1)	0.03 (1.8)
Si	26 (1.7)	8.9 (0.5)	4.2 (1.1)	11 (0.7)
10 th percentile size (µm)	2.9 (0.7)	3.1 (0.4)	2.8 (0.3)	3.5 (0.6)
50 th percentile size (µm)	12 (1.0)	13 (0.6)	15 (0.4)	14 (0.7)
90 th percentile size (µm)	45 (0.5)	41 (0.5)	46 (0.4)	47 (0.6)
Toxicity (% light decrease)	25 (0.7)	13 (0.5)	16 (0.8)	10 (1.1)

Mass Discharges of Nutrients and other Water Quality Constituents

The mass discharges of water and nutrients were calculated for each sampling period. As noted previously, 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, at least for a few years. Table 25 summarizes these expected changes in surface runoff and subsurface flow mass pollutant discharges associated with compost-amended soils, using the paired data only. The paired data concentration increases were multiplied by the runoff reduction factors to obtain these relative mass discharge changes. The runoff volume decreases used were for the newer test sites. The older test sites had less dramatic reductions in runoff values. Simultaneously, the older sites also had smaller concentration increases associated with adding compost to the soil. All of the surface runoff mass discharges are reduced by large amounts (to 2 to 50 percent of the unamended discharges). However, many of the subsurface flow mass discharges are expected to increase, 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 decrease.

The compost likely has significant sorption capacity and ion exchange capacity that is responsible for pollutant reductions in the infiltrating water. However, the compost is also leaching large amounts of nutrients to the surface and subsurface waters.

Table 25. Changes in Pollutant Discharges from Surface Runoff and Subsurface Flows at New Compost-Amended Sites, Compared to Soil-Only Sites

Constituent	Surface Runoff Discharges, Amended-Soil Compared to Unamended Soil	Subsurface Flow Discharges, 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

Since this table was based on paired analyses only (requiring both surface runoff and subsurface flow data for the calculations), the values may over-predict the benefits of compost-amended soils because it does not consider the three samples that had very high concentrations that were observed, but without the appropriate paired data for comparison/confirmation..

Conclusions

Groundwater Contamination Potential Associated with Stormwater Infiltration

It has been suggested that, with a reasonable degree of site-specific design considerations to compensate for soil characteristics, infiltration can be very effective in controlling both urban runoff quality and quantity problems (EPA 1983). This strategy encourages infiltration of urban runoff to replace the natural infiltration capacity lost through urbanization and to use the natural filtering and sorption capacity of soils to remove pollutants. However, potential groundwater contamination through infiltration of some types of urban runoff requires some restrictions. Infiltration of urban runoff having potentially high concentrations of pollutants that may pollute groundwater requires adequate pretreatment, or the diversion of these waters away from infiltration devices.

Compacted Urban Soils and Infiltration

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 bioretention 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 in each category were generally high (0.5 to 2), they were 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.

Water Quality and Quantity Effects of Amending Soils with Compost

There was a substantial difference in appearance of amended and unamended plots. There was insufficient grass growth in the unamended plots, even following initial establishment fertilization. The compost-amended plots were very attractive and needed no fertilization. In fact, the initial establishment fertilization probably wasn't necessary either, based on studies of turf grass growth in compost-amended soils without inorganic fertilization at the University of Washington on similar soils. Besides fertilizer applications, other external sources of nutrients to the test plots included wildlife, especially geese that were noted to selectively graze the compost-amended plots.

Application of compost material similar to that used during these studies would be possible by applying 4 inches of compost onto the surface of an Alderwood soil and tilling to a total depth of 12 inches, including the compost

amendment (8 inches into the soil). This mixing would probably need to be thorough and deep to achieve the conditions of this study. However, this is not likely to be possible with most existing equipment.

The results of these studies clearly show that amending soil with compost alters soil properties known to affect water relations of soils, including the water holding capacity, porosity, bulk density, and structure, as well as increasing soil C and N, and probably other nutrients as well. The mobilization of these constituents probably led to observed increases in P and N compounds in surface runoff compared to unamended soil plots.

Results of the earlier Redmond-sponsored tests (Harrison, *et al.* 1997) were somewhat different than obtained from the current tests. Some of these differences were likely associated with the age of the test plots, plus different rainfall conditions, and other site characteristics. The results of the earlier study clearly showed that compost amendment is likely an effective means of decreasing peak flows from all but the most severe storm events, even following very wet antecedent conditions. The increases in water holding capacity with compost amendment shows that storms up to 0.8 inches total rainfall would be well buffered in amended soils and not result in significant peak flows, whereas without the amendment, a storm about 0.4 inches total rainfall would be similarly buffered.

If a significant percentage of disturbed glacial till soils were amended with compost in this manner, it would have a significant beneficial effect on watershed hydrology. The absolute amount depends on many factors, but it is clear that compost amendment is an excellent means of retaining runoff on-site and reducing the rate of runoff from all but the most intense storm events, especially during the early critical years following development.

This study found that surface runoff decreased by five to ten times after amending the soil with compost, 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. Subsurface flow concentration increases for the compost-amended soil test sites were also common and about as large. The only exceptions being for Fe, Zn, and toxicity. Toxicity tests indicated reduced toxicity with filtration at both the soil-only and at the compost-amended test sites, likely due to the sorption or ion exchange properties of the compost.

When the decreased surface flow quantities were considered in conjunction with the increased surface runoff concentrations, it was found that all of the surface runoff mass discharges were reduced by large amounts (to 2 to 50 percent of the unamended discharges). However, many of the subsurface flow mass discharges are expected to increase, especially for ammonia, phosphate, total phosphorus, nitrates, and total nitrogen. The large phosphorus and nitrogen compound concentrations found in surface runoff and subsurface flows at the compost-amended soil sites decreased significantly during the time of the tests (about 6 months). However, the several year old test sites also tested had less, but still elevated concentrations compared to unamended soil-only test plots.

In conclusion, adding large amounts of compost to marginal soils enhances many desirable soil properties, including improved water infiltration (and attendant reduced surface runoff), increased fertility, and significantly enhanced aesthetics of the turf. Unfortunately, the compost also increased the concentrations of many nutrients in the runoff, especially when the site was newly developed, but with the increased infiltration of the soil, the nutrient mass runoff was likely significantly decreased. This is especially likely when the need for continuous fertilization to establish and maintain the turf is reduced, if not eliminated, at compost-amended sites.

References

- Akan, A. O., *Urban Stormwater Hydrology: A Guide to Engineering Calculations*. Lancaster, PA: Technomic Publishing Co., Inc., 1993.
- Alhajjar, B.J., G.V. Simsman and G. Chesters (1990). "Fate and Transport of Alachlor, Metolachlor and Atrazine in Large Columns." *Water Science and Technology*. Volume 22, number 6, pp. 87-94.

- American Society of Testing and Materials. 1996 *Annual Book of ASTM Standards*. West Conshohocken. PA: ASTM vol. 04.08,1994.
- Andelman, J., H. Bauwer, R. Charbeneau, R. Christman, J. Crook, A. Fan, D. Fort, W. Gardner, W. Jury, D. Miller, R. Pitt, G. Robeck, H. Vaux, Jr., J. Vecchioli, and M. Yates (1994). *Ground Water Recharge Using Waters of Impaired Quality*. Committee on Ground Water Recharge. Water Science and Technology Board. National Research Council. National Academy Press. Washington, D.C.
- Armstrong, David E. and Reynaldo Llena (1992). *Stormwater Infiltration: Potential for Pollutant Removal*. Report prepared for the Wisconsin Department of Natural Resources (Madison) and the U.S. Environmental Protection Agency, Chicago.
- AWWA (American Water Works Association) (1990). "Fertilizer Contaminates Nebraska Groundwater." *AWWA Mainstream*. Volume 34, number 4, p. 6.
- Bao-rui, Yan (1988). "Investigation into Mechanisms of Microbial Effects on Iron and Manganese Transformations in Artificially Recharged Groundwater." *Water Science and Technology*. Volume 20, number 3, pp. 47-53.
- Bedient, P.B., and Huber, W.C. *Hydrology and Floodplain Analysis*. New York: Addison-Wesley Publishing Co., 1992.
- Boggess, D.H. (1975). *Effects of a Landfill On Ground-Water Quality*. United States Department of the Interior Geological Survey, Open File Report 75-594. Prepared in cooperation with the City of Fort Myers. U.S. Government Printing Office, Washington, D.C.
- Bouwer, Edward J., Perry L. McCarty, Herman Bouwer and Robert C. Rice (1984). "Organic Contaminant Behavior During Rapid Infiltration of Secondary Wastewater at the Phoenix 23rd Avenue Project." *Water Resources*. Volume 18, number 4, pp. 463-472.
- Bouwer, Herman (1985). "Renovation of Wastewater with Rapid-Infiltration Land Treatment Systems." In: *Artificial Recharge of Groundwater*. Edited by Takashi Asano. Butterworth Publishers, Boston, pp. 249-282.
- Bouwer, Herman (1987). "Effect of Irrigated Agriculture on Groundwater." *Journal of Irrigation and Drainage Engineering, ASCE*. Volume 113, number 1, pp. 4-15.
- Bouwer, Herman and Emanuel Idelovitch (1987). "Quality Requirements for Irrigation with Sewage Water." *Journal of Irrigation and Drainage*. Volume 113, number 4, pp. 516-535.
- Bowman, D.C., J.L. Paul, W.B. Davis, and S.H. Nelson (1987). "Reducing Ammonia Volatilization from Kentucky Bluegrass by Irrigation." *Horticulture Science*. Vol. 22, pp. 84-87.
- Box, G.E.P., Hunter, W.G., and Hunter, J.S.. *Statistics for Experimenters*. John Wiley and Sons, New York, 1978.
- Butler, Kent S. (1987). "Urban Growth Management and Groundwater Protection: Austin, Texas." *Planning for Groundwater Protection*. Academic Press, Inc. New York. pp. 261-287.
- Chang, A.C., A.L. Page, P.F. Pratt and J.E. Warneke (1988). "Leaching of Nitrate from Freely Drained-Irrigated Fields Treated with Municipal Sludge." *Planning Now for Irrigation and Drainage in the 21st Century: Proceedings of a Conference Sponsored by the Irrigation and Drainage Division of the American Society of Civil Engineers*. Lincoln, Nebraska, pp. 455-467. ASCE, New York.
- 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.
- Close, M.E. (1987). "Effects of Irrigation on Water Quality of a Shallow Unconfined Aquifer." *Water Resources Bulletin*. Volume 23, number 5, pp. 793-802.
- Craun, Gunther F. (1979). "Waterborne Disease -- A Status Report Emphasizing Outbreaks in Ground-Water Systems." *Groundwater*. Volume 17, number 2, pp. 183-191.
- Crites, Ronald W.(1985). "Micropollutant Removal in Rapid Infiltration." In: *Artificial Recharge of Groundwater*. Edited by Takashi Asano. Butterworth Publishers, Boston, pp. 579-608.
- Das, B.M. *Principals of Geotechnical Engineering* Boston: PWS Publishing Co., 1994.
- Dickey-John Corporation. *Installation Instructions Soil Compaction Tester*. Auburn, Illinois, 1987.
- Domagalski, Joseph L. and Neil M. Dubrovsky. (1992). "Pesticide Residues in Groundwater of the San Joaquin Valley, California." *Journal of Hydrology*. Volume 130, number 1-4, pp. 299-338.
- Ehrlich, Garry G., Henry F.H. Ku, John Vecchioli and Theodore A. Ehlke (1979a). *Microbiological Effects of Recharging the Magothy Aquifer, Bay Park, New York, with Tertiary-Treated Sewage*. Geological Survey

- Professional Paper 751-E. Prepared in cooperation with the Nassau County Department of Public Works. USGS, Washington, D.C.
- Ehrlich, G.G., E.M. Godsy, C.A. Pascale and John Vecchioli (1979b). "Chemical Changes in an Industrial Waste Liquid during Post-Injection Movement in a Limestone Aquifer, Pensacola, Florida." *Groundwater*. Volume 17, number 6, November-December, pp. 562-573.
- EPA (1992). *Manual: Guidelines for Water Reuse*. EPA Document No. EPA/625/R-92/004. U.S. Government Printing Office, Washington, D.C.
- EPA (U.S. Environmental Protection Agency) (1983). *Results of the Nationwide Urban Runoff Program*. Water Planning Division, PB 84-185552, Washington, D.C.
- Ferguson, R.B., D.E. Eisenbauer, T.L. Bockstadter, D.H. Krull and G. Buttermore (1990). "Water and Nitrogen Management in Central Platte Valley of Nebraska." *Journal of Irrigation and Drainage*. Volume 116, number 4, pp. 557-565.
- Ford, M. and J.H. Tellam (1994). "Source, type and extent of inorganic contamination within the Birmingham urban aquifer system, UK." *Journal of Hydrology*. Volume 156, pp. 101-145.
- Gerba, Charles P. and Charles N. Haas (1988). "Assessment of Risks Associated with Enteric Viruses in Contaminated Drinking Water." *Chemical and Biological Characterization of Sludges, Sediments, Drudge Spoils and Drilling Muds, ASTM STP 976*. American Society for Testing and Materials, Philadelphia, Pennsylvania. pp. 489-494.
- German, Edward R. (1989). *Quantity and Quality of Stormwater Runoff Recharged to the Floridan Aquifer System Through Two Drainage Wells in the Orlando, Florida Area*. U.S. Geological Survey - Water Supply Paper 2344. Prepared in cooperation with the Florida Department of Environmental Regulation. USGS, Denver, Colorado.
- Gilbert, R.O., *Statistical Methods for Environmental Pollution Monitoring* New York: Van Nostrand Reinhold Publishing Co., 1987.
- Gold, A.J. and P.M. Groffman (1993). "Leaching of Agrichemicals from Suburban Areas." In: *Pesticides in Urban Environments - Fate and Significance*. Racke, K.D. and A.R. Leslie, editors. ACS Symposium Series No. 522. American Chemical Society, Washington, D.C.
- Goldshmid, J. (1974). "Water-Quality Aspects of Ground-Water Recharge in Israel." *Journal of the American Water Works Association*. Volume 66, number 3, pp. 163-166.
- Goolsby, Donald A. (1972). "Geochemical Effects and Movement of Injected Industrial Waste in a Limestone Aquifer." *Underground Waste Management and Environmental Implications (Reprint)*. The American Association of Petroleum Geologists, Memoir No. 18.
- Greene, Gerald E. (1992). "Ozone Disinfection and Treatment of Urban Storm Drain Dry-Weather Flows: A Pilot Treatment Plant - Demonstration Project on the Kenter Canyon Storm Drain System in Santa Monica." *Santa Monica Bay Restoration Project*. Office of the City Engineer, Santa Monica, California.
- Hampson, Paul S. (1986). *Effects of Detention on Water Quality of two Stormwater Detention Ponds Receiving Highway Surface Runoff in Jacksonville, Florida*. U.S. Geological Survey Water-Resources Investigations Report 86-4151. Prepared in cooperation with the Florida Department of Transportation. USGS, Denver, Colorado.
- Harper, Harvey H. (1988). *Effects of Stormwater Management Systems on Groundwater Quality*. Final Report for DER Project WM190. Florida Department of Environmental Regulation.
- 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.
- Hickey, John J. and John Vecchioli (1986). *Subsurface Injection of Liquid Waste with Emphasis on Injection Practices in Florida*. U.S. Geological Survey Water-Supply Paper 2281. USGS, Denver, Colorado.
- Higgins, Andrew J. (1984). "Impacts on Groundwater due to Land Application of Sewage Sludge." *Water Resources Bulletin*. Volume 20, number 3, pp. 425-434.
- Horsley, Scott W. and John A. Moser (1990). "Monitoring Groundwater for Pesticides at a Golf Course -- A Case Study on Cape Cod, Massachusetts." *Groundwater Monitoring Review*. Volume 10, pp. 101-108.
- Horton, R.E. "An approach toward a physical interpretation of infiltration capacity." *Transactions of the American Geophysical Union*. Vol. 20, pp. 693 – 711. 1939.
- Hütter, U. and F. Remmler (1996). "Stormwater infiltration at a site with critical subsoil conditions: Investigations of soil, seepage water and groundwater." *7th International Conference on Urban Drainage*. Hannover, Germany. Edited by F. Sieker and H-R. Verworn. International Association on Water Quality, London. pp. 713 – 718 .

- Jansons, Janis, Lindsay W. Edmonds, Brent Speight and Marion R. Bucens (1989a). "Movement of Viruses after Artificial Recharge." *Water Research*. Volume 23, number 3, pp. 293-299.
- Jansons, Janis, Lindsay W. Edmonds, Brent Speight and Marion R. Bucens (1989b). "Survival of Viruses in Groundwater." *Water Research*. Volume 23, number 3, pp. 301-306.
- 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.
- Jury, W.A., W.F. Spencer, and W.J. Farne (1983). "Model for Assessing Behavior of Pesticides and other Trace Organics using Benchmark Properties. I. Description of Model". *J. Environmental Quality*. No. 12. pp. 558-564.
- Knisel, W.G. and R.A. Leonard (1989). "Irrigation Impact on Groundwater: Model Study in Humid Region." *Journal of Irrigation and Drainage, ASCE*. Volume 115, number 5, pp. 823-838.
- Krawchuk, Bert P. and G.R. Barrie Webster (1987). "Movement of Pesticides to Groundwater in an Irrigated Soil." *Water Pollution Research Journal of Canada*. Volume 22, pp. 129-146. number 1.
- Ku, Henry F. H. and Dale L. Simmons (1986). *Effect of Urban Stormwater Runoff on Groundwater beneath Recharge Basins on Long Island, New York*. U.S. Geological Survey Water-Resources Investigations Report 85-4088. Prepared in cooperation with Long Island Regional Planning Board, Syosset, New York. USGS, Denver, Colorado.
- Lager, J.A. (1977). *Urban Stormwater Management and Technology: Update and User's Guide*. Rep No. EPA-600/8-77-014. U.S. Environmental Protection Agency, Cincinnati, Ohio. pp. 90-92.
- Lauer, D.A. (1988). "Vertical Distribution in Soil of Unincorporated Surface-Applied Phosphorus under Sprinkler Irrigation." *Soil Science Society of America Journal*. Volume 52, number 6, pp. 1685-1692.
- Lloyd, J.W., D.N. Lerner, M.O. Rivett and M. Ford (1988). "Quantity and Quality of Groundwater beneath an Industrial Conurbation -- Birmingham, UK." In: *Proceedings of the Conference -- Hydrological Processes and Water Management in Urban Areas*. Duisburg, Federal Republic of Germany. International Hydrological Programme, UNESCO, pp. 445-452.
- Martinelli, I. and E. Alfakih (1998). "Inclusive modelling methodology for urban stormwater infiltration and associated pollutants transfer." Presented at the *Fourth International Conference on Development in Urban Drainage Modelling*. Imperial College of Science, Technology & Medicine, London, UK. September 21 – 24, 1998. pp. 601 – 608. IAWQ. London.
- Marton, J. and I. Mohler (1988). "The Influence of Urbanization on the Quality of Groundwater." In: *Proceedings of the Conference -- Hydrological Processes and Water Management in Urban Areas*. Duisburg, Federal Republic of Germany. International Hydrological Programme, UNESCO, pp. 453-460.
- Marzouk, Yosef, Sagar M. Goyal and Charles P. Gerba (1979). "Prevalence of Enteroviruses in Groundwater of Israel." *Groundwater*. Volume 17, number 5, pp. 487-491.
- McCuen, R. *Hydrologic Analysis and Design, 2nd edition*. Prentice Hall. 1998.
- Merkel, B., J. Grossman and P. Udluft (1988). "Effect of Urbanization on a Shallow Quaternary Aquifer." *Proceedings of the Conference -- Hydrological Processes and Water Management in Urban Areas*. Duisburg, Federal Republic of Germany. International Hydrological Programme, UNESCO, pp. 461-468.
- Mikkelsen, P.S., H. Madsen, H. Rosbjerg, and P. Harremoës (1996a). "Properties of extreme point rainfall III: Identification of spatial inter-site correlation structure." *Atmospheric Research*.
- Mikkelsen, P.S., K. Arngjerg-Nielsen, and P. Harremoës (1996b). "Consequences for established design practice from geographical variation of historical rainfall data." *Proceedings: 7th International Conference on Urban Storm Drainage*. Hannover, Germany.
- Morel-Seytoux, H.J. "Derivation of Equations for Variable Rainfall Infiltration." *Water Resources Research*. pp. 1012-1020. August 1978.
- Mull, R. (1996). "Water exchange between leaky sewers and aquifers." *7th International Conference on Urban Storm Drainage*. Hannover, Germany. Sept. 9-13, 1996. Edited by F. Sieker and H-R. Verworn. IAHR/IAWQ. SuG-Verlagsgesellschaft. Hannover, Germany. pp. 695-700.
- Nellor, Margaret H., Rodger B. Baird and John R. Smyth (1985). "Health Aspects of Groundwater Recharge." In: *Artificial Recharge of Groundwater*. Edited by Takashi Asano. Butterworth Publishers, Boston, pp. 329-356.
- Nightingale, Harry I. and William C. Bianchi (1977). "Ground-Water Chemical Quality Management by Artificial Recharge." *Groundwater*. Volume 15, number 1, pp. 15-22.

- Nightingale, Harry I. (1987a). "Water Quality beneath Urban Runoff Water Management Basins." *Water Resources Bulletin*. Volume 23, number 2, pp. 197-205.
- Nightingale, Harry I. (1987b). "Accumulation of As, Ni, Cu and Pb in Retention and Recharge Basins Soils from Urban Runoff." *Water Resources Bulletin*. Volume 23, number 4, pp. 663-672.
- Norberg-King, Teresa J., Elizabeth J. Durhan, Gerald T. Ankley and Eric Robert (1991). "Application of Toxicity Identification Evaluation Procedures to the Ambient Waters of the Colusa Basin Drain, California." *Environmental Toxicology and Chemistry*. Volume 10, pp. 891-900.
- 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.
- Peterson, David A. (1988). "Selenium in the Kendrick Reclamation Project, Wyoming." *Planning Now for Irrigation and Drainage in the 21st Century*. Proceedings of a conference sponsored by the Irrigation and Drainage Division of the American Society of Civil Engineers, Lincoln, Nebraska, pp. 678-685. ASCE, New York.
- Petrovic, A. Martin (1990). "The Fate of Nitrogenous Fertilizers Applied to Turfgrass." *Journal of Environmental Quality*. Volume 19, number 1, pp. 1-14.
- Pierce, Ronald C. and Michael P. Wong (1988). "Pesticides in Agricultural Waters: The Role of Water Quality Guidelines." *Canadian Water Resources Journal*. Volume 13, number 3, pp. 33-49.
- 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., 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. 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.E., R. Field, M. Lalor, and M. Brown (1995). "Urban Stormwater Toxic Pollutants: Assessment, Sources, and Treatability." *Water Environment Research*. Vol. 67, No. 3, pp. 260-275.
- 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.
- Pitt, R., B. Robertson, P. Barron, A. Ayyoubi, and S. Clark (1999). *Stormwater Treatment at Critical Areas, Vol. 1: The Multi-Chambered Treatment Train (MCTT)*. U.S. Environmental Protection Agency, Water Supply and Water Resources Division, National Risk Management Research Laboratory. EPA/600/R-99/017. Cincinnati, Ohio. 492 pgs.
- 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, Volume 7*. (Edited by W. James). Computational Hydraulics International, Guelph, Ontario and Lewis Publishers/CRC Press. 1999.
- Pitt, R., S. Clark, and R. Field. "Groundwater contamination potential from stormwater infiltration practices." *Urban Water*: Vol. 1, no. 3, pp. 217-236. September 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.
- 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 1999b.

- Pitt, R., S. Clark, and R. Field. "Groundwater contamination potential from stormwater infiltration practices." *Urban Water*. Vol. 1, no. 3, pp. 217-236. September 1999c.
- Pitt, R. and J. Lantrip. "Infiltration through disturbed urban soils." In: *Advances in Modeling the Management of Stormwater Impacts, Volume 8*. (Edited by W. James). Computational Hydraulics International, Guelph, Ontario. 2000.
- Power, J.F. and J.S. Schepers (1989). "Nitrate Contamination of Groundwater in North America." *Agriculture, Ecosystems and Environment*. Volume 26, pp. 165-187. number 3-4.
- Pruitt, Janet B., David E. Troutman and G.A. Irwin (1985). *Reconnaissance of Selected Organic Contaminants in Effluent and Groundwater at Fifteen Municipal Wastewater Treatment Plants in Florida, 1983-84*. U.S. Geological Survey Water-Resources Investigation Report 85-4167. Prepared in cooperation with the Florida Department of Environmental Regulation. USGS, Denver, Colorado.
- Racke, K.D. and A.R. Leslie, editors (1993). *Pesticides in Urban Environments - Fate and Significance*. ACS Symposium Series No. 522. American Chemical Society, Washington, D.C.
- Ragone, Stephen E. (1977). *Geochemical Effects of Recharging the Magothy Aquifer, Bay Park, New York, with Tertiary-Treated Sewage*. Geological Survey Professional Paper 751-D. Prepared in cooperation with the Nassau County Department of Public Works. USGS, Washington, D.C.
- Razack, M., C. Drogue and M. Baitelem (1988). "Impact of an Urban Area on the Hydrochemistry of a Shallow Groundwater (Alluvial Reservoir) Town of Narbonne, France." *Proceeding of the Conference -- Hydrological Processes and Water Management in Urban Areas*. International Hydrological Programme, UNESCO, pp.487-494. Duisburg, Federal Republic of Germany.
- Reichenbaugh, R.C. (1977). *Effects on Ground-Water Quality from Irrigating Pasture with Sewage Effluent Near Lakeland, Florida*. U.S. Geological Survey Water-Resources Investigations Report 76-108. Prepared in cooperation with Southwest Florida Water Management District. USGS, Denver, Colorado.
- Rice, R.C., D.B. Jaynes and R.S. Bowman (1991). "Preferential Flow of Solutes and Herbicide under Irrigated Fields." *Transactions of the American Society of Agricultural Engineers*. Volume 34, number 3, pp. 914-918.
- Ritter, W.F., F.J. Humenik and R.W. Skaggs (1989). "Irrigated Agriculture and Water Quality in the East." *Journal of Irrigation and Drainage Engineering, ASCE*. Volume 115, number 5, pp. 807-821.
- Ritter, W.F., R.W. Scarborough and E.M. Chirnside (1991). "Nitrate Leaching under Irrigation on Coastal Plain Soil." *Journal of Irrigation and Drainage Engineering, ASCE*. Volume 117, number 4, pp. 490-502.
- Robinson, J. Heyward and H. Stephen Snyder (1991). "Golf Course Development Concerns in Coastal Zone Management." *Coastal Zone '91: Proceedings of the Seventh Symposium on Coastal and Ocean Management*. Long Beach, California, pp. 431-443. ASCE, New York.
- Sabatini, David A. and T. Al Austin (1988). "Adsorption, Desorption and Transport of Pesticides in Groundwater: A Critical Review." *Planning Now for Irrigation and Drainage in the 21st Century*. Proceedings of a Conference Sponsored by the Irrigation and Drainage Division of the American Society of Civil Engineers, Lincoln, Nebraska, pp. 571-579. ASCE, New York.
- Sabol, George V., Herman Bouwer and Peter J. Wierenga (1987). "Irrigation Effects in Arizona and New Mexico." *Journal of Irrigation and Drainage Engineering, ASCE*. Volume 113, number 1, pp. 30-57.
- Salo, John E., Doug Harrison and Elaine M. Archibald (1986). "Removing Contaminants by Groundwater Recharge Basins." *Journal of the American Water Works Association*. Volume 78, number 9, pp. 76-81.
- Schiffer, Donna M. (1989). *Effects of Three Highway-Runoff Detention Methods on Water Quality of the Surficial Aquifer System in Central Florida*. U.S. Geological Survey Water-Resources Investigations Report 88-4170. Prepared in cooperation with the Florida Department of Transportation. USGS, Denver, Colorado.
- Schmidt, Kenneth D. and Irving Sherman (1987). "Effect of Irrigation on Groundwater Quality in California." *Journal of Irrigation and Drainage Engineering, ASCE*. Volume 113, number 1, pp. 16-29.
- Schneider, Brian J., Henry F.H. Ku and Edward T. Oaksford (1987). *Hydrologic Effects of Artificial-Recharge Experiments with Reclaimed Water at East Meadow, Long Island, New York*. U.S. Geological Survey Water-Resources Investigations Report 85-4323. Prepared in cooperation with the Nassau County Department of Public Works, Syosset, New York. USGS, Denver, Colorado.
- Seaburn, G.E. and D.A. Aronson (1974). *Influence of Recharge Basins on the Hydrology of Nassau and Suffolk Counties, Long Island, New York*. U.S. Geological Survey Water-Supply Paper 2031. USGS, Washington, D.C.
- Shirmohammadi, A. and W.G. Knisel (1989). "Irrigated Agriculture and Water Quality in the South." *Journal of Irrigation and Drainage Engineering*. ASCE, 115 (5) 791-806.

- Smith, Sheldon O. and Donald H. Myott (1975). "Effects of Cesspool Discharge on Ground-Water Quality on Long Island, N.Y." *Journal of the American Water Works Association*. Volume 67, number 8, pp. 456-458.
- Spalding, Roy F. and Lisa A. Kitchen (1988). "Nitrate in the Intermediate Vadose Zone Beneath Irrigated Cropland." *Groundwater Monitoring Review*. Volume 8, number 2, pp. 89-95.
- Steenhuis, Tammo, Robert Paulsen, Tom Richard, Ward Staubitz, Marc Andreini and Jan Surface (1988). "Pesticide and Nitrate Movement under Conservation and Conventional Tilled Plots." *Planning Now for Irrigation and Drainage in the 21st Century*. Proceedings of a conference sponsored by the Irrigation and Drainage Division of the American Society of Civil Engineers, Lincoln, Nebraska, pp. 587-595. ASCE, New York.
- Tim, Udoyara S. and Saied Mostaghim (1991). "Model for Predicting Virus Movement through Soil." *Groundwater*. Volume 29, number 2, pp. 251-259.
- Treweek, Gordon P. (1985). "Pretreatment Processes for Groundwater Recharge." In: *Artificial Recharge of Groundwater*. Edited by Takashi Asano. Butterworth Publishers, Boston, pp. 205-248.
- Troutman, D.E., E.M. Godsy, D.F. Goerlitz and G.G. Ehrlich (1984). *Phenolic Contamination in the Sand-and-Gravel Aquifer from a Surface Impoundment of Wood Treatment Wastewaters, Pensacola, Florida*. U.S. Geological Survey Water-Resources Investigations Report 84-4230. Prepared in cooperation with the Florida Department of Environmental Regulation. USGS, Denver, Colorado.
- 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.
- Vaughn, J.M., E.F. Landry, L.J. Baranosky, C.A. Beckwith, M.C. Dahl and N.C. Delihias (1978). "Survey of Human Virus Occurrence in Wastewater Recharged Groundwater on Long Island." *Applied and Environmental Microbiology*. Volume 36, number 1, pp. 47-51.
- Waller, Bradley, Barbara Howie and Carmen R. Causaras (1987). *Effluent Migration from Septic Tank Systems in Two Different Lithologies, Broward County, Florida*. U.S. Geological Survey Water Resources Investigations Report 87-4075. Prepared in cooperation with Broward County. USGS, Denver, Colorado.
- Wanielista, Martin P., Julius Charba, John Dietz, Richard S. Lott and Bryon Russell (1991). *Evaluation of the Stormwater Treatment Facilities at the Lake Angel Detention Pond, Orange County, Florida*. State of Florida Department of Transportation Environmental Research Paper FL-ER-49-91. National Technical Information Service, Springfield, VA.
- Wellings, F.M. (1988). "Perspective on Risk of Waterborne Enteric Virus Infection." *Chemical and Biological Characterization of Sludges, Sediments, Dredge Spoils, and Drilling Muds*. ASTM STP 976. American Society for Testing and Materials, Philadelphia, Pennsylvania, pp. 257-264.
- White, Everett M. and James N. Dornbush (1988). "Soil Change Caused by Municipal Waste Water Applications in Eastern South Dakota." *Water Resources Bulletin*. Volume 24, number 2, pp. 269-273.
- Wilde, F. D. (1994). *Geochemistry and Factors Affecting Ground-Water Quality at Three Storm-Water-Management Sites in Maryland: Report of Investigations No. 59*. Department of Natural Resources, Maryland Geological Survey, Baltimore, Maryland. (Prepared in Cooperation with the U.S. Department of the Interior Geological Survey, The Maryland Department of the Environment, and The Governor's Commission on Chesapeake Bay Initiatives).
- 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.
- Wilson, L.G., M.D. Osborn, K.L. Olson, S.M. Maida and L.T. Katz (1990). "The Groundwater Recharge and Pollution Potential of Dry Wells in Pima County, Arizona." *Groundwater Monitoring Review*. Volume 10, pp. 114-121.