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Contamination of Soil and Groundwater Due to Stormwater Infiltration Practices

A Literature Review

By

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Executive Summary

Recently, there has been an increased interest in the use of infiltration as a method of managing stormwater. Infiltration practices promote groundwater recharge, reduce runoff peak flows and volumes, and can lessen the transport of non-point source pollutants to surface water bodies. However, because stormwater infiltration systems are designed to discharge runoff into the soil, there has been concern that pollutants present in stormwater could contaminate groundwater wells. Thus, to understand the relative risks and benefits of infiltration, the fate of stormwater pollutants must be well understood. The fate of contaminants infiltrated from stormwater runoff and the potential for groundwater contamination was investigated by reviewing literature published in peer-reviewed scientific and engineering journals. This review examines common stormwater infiltration techniques, priority pollutants in urban stormwater runoff, and investigates the fate of these pollutants after infiltration.

Priority pollutants in urban stormwater runoff include nutrients (i.e. nitrogen and phosphorus), heavy metals (i.e. Pb, Zn, Cu, Cd), organics (e.g. petroleum hydrocarbons), pathogens, suspended solids, and salts. The potential for groundwater contamination is a complex function of soil and contaminant properties and the depth to the water table. Karst geology in particular can provide pathways for rapid and extensive groundwater contamination from infiltration systems.

Heavy metals are often present at very low levels in urban stormwater. Fortunately, studies have demonstrated that metals are generally retained in the upper soil layers via adsorption to solid particles. However, eventual breakthrough can occur due to the finite sorption capacities of the soil media. Periodic replacement of the upper soil layer within infiltration systems has been suggested as a method of preventing possible groundwater contamination and maintaining low soil concentrations.

Suspended solids are usually removed via straining by the soil. Because they pose little health risk, suspended solids are mainly a concern because they may clog the infiltration system. Phosphorus and nitrogen can also be removed within the soil media; phosphorus by precipitation or adsorption reactions and nitrogen by bacterial denitrification. Phosphorus is a concern because excess quantities cause eutrophication of surface water bodies. Studies have shown varied results regarding phosphorus removal via infiltration. Nitrates present in drinking water supplies can pose a health concern to certain target groups (fetuses, infants). Most studies indicated that nitrate is poorly retained in infiltration devices due to high solubility. However, the low levels typically found in urban stormwater make nitrate pollution a low concern (most problems are associated with ammonia-based agricultural fertilizers).

Anthropogenic organic pollutants, such as petroleum hydrocarbon residues, are typically present at low levels in urban runoff. There have been only a few published studies which have examined the fate of these compounds in stormwater, but the limited results appear promising. Many organic pollutants, such as oils and gasoline, have a high soil affinity and can also be biodegraded. Degradation rates and the contaminant capacity of the soil, however, have largely been unexplored. Some organic compounds are less likely to be retained by the soil, and certain

practices (such as subsurface injection) have been documented to increase the risk of groundwater contamination. Subsurface injection provides a more direct conduit to groundwater, and does not allow infiltration through the aerobic vadose zone where biodegradation is enhanced.

Few studies have examined the efficacy of infiltration practices for pathogenic organism (e.g. fecal coliform, viruses, and other bacteria) removal. However, the outlook appears to be positive, in that pathogens can be physically strained by the soil similar to a sand filter at a drinking water treatment plant. However, documented cases of bacterial contamination of groundwater wells exist; certain practices (e.g. subsurface injection) may increase the risks. Pathogens may move vertically and/or horizontally with subsurface water flow and survive for days. The fate and survival of pathogens depends upon multiple parameters and is not thoroughly understood. Contamination of groundwater by pathogens has been documented and thus cannot be ignored.

Finally, it is known that soil media has no appreciable retention of salts. Thus, salts have a high potential for groundwater contamination and documented cases of groundwater contamination by salts exist. Placement of the infiltration device largely dictates the influence of saline pollution.

In summary, increased application of stormwater infiltration practices necessitates examination of possible contamination to soil and groundwater—a legitimate concern for the protection of human and environmental health. This review provides a valuable synopsis of the state of current research regarding stormwater infiltration and the associated possibilities for contamination. Although a fair number of studies in this pioneering field are available, some areas have been neglected and most warrant further study. Therefore, the appropriate information regarding the pollution risks associated with choosing infiltration—and the often greater pollution risks of *not* choosing infiltration—must be available to optimize and execute appropriate water resources management decisions.

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1. Introduction

1.A. Background

Non-point source pollution from stormwater runoff is well-documented as a leading cause of impairment of freshwater lakes, rivers, and estuaries (U.S. EPA, 2000; U.S. EPA, 2005). When impervious surfaces such as roads, parking lots, and rooftops replace areas that previously allowed infiltration of stormwater, the resulting stormwater runoff has typically been conveyed to storm or sanitary sewers which may act as conduits that carry pollutants (e.g., sediments, nutrients, metals, petrochemicals) to receiving water bodies. Stormwater management is an issue of importance to the health of the general public and environment; thus municipalities throughout the nation have been seeking improved methods of managing stormwater. One increasingly popular technique is Low Impact Development (LID). LID is gaining popularity because it promotes more sustainable water resources management while recognizing the needs of economic growth within local communities (Coffman, 2002). Additionally, LID may also benefit air quality and the quality of life (Coffman, 2002). Several modeling experiments have shown that LID—when properly implemented—is capable of restoring nearly the predevelopment hydrologic regime (Brander et al., 2004; Holman-Dodds et al., 2003).

Historically, stormwater management has consisted of reducing the peak flow of runoff from developed watersheds with little or no thought given to water quality. Methods used to accomplish this goal have often involved the construction of detention ponds which, although they reduce peak flows and can remove a fraction of solid particles, have been shown to be inadequate at addressing ecological stream degradation (Booth et al., 2002). With greater attention now being given to water quality issues, alternative stormwater management approaches within the framework of LID are being implemented. LID seeks to reduce the volume of runoff from developed sites while focusing on both water quantity and water quality. Alternative stormwater management utilizes small, decentralized infiltration structures to mimic the predevelopment hydrologic regime (US EPA, 2005). Examples of alternative stormwater management techniques being implemented include greenroofs, infiltration trenches, constructed wetlands, and rain gardens. Greenroofs are planted atop buildings and slow runoff flows while enabling evapo-transpiration (Teemusk and Mander, 2007). Infiltration trenches are underground chambers that store runoff and allow it to infiltrate into the existing soil. Initially, constructed wetlands had been utilized to treat municipal wastewater, but more recently they have also been used to treat stormwater (Walker and Hurl, 2002; Schutes et al., 1997). Rain gardens are shallow vegetated depressions into which stormwater is directed for recharge (US EPA, 2000). These techniques, among others, are collectively known as stormwater Best Management Practices (BMPs) in protecting water quality (Clary et al., 2002).

Many alternative stormwater management techniques rely on infiltration of stormwater into the soil where sub-surface flow and groundwater recharge may occur. This provides a reduction in runoff quantity and may promote pollutant removal through physical, chemical, and biological means. Current infiltration practices include rain gardens, bioretention systems, and infiltration basins and trenches. Other stormwater management techniques may not use infiltration as their primary treatment method but do infiltrate stormwater. For example, a standard detention pond relies primarily on sedimentation to remove contaminants but infiltration usually occurs through the bottom and sides of the pond.

With the rise in popularity of stormwater management techniques that infiltrate polluted runoff, concern has arisen regarding potential groundwater contamination. After a summary of common stormwater infiltration practices and pollutants typically found in urban stormwater, this paper provides a literature review of existing work that has investigated the fate of stormwater pollutants once infiltration has occurred and the potential for groundwater contamination.

For more detailed information on groundwater contamination resulting from stormwater infiltration, please see Pitt et al., (1996) which is referenced several times in the following sections.

1.B Stormwater Infiltration Practices

As previously discussed, in an attempt to reduce runoff volumes, many stormwater management practices seek to infiltrate stormwater runoff into the soil where it can be transported by subsurface flow and/or recharge groundwater aquifers. For example, bioretention systems, rain gardens, and infiltration trenches are all designed based on a desired volume of infiltration. Other practices, such as detention ponds, have typically been designed based on the desired reduction in the peak runoff flow rate and, when considering only water quantity, are often assumed to have no infiltration capacity. While this is an acceptable and conservative assumption when considering peak flows, pond infiltration cannot be ignored when considering the fate of pollutants and potential groundwater contamination. Thus, this paper reviews information on stormwater management techniques that rely primarily on infiltration as well as other techniques, such as detention ponds, whose primary function or processes are not infiltration but that have the capacity to infiltrate stormwater. A summary of such practices is given below.

1.B.1 Infiltration Basins

Infiltration basins are constructed with the intent of storing and infiltrating stormwater runoff up to a targeted design volume. As defined in the “Assessment of Stormwater Best Management Practices” manual published by the University of Minnesota (UM, 2007):

“An infiltration basin is a natural or constructed impoundment that captures, temporarily stores, and infiltrates the design volume within an acceptable time period. Infiltration basins contain a flat, densely vegetated floor situated over naturally permeable soils. Nutrients and pollutants are removed from the infiltrated stormwater through chemical, biological, and physical processes. Infiltration basins are well suited for drainage areas of 5 to 50 acres (2.03–20.25 hectares) with land slopes that are less than 20 percent, with typical depths in the basin ranging from 2 to 12 feet (0.61–3.66 meters).”

Infiltration basins often require relatively large land areas and, with well chosen vegetation, are often aesthetically pleasing.

1.B.2 Infiltration Trenches

The primary purpose of infiltration trenches is to collect stormwater and reduce runoff volumes by allowing the water to infiltrate into the surrounding soil. The “Assessment of Stormwater Best Management Practices” manual defines infiltration trenches as follows:

“An infiltration trench is a shallow excavated trench, typically 3 to 12 feet deep (0.91–3.66 meters), that is backfilled with a coarse stone aggregate allowing for the temporary storage of runoff in the void space of the material. Discharge of this stored runoff occurs through infiltration into the surrounding naturally permeable soil. Infiltration trenches are well suited for drainage areas of 5 acres (2.03 hectares) or less.”

1.B.3 Porous Pavements

The primary purpose of porous pavements is to reduce runoff volumes by allowing stormwater to pass through the pavement structure and infiltrate into the underlying soil. While porous asphalt and concrete are the most obvious varieties of porous pavements, Ferguson (2005) lists a total of nine categories with this classification. These include porous aggregate, porous turf, plastic geocells, open-jointed paving blocks, open-celled paving grids, porous concrete, porous asphalt, soft porous surfacing, and decks. For the case where the porous pavement is either asphalt or concrete, the pavement system is designed such that storm water infiltrates through the porous upper pavement layer and then into a reservoir of stone or rock below. Water from the reservoir then either percolates into the underlying soil or is collected by a perforated pipe underdrain system and carried to a surface discharge location.

Porous pavements are gaining in popularity; however, their use is sometimes met with (not necessarily valid) concerns of increased maintenance costs and decreased durability.

1.B.4 Rain Gardens

Rain gardens are low lying areas, natural or excavated, that are planted with vegetation and receive stormwater runoff from nearby impervious surfaces via stormwater conveyances, such as curb cuts. The collected stormwater exits the rain garden primarily via infiltration, reducing runoff volume and potentially recharging groundwater. Alternatively, some rain gardens are equipped with underdrains and are typically used when the underlying soil has a low infiltration capacity. Such rain gardens are constructed by excavating the soil, placing a drain tile or perforated pipe collection system at the bottom, backfilling with high permeability soil, and then planting with vegetation. In these systems the collection pipe discharges the water out of the rain garden and groundwater contamination is most likely of little concern.

1.B.5 Swales and Filter Strips

Swales are vegetated canals or trenches that convey stormwater, filter and/or settle solids, and infiltrate a portion of the runoff. Other names for swales are ditches, grassed channels, dry swales, vegetated swales, wet swales, biofilters, or bioswales. Permeable structures (e.g., check dams) are sometimes installed in the flow path to reduce flow velocities and increases infiltration volumes.

Filter strips are vegetated areas specifically designed and positioned for overland sheet flow of stormwater runoff. The vegetation filters particulate pollutants and reduces runoff velocities, which increases infiltration. Filter strips may also be called buffer strips or buffers. Sheet flow is required for filter strips to effectively treat stormwater runoff.

1.B.6 Detention Ponds

Detention ponds are depressions in the soil surface that are designed to collect and store stormwater runoff. Dry detention ponds have their main outlet elevation at the pond bottom and are typically designed to discharge all stormwater within 48 hours of the end of a runoff event. Historically, dry detention ponds have been designed solely on the basis of peak flow reduction. More recently, however, dry detention ponds, which can settle an appreciable amount of solids, have been designed with water quality as a driving factor. In both cases, significant infiltration can occur if the pond is not lined with an impermeable membrane or clay barrier.

Wet detention ponds have their main outlet above the pond bottom and are designed to store a portion of the runoff volume long after the runoff event ends with the intent of infiltrating a portion of the runoff volume.

2. Stormwater Pollutants and their Fate in Infiltration Systems

Urban stormwater runoff can originate from roads, buildings, houses, lawns, industrial parks, and a host of other human-made structures. Many studies on the contaminants and the concentration ranges common to urban runoff have been performed (Moxness 1986, 1987, 1988; Driscoll et al. 1990; Oberts 1994; Barrett et al. 1995; Stanley 1996; Wu et al. 1996; Sansalone and Buchberger 1997; Barrett et al. 1998; Anderle 1999; Legret and Colandini 1999; Waschbusch et al. 1999; Carleton et al. 2000; Drapper et al. 2000; Brezonik and Stadelmann 2002; Harper et al. 1999). Typical urban runoff contaminants include nutrients, heavy metals, suspended solids, petroleum hydrocarbons, pathogens, and salts. Studies have shown that contaminant concentrations in urban runoff can vary widely by season, location, traffic volumes, and rainfall volumes and intensity. If infiltrated into the soil, these contaminants have the potential to degrade soil and groundwater quality and, therefore, are of concern. A discussion of each of these contaminants follows.

2.A Nutrients

The most common and influential forms of nutrient pollution to stormwater are phosphorous and nitrogen, the latter of which may be in many forms. Nutrients are of concern because of their ability to cause algal blooms and consequential eutrophication in receiving water bodies. Nitrate is a common contaminant in groundwater but phosphorus contamination has not been as common or as severe. Nutrient contamination of groundwater is, however, not always caused by stormwater infiltration. For example, in areas of some sedimentary deposits, ammonium in the soil can be oxidized to nitrate (Pitt et al., 1999). Some common nutrient sources include animal waste, septic leakage, fertilizers, and atmospheric deposition (U.S. EPA, 1999). While typical stormwater nutrient loadings are significantly less than that of treated wastewater they can be higher under some wet weather conditions (U.S. EPA, 1999). Event mean concentrations vary seasonally and by land use; correlations between these parameters and nutrient flux have been ascertained but are weak because of the complexity inherent in non-point source pollution (Brezonik and Stadelmann, 2002).

2.A.1 Phosphorous

Phosphorous is of concern because it can cause algal blooms and subsequent eutrophication to receiving water bodies where P is the limiting nutrient (Mihelcic, 1999). Sources of phosphorus include motor oil (Pitt et al., 1999), animal wastes and remains, plant material, and fertilizers. Orthophosphate is the form of P most readily available (U.S. EPA, 1999) to aquatic life and is the most common form occurring in stormwater. It has been found that phosphorous loading is correlated to intensity of urban land use. Phosphorous was found to increase logarithmically with impervious surface area in construction using curb-and-gutter style stormwater management. No such trend was observed for a comparable LID development (Dietz and Clausen, 2008). In this study, traditional development produced a phosphorous export rate of 2 kg/ha/yr, while 0.4 kg/ha/yr was observed in the LID development.

Once in the soil media, orthophosphate (PO_4^{-3}) can be removed from infiltrated stormwater via precipitation or chemical adsorption to onto soil particle surfaces through reactions with iron, calcium, or aluminum. The dominant precipitation reactions are pH dependent and typically form iron and aluminum phosphates under low pH conditions and calcium phosphates under high pH conditions. In neutral conditions the precipitation reactions are rate-limited, thus the solubility appears to be higher than the solubility in basic or acidic conditions (Pitt et al., 1999).

Wu et al. (1996) conducted a study of stormwater pollutants in urban wet-detention ponds, and found removal rates ranging from -55% (phosphorus was released) to 100%. Dietz and Claussen (2005, 2006) also found negative phosphorus removal rates. In the latter study, a rain garden field site was constructed in a residential area of Connecticut with roof runoff directed to it and an underdrain included for effluent sample collection. It was found that instead of retaining phosphorus, the rain garden was a source of phosphorus. One possible cause was partial breakdown of new plant materials in the new rain garden, resulting in a short term P release. When monitored over time, however, the investigators found the influent-effluent differential to decay over time and expected a steady-state equilibrium to be reached. It was found that the highest phosphorus retention occurred in the mulch of the rain garden while the soil media retained the nutrients at much lower levels. Plants assimilated only 3% of the total P entering the

test cell. Overall, due to leaching of phosphorus, it was concluded that long term phosphorous retention would be minimal.

One column study found reductions in phosphorous equaling approximately 70% when synthetic stormwater was fed to a column of media designed to mimic a rain garden (Davis et al., 2001). Subsequent studies, however, have revealed much lower P retention rates, 41-48% (Hsieh and Davis, 2005). Hsieh and Davis (2005) predicted that addition of mulch would aid in removal by retaining P complexes, but found that this was largely not the case. Different media compositions were tested because it was hypothesized that silt/clay complexes would increase phosphorous sorption. It was found, however, that P removal did not correlate to media composition. It was also determined that sorbed phosphorus leached out the bottom of the columns.

In contrast, some studies have found media composition to be influential to P retention (Arias et al., 2001). After studying a subsurface flow constructed wetland for phosphorous retention, it was determined that chemical properties of the granular media can have a large impact on removal efficiencies. Under neutral and slightly basic conditions, higher calcium content increased P removal via precipitation. Under more acidic conditions, iron and aluminum composition in the sand may be influential. Phosphorus was also found to be incorporated into biofilms and plants in addition to being sorbed. Erickson et al. (2007) found that adding chopped granular steel wool to soil media can increase the removal of dissolved phosphorus without negatively impacting the quality of effluent.

2.A.2 Nitrogen

Sources of nitrogen in stormwater may be natural or anthropogenic. Nitrogen in the atmosphere can be in the form of nitrate which results from combustion or ammonium which may originate from volatilization of ammonia in soils, animal wastes, or other sources (Pitt et al., 1996). In urban areas, the major source of groundwater nitrogen contamination is from road runoff.

Nitrogen in stormwater may be present in many forms with ammonia (NH_4^+) being most toxic to aquatic life. Nitrate (NO_3^-) and nitrite (NO_2^-) are other common inorganic forms, and although nitrate is highly soluble (Pitt et al., 1999), very little nitrite is usually found in urban stormwater (U.S. EPA, 1999). High nitrate concentrations have, however, been found in some heavily industrialized areas and groundwater contamination due to stormwater infiltration has been documented (Pitt et al., 1999). Total Kjeldahl nitrogen (TKN) measures ammonia and organic forms. The nitrate standard for drinking water is 10 mg/L due to negative health impacts upon fetuses/infants (U.S. EPA, 2008). Because of the health effects and solubility in groundwater, nitrate is often the most examined form of nitrogen groundwater pollution. According to Pitt et al. (1996) nitrate, due to its typical low concentrations, has low to moderate groundwater contamination potential for surface percolation and subsurface infiltration/injection systems. If nitrate concentrations are high, however, the groundwater contamination potential would also be high.

Nitrogen in stormwater may be used by plants, but if it is not, it can travel towards the groundwater with some being removed by the soil in processes that depend on a complex series of variables. The movement of nitrate towards the groundwater is dependent on the rate and volume of infiltration, horizontal and vertical groundwater flow, depth to the water table, and preferential flow paths.

In areas with traditional development (i.e. no LID), nitrate export was found to increase logarithmically with increased impervious area. In LID areas, nitrate export did not correlate with impervious surface area (Dietz and Clausen, 2008). For ammonia-N, nitrate concentrations were found to be lower in LID developments compared to non-LID developments. Total N export from traditional developments averaged 10kg/ha/yr and 2 kg/ha/yr for LID developments.

One column study in which stormwater was allowed to pass through a soil media found TKN was reduced 65% to 75% and ammonia reduced 60-80% (Davis et al., 2001). However, nitrate concentrations within the column media and in the effluent increased. This increase was determined to be most likely attributable to biological activity. In subsequent studies, material was added to serve as organic and inorganic electron donors, forming saturated layers and creating an anaerobic environment to biologically remove nitrates via denitrification (Kim et al., 2003). Of the material tested, it was found that newspaper clippings were most successful at increasing denitrification. Other materials tested included sawdust, elemental sulfur, alfalfa, straw, and woodchips. Pilot scale studies indicated overall nitrate-nitrite retention of 70-80%.

Another study examined the influence of media upon nitrate removal (Hsieh and Davis, 2005). Nitrate removal in columns was found to range from 1-43% with ammonia removal rates of 2-26%. Mulch was found to be most effective at removing nitrates and ammonia, followed by native soil. Sand was determined to be least effective. Removal efficiency was not found to correlate with clay or silt composition. During a field component of this study, six rain gardens were examined for nitrogen retention with all having less than 10% removal.

Other researchers have had similar results. At a field rain garden constructed for a pollutant removal study, less than 36% retention of total nitrogen, TKN, organic nitrogen, and nitrate was observed (Dietz and Clausen, 2005). Ammonia, however was retained at 85%. Another study examined the use of a saturated mulch layer in an attempt to increase denitrification (Dietz and Clausen, 2006). With the added saturated layer (previously discussed) in the rain garden, nitrate effluent concentrations were significantly reduced as was the ammonia in the rain garden. During the entire study period, 51% of TN was retained (33% by mulch and 0.3% by plants). Mulch TN increased, while soil TN decreased throughout the study period. Ammonia was found to have the highest retention rate (86 in the unsaturated layer and 69% in the saturated material), with nitrification and adsorption being the primary removal mechanisms. Hunt et al. (2006) similarly found total nitrogen mass removal rates of 40 percent, with nitrate removal being highly variable (13 to 75 percent). Sharkey and Hunt (2005) also experimented with a saturated layer for nitrate removal and observed a 77 percent decrease; however, TKN and ammonia levels increased.

Efforts have been made to engineer bioretention facilities to improve nitrate removal via denitrification; some results have shown some improvement when a carbon source was added

(Kim et al., 2003), while others had no statistically significant improvement (Dietz and Clausen, 2006).

2.B Heavy Metals

According to Pitt et al. (1996), nickel and zinc would have high groundwater contamination potential in infiltration/injection systems and chromium and lead, moderate potential. Pitt et al. (1996) assert that if sedimentation pretreatment were used, all metals would probably have low groundwater contamination potential.

Lead, zinc, copper, and cadmium are the primary heavy metals of concern in urban stormwater runoff (Weiss et al., 2006); each is discussed below. At high levels, heavy metals are considered a threat to human health and the environment; different organisms have different tolerances to different metals. Heavy metals are present in stormwater in dissolved phases, but a large fraction of most metals are usually bound to suspended solids (Davis and McCuen, 2005; Marsalek et al., 2001). Of the heavy metals typically found in stormwater, lead has largest tendency to adsorb to solids. The ranking of adsorption potential for some common heavy metals to soil particles is, with lead having the highest potential, as shown below (Pitt et al., 1995).

Lead>copper>nickel>cobalt>zinc>cadmium

Other metal removal mechanisms include precipitation, occlusion with other precipitates, diffusion into solid particles, biological uptake (Pitt et al., 1999).

Since heavy metals are often bound to solid particles, removal of suspended solids can be an effective method of reducing heavy concentrations in stormwater. Heavy metals do not degrade, however, and stormwater loading into rain gardens (or other stormwater structures) with subsequent detention will result in metal accumulation. Using stormwater pollutant loading and soil capacity estimates, Davis et al. (2003) estimated that, after 20 years, concentrations of cadmium, lead, and zinc would reach or exceed levels permitted by EPA biosolids land application regulations.

Assuming that all metals are retained in a soil layer, the soil metal concentration, C_s (metal mass/soil mass), can be estimated by (Marsalek et al., 2001):

$$C_s = C_w \cdot \frac{A_r}{A_i} \cdot \frac{MAR}{d \cdot \rho} \cdot t$$

where C_w is the concentration of metal in the runoff water, A_r/A_i is the ratio of runoff area catchment to infiltration area, MAR is the mean annual rainfall, d is the thickness of the soil layer, ρ is the soil bulk density, and t is time in years. Consistent units must be used when applying this equation.

Once sorbed to the surface of a soil particle, metals may not remain stationary. Depending on soil conditions such as pH, metals may be released from the solid surface. The relative mobility of stormwater heavy metals in soil is (Pitt et al., 1995):

Zinc>lead>cadmium>copper>nickel

Some heavy metals are also micronutrients needed by plants and may be accumulated into plant biomass as the plant grows. Some plants uptake metals at a much higher rate than others (known as hypoaccumulators) while others have greater tolerances to high heavy metal concentrations. Ideally, if used for removal of a heavy metal from stormwater or soils, a plant would have a high uptake rate and a tolerance to high metal concentrations within the plant material. Sun and Davis (2007) ranked the following regarding the general tendency of metals to accumulate in plants (Sun and Davis, 2007), with zinc having the highest tendency, as:

Zinc>Copper>lead>cadmium

2.B.1 Lead

Lead can originate from a variety of sources. Davis and Burns (1999) studied lead contribution of buildings to stormwater by spraying synthetic rainwater on buildings of various ages. The lead concentrations in the runoff were several orders of magnitude higher than drinking water standards. The study reported that the major source of lead was older weathered paints and that most lead was in the particulate form and could potentially be removed by filtration. The EPA action level for lead in drinking water is 15 µg/l (U.S. EPA, 2008).

Wu et al. (1998) examined the presence of lead in highway runoff in Charlotte, NC and found that the event mean concentration for each of the 10 storms monitored ranged from 6 to 15 mg/l. Increases in lead levels at the study sites correlated to increases in the percentage of impervious surface area and traffic intensity (number of average daily trips). Pitt et al. (1996) stated that lead concentrations are generally highest in runoff from streets and parking areas.

In a groundwater contamination study in France, stormwater from a mostly industrial area was found to have lead concentrations between less than 5 and 90 µg/l (Bardin et al., 2001). In another French study on infiltration practices, lead levels in stormwater from a heavily used roadway prior to recharge were found to average 98 µg/l (Barraud et al., 1999).

The main removal mechanisms of lead from stormwater are sorption to solids, ion exchange, and precipitation (Pitt et al., 1995). In three different studies, synthetic runoff (containing a suite of heavy metals including lead) was added to rain garden column reactors. Lead removal rates from 62% to more than 99% (Hsieh and Davis, 2005); greater than 98% (Davis et al., 2001), and 95% to 97% (Sun and Davis, 2007) were found. It was noted that 56% of the lead in the synthetic stormwater was sorbed to the suspended solids added and that lead retention directly correlated to the suspended solids retention rate. Davis et al. (2003) observed a 94-99% lead removal rate. With respect to lead, the study found sand to be a more effective media than mulch. As a field component of the study, four test sites were dosed with synthetic stormwater

and removal rates of 80% to 98% were observed. Event mean pollutant concentrations of actual rainstorms were monitored, and similar removal rates were found. Dietz and Clausen (2006) found that that mulch was less effective at retaining lead as it retained only 36% in their study. Plants in the rain garden (chokeberry [*Aronia prunifolia*], winterberry [*Ilex verticillata*], and compact inkberry [*Ilex glabra compacta*]) were found to accumulate none of the lead.

Most field studies have yielded lower lead removal efficiencies than laboratory studies; however, in some cases, higher removal rates have been observed. For example, Hunt et al. (2006) observed an 81 percent reduction in mass loading for lead in a rain garden field study and Davis (2007) observed a lead removal rate of 83 percent in a field study.

2.B.2 Zinc

Zinc is a common stormwater heavy metal pollutant (Davis et al., 2001; Dietz and Clausen, 2006). Concentrations of zinc are typically greatest in parking lot and street runoff (Pitt et al., 1996). In a groundwater contamination study in France, stormwater from a mostly industrial area was found to have zinc concentrations between 126 and 681 $\mu\text{g/l}$ (Bardin et al., 2001). In another French study, zinc levels in stormwater from a heavily used roadway prior to recharge were found to average 802 $\mu\text{g/l}$; zinc also had the highest variability among all pollutants studied (Barraud et al., 1999). The principle removal mechanisms of zinc from stormwater are precipitation, sorption, and ion exchange (Pitt et al., 1996).

In both laboratory and field studies, zinc has been found to be removed from stormwater via infiltration devices. Zinc retention was found to be similar to lead in a laboratory column study (>98%), and was removed more effectively than copper (Davis et al., 2001). Another column laboratory study found zinc removal efficiencies to be 94 to 97% (Sun and Davis, 2007).

Field studies, in general, have had lower zinc removal efficiencies than laboratory studies. For a field site in Maryland dosed with synthetic stormwater, 64 percent of zinc was removed (with a standard deviation of 42%) (Davis et al., 2003). A subsequent performance study revealed 50 to 70 percent removal of zinc (Davis, 2007). Another field study examined the fate of zinc loading in rain gardens, and found only 16% to be retained in a mulch layer with another 0.2% being assimilated into plant material. Most of the remaining zinc was assumed to be sorbed to sediment because zinc was not detected in the effluent (Dietz and Clausen, 2006). Backstrom (2003) studied grassed swale field sites and found a total reduction of 66 percent for zinc. However, higher reductions have been also been observed; Hunt et al. (2006) observed a 98 percent reduction in mass loading for zinc in a rain garden field study.

2.B.3 Copper

Copper was found in virtually all stormwater samples analyzed by Pitt et al. (1995); the highest median copper concentrations were from urban stream samples (160 $\mu\text{g/l}$) and street runoff had the highest single concentration (1250 $\mu\text{g/l}$) (Pitt et al., 1995). Wu et al. (1998) found copper concentrations in highway runoff to be between 2.5 and 15 $\mu\text{g/l}$ for ten storms in North Carolina.

Copper loading correlated well to increased impervious cover and vehicle usage intensity. In a groundwater contamination study in France, stormwater from a mostly industrial area was found to have copper concentrations between 7 and 49 $\mu\text{g/l}$ (Bardin et al., 2001). The chief copper removal mechanisms of from stormwater in soil are sorption, complex ion formation, and ion exchange (Pitt et al., 1996). The EPA drinking water action level is 1.5 mg/L (U.S. EPA, 2008)

In one laboratory column study, 93% of copper in a synthetic stormwater influent was removed (Davis et al., 2001). A subsequent laboratory study found copper removal efficiencies of between 88 and 93% (Sun and Davis, 2007).

Field studies indicated lower removal efficiencies. 43% (standard deviation of 11%) of copper was removed in one field study (Davis et al., 2003); another subsequent study indicated between 50 and 60 percent removal (Davis, 2007). The results of a study that examined the fate of copper within bioretention cells, found 98% of the influent copper was retained in a mulch layer, with 0.1% uptake by plants (Dietz and Clausen, 2006). Backstrom (2003) studied field sites with grassed swales and found the swale provided a total reduction of 34 percent for copper. Higher reductions have been observed; Hunt et al. (2006) observed a 99 percent reduction in mass loading for copper in a rain garden field study.

2.B.4 Cadmium

Cadmium is of concern due to human health implications; although it is commonly detected, concentrations in stormwater are typically very low (Pitt et al., 1995). In a French study on an existing infiltration practice, cadmium levels in stormwater from a heavily used roadway were found to average less than 5 $\mu\text{g/l}$ (Barraud et al., 1999). Concentrations detected from highway runoff at three sites near Charlotte, North Carolina averaged 2.5 $\mu\text{g/l}$ average for ten storms (Wu et al., 1998); for reference, the EPA drinking water standard for cadmium is 5 $\mu\text{g/l}$ (EPA website). In contrast to the other studies, a correlation between impervious surface cover or vehicle use intensity and cadmium concentration in runoff was not found. This finding supports the suggestion that the main source of cadmium is wet deposition (Davis et al., 2001). Pitt et al. (1995), however, found the highest median cadmium concentrations (8 $\mu\text{g/l}$) in vehicle service runoff, and samples from one street that was examined contained 220 $\mu\text{g/l}$ which was the highest of all samples collected (Pitt et al., 1995).

Because the main removal mechanisms for cadmium are ion exchange, sorption, and precipitation (Pitt et al., 1996), soil media may retain cadmium. Laboratory column studies conducted thus far have shown removal rates of up to 95% or higher (Sun and Davis, 2007). Due to the typically low concentrations of cadmium found in stormwater, most studies to date have neglected cadmium.

2.C Suspended Solids

Suspended solids, which are almost always found in stormwater runoff samples, can degrade water quality (U.S. EPA, 1999). Construction and land-disturbing activities are the leading source of suspended solids in stormwater (U.S. EPA, 1999). Because metals, pesticides, and petroleum hydrocarbons often are sorbed to solid surfaces, solids provide a means of transport

and accumulation of pollutants (U.S. EPA, 1999). Additionally, suspended solids can degrade aquatic ecosystems by covering fish spawning areas (U.S. EPA, 1999) and blocking sunlight.

In a study on the cost and effectiveness of stormwater BMPs, Weiss et al. (2007) collected stormwater suspended solids data from existing literature. Data was gathered from study sites located across the United States and the average suspended solids concentration was found to be 143 mg/L (+- 77 mg/L 67% confidence interval).

The primary removal mechanisms of suspended solids are physical filtration and sedimentation. Infiltration systems provide filtration of runoff but the percent removal of solids depends on, among other variables, particle size and the size of the pore opening between soil particles. Hsieh and Davis (2005) conducted both laboratory column tests and field studies. Column percent removal of suspended solids ranged from 29% to greater than 96% and removal at six field sites ranged from 77% to 99%. Only 43% and 47 % of the suspended solids were removed (on average) from two bioretention cells monitored for twelve storm events (Davis, 2007). Hunt et al. (2006) observed an increase in suspended solids concentrations in a rain garden field study (Hunt et al., 2006). Backstrom (2003) observed a 70% TSS removal efficiency at a roadside grassed swale field site, and the International Stormwater Best Management Practices Database (1999-2007) reports a median removal value of 66% for biofilters (ASCE, 2007).

2.D Organic Compounds

Organic compounds can be naturally occurring (e.g. animal waste, vegetation, soil organisms) or anthropogenic in origin (e.g. petroleum hydrocarbons, automobile tire particles). As applied to groundwater contamination, the focus has been mainly on man-made organic compounds including petroleum hydrocarbons. Organic compound removal may occur through volatilization, sorption, and degradation (Pitt et al., 1999).

Pitt et al. (1996) report that 1,3-dicholobenzene, pyrene, and flourene may have high groundwater contamination potential in subsurface infiltration/injection systems without pretreatment, but would probably have lower contamination potential in surface infiltration systems. A series of other organics may have moderate groundwater contamination potential when no pretreatment system is used.

In a Florida study that investigated the fate of organic compounds that were infiltrated with stormwater, most organic hydrocarbons were found to be attenuated in the soil but one compound was detected in the groundwater. In an Arizona study, base and neutral compounds from residential areas were found in the groundwater and phenol contaminated groundwater was found near a commercial site. Organic compounds were also found to have contaminated groundwater in studies in Long Island, Maryland, and the United Kingdom (Pitt et al., 1999).

Sources of petroleum hydrocarbons include leaky storage tanks, parking lot and roadway runoff, automotive emissions, illicit dumping, and spills (U.S. EPA, 1999). Petroleum hydrocarbons are classified as priority pollutants; some are at least somewhat water soluble, making them a common groundwater contaminant (Płaza et al., 2007). Petroleum hydrocarbons are known for their acute toxicity (U.S. EPA, 1999); thus small concentrations can be of concern. Despite their

toxicity and presence in stormwater runoff, existing literature on the fate of hydrocarbons in infiltration systems is much less than for heavy metals and nutrients.

A study from the Washington, DC metropolitan area found petroleum hydrocarbon median levels of 0.7 to 6.6 mg/L; the level used for protection of fisheries is 0.01 to 0.1 mg/L (Shepp, 1996). Oil and grease levels found in highway runoff ranged between 1.3 and 3.3 µg/L (Kim et al., 2007; Wu et al., 1998). Other studies, however, have found higher concentrations of oils/grease. Kim et al. (2005) observed 5.23 µg/L, Zanoni et al. (1986) observed 2-79 µg/L, and Barraud et al (1999) observed 110 µg/L in stormwater.

Hsieh and Davis (2005) tested laboratory column rain gardens with sandy and sandy loam media and found greater than 96% removal of oils and grease (introduced as used motor oil through synthetic stormwater). The results of their field study found that removal efficiency of oils and greases from a synthetic stormwater was nearly 100 percent and 99% removal was observed during a natural rain event.

2.E Pathogens

For residential and light commercial developments, pathogens (i.e. bacteria and virus) in stormwater are a primary pollutant of concern and they may be present in high concentrations and not retained well in the soil (Pitt, 1999). The highest bacteria and virus concentrations in groundwater were found to occur when the water table was near the land surface (Pitt et al., 1999). *Fecal streptococci* and *E. coli* were found in 94% and 95.5%, respectively, of municipal separate storm sewer systems (MS4) outfalls monitored (Clark and Pitt, 2007).

Pitt et al. (1996) rate enteroviruses as having high groundwater contamination potential for all surface and subsurface infiltration/injection systems and a variety of other pathogens as having high groundwater contamination potential for subsurface infiltration/injection systems. Bacteria may be removed by straining at the soil surface and sorption to solid particles. Once removed from the water, the ability of bacteria to survive is a function on factors such as temperature, pH, presence of metals, etc. Bacteria survive longer in acidic soils and in soils with large amounts of organic matter. Bacteria survival for up to five years has been documented (Pitt et al., 1999).

As part of the National Urban Runoff Program, fecal coliform was evaluated at 17 sites for 156 storm events and, based on the results, concluded that coliform bacteria are present at high levels in urban runoff and may exceed EPA water quality criteria during and after storm events, (U.S. EPA, 1999). There existed a high degree of variability within the data but land use did not appear to correlate with coliform concentration. During warmer months, concentrations were approximately 20 times higher than cold months.

Groundwater contamination potential depends on the soil chemical properties, adsorption capability, the ability of the soil to physically strain the pathogens, and pathogen survival. Bacteria survive longer in low pH soils and in soils with high organic content. Bacteria and viruses can move through soil media and may be transported to aquifers by infiltrating

stormwater. The transport distance of bacteria seems to be a function of bacteria density and water velocity through the soil (Camesano and Logan, 1998; Unice and Logan 2000).

Documented pathogen contamination of groundwater due to infiltration practices has occurred (Clark et al. 2006) and *E. coli* have been shown to pass through stormwater sand filters (Clark and Pitt 2007). Dietz and Clausen (2005) found fecal coliforms concentrations to be less than 10 colony forming units (CFU) per 100 mL in both the influent (from roof runoff) and the effluent from a rain garden. Rusciano and Obropta (2007) created a bioretention column in which horse manure was fed to the column to simulate a bacterial pollutant source. Results indicated the median reduction in fecal coliform was 98.6%.

2.F Salts

Because salts are soluble, easily transported in surface and sub-surface flow, non-filterable, and do not readily sorb to solids, they have high potential for groundwater contamination (Pitt et al. 2002). Rather than being reduced, salt concentrations typically increase as water moves through soil due to leaching of salts into the water (Pitt et al., 1999).

Research has shown that concentrations of chloride have been increasing in local waterways in New England (Kaushal et al. 2005), and if current trends continue, chloride levels in streams will reach levels that will threaten aquatic life.

Although it is not a salt, anti-caking agents are usually used to prevent road salts from forming clumps (i.e. caking). Paschka et al. (1999) investigated the potential affects of anticaking agents used in road salt on water quality. The major pollutant of concern in anticaking agents is cyanide which may occur as HCN. Although HCN is usually assumed to leave the water surface quickly due to its volatility, there is insufficient data in the literature to confirm this assumption. The extensive literature reviewed contained studies on the influence of anticaking agents on surface water quality but none, however, addressed the issue of groundwater contamination.

3. Groundwater and Soil Contamination

The previous section focused on some of the most common contaminants found in urban stormwater runoff and the capability of infiltration systems to retain them. Most of those studies monitored influent and effluent contaminant concentrations or mass loads and, by comparing the difference between them, calculated a percent removal. Other studies have investigated whether or not stormwater runoff contaminants have the ability and/or potential to travel to aquifers and pollute groundwater. Other investigations have focused on the soil (or other media) contaminant concentration that result from infiltration of polluted stormwater. Groundwater and soil protection from stormwater contamination are legitimate concerns for protecting human and environmental health and merit further investigation (Lind and Karro, 1995).

This section provides a literature review of the latter two types of studies: groundwater contamination and soil contamination that results from infiltrating stormwater. The section ends

with a review of published literature that compiled results and presented summaries of these two topics or that involved model simulations of pollutant transport related to infiltration systems.

3.A. Groundwater Contamination

Stephenson et al. (1999) performed a monitoring and die tracer study on highway runoff entering a sinkhole in Knoxville, Tennessee and a spring located 420 feet away from the sinkhole. Total zinc, dissolved lead, total lead, total petroleum hydrocarbons, polycyclic aromatic hydrocarbons, total dissolved solids, total suspended solids, and total volatile solids concentrations at the sinkhole and spring were monitored for only one runoff event. For most contaminants, the peak contaminant loading at the spring lagged behind that at the sinkhole by about one hour. The peak loading of TDS, however, lagged by only 20 minutes. The peak loading rates were higher at the spring than at the sinkhole for TDS and dissolved zinc, but lower for all other contaminants. The authors concluded that groundwater is more susceptible to contamination by highway runoff in karst aquifers because these aquifers can transport runoff and pollutants with little or no attenuation or filtration.

Fisher et al. (2003) compared groundwater beneath 16 detention basins in developed areas with groundwater from monitoring wells in undeveloped areas. Results showed that some pesticide concentrations were higher in the undeveloped areas but in some cases the opposite was true. The authors suggested that with some pesticides the water from the monitoring well was diluted by the infiltration of detention basin water. The most noticeable difference was that the water beneath the detention basin had much lower dissolved oxygen (DO) levels than the monitoring wells in undeveloped areas. Also, concentrations of nitrite and nitrate were typically higher in the high DO monitoring well water. This was probably due to the fact that nitrification occurs more readily under aerobic conditions. Concentrations of ammonia and organic nitrogen, however, were found to be higher in the water beneath the detention basins (i.e. low DO water). Chloride concentrations were found to be roughly the same; however, the study was conducted in a year with low snowfall and consequently lesser quantities of salts were applied to the roadways during the course of the study.

Datry et al. (2004) investigated the effects of infiltration on water quality beneath an infiltration basin. The authors found that groundwater at a depth of 1 meter below the water table consisted almost entirely of stormwater and that stormwater did not penetrate to depths greater than 3 meters below the water table. Dissolved phosphate in the groundwater was higher in concentration than the stormwater influent indicating that it was produced or added in the infiltration bed; the authors concluded that mineralization of organic sediments was the most likely cause. There was also evidence that organic sediments increased the dissolved organic content of the water.

Within the water samples collected from the infiltration bed, ammonium was the dominant nitrogen species during dry weather but it was not found in the groundwater. This indicates that ammonium was oxidized to nitrate within the infiltration bed. Also heavy metals and hydrocarbons were not detected in the groundwater. The authors found that stormwater

infiltration during cold rains raised the groundwater DO slightly and decreased the DO during warm rains.

3.B Soil/Media Contamination

Studies have found that most hydrocarbons are trapped in the first few centimeters of soil in infiltration basins (Barraud et al., 1999; Dierkes and Geiger, 1999). The type of hydrocarbon appears to affect the fate; Dierkes and Geiger (1999) found that 'mineral oil' type hydrocarbons (MOTHs) were more likely to be captured in soil and degraded than polycyclic aromatic hydrocarbons (PAHs). Mikkelsen et al. (1997) researched soil and groundwater contamination of infiltration sites due to stormwater contaminated with PAHs and found that they readily sorb to soil particles. The authors concluded that these contaminants posed little risk of groundwater contamination. Bardin et al. (2001) studied the performance of various pretreatment facilities just upstream of infiltration basins and found simple sand filters to be ineffective at capturing hydrocarbons.

The only study found that explicitly examined the fate and biodegradation potential within infiltration systems was Hong et al (2006). Based upon the work of Hsieh and Davis (2005) which found that the majority of oils and greases were trapped (but not degraded) in a top mulch layer in a column study, Hong et al. (2006) created a reactor in which a 3 cm deep mulch layer was fed synthetic stormwater containing, among other contaminants, naphthalene, toluene, motor oil, and particulate associated naphthalene. Mulch, having a high lignin content, has a strong affinity for nonionic organic compounds and will retain hydrocarbons. Filtration may also be a significant removal process because 74% to 90% of hydrocarbons in urban runoff are associated with particulate matter. For biodegradation to occur, the mulch layer must support a large enough population of hydrocarbon degrading microbes and allow for a long enough contact time.

Naphthalene was removed from the stormwater at approximately 90%, toluene and motor oil by 80%, and particulate associated naphthalene by 97%. By observing the decrease in concentrations of contaminants in the mulch layer and the rise of the microbe population after the contaminant was captured, it was determined that biodegradation was occurring in the mulch.

Mikkelsen et al. (1997) investigated the pollutant level in soil media of several contaminants in the soil of an infiltration system in Switzerland. Soil characteristics and pollutant concentrations were determined at two sites and soil conditions at both sites were dominated by calcerous gravel deposits. One site (Site A) received runoff from a watershed that included agricultural, residential, and light industrial land uses. The road near the infiltrates test section had an average traffic density of 37,000 vehicles per day. The second site (Site B) was located near a city with heavy traffic and a waste incineration plant. Results of soil properties and contaminant concentrations at each site are shown in Figure 1.

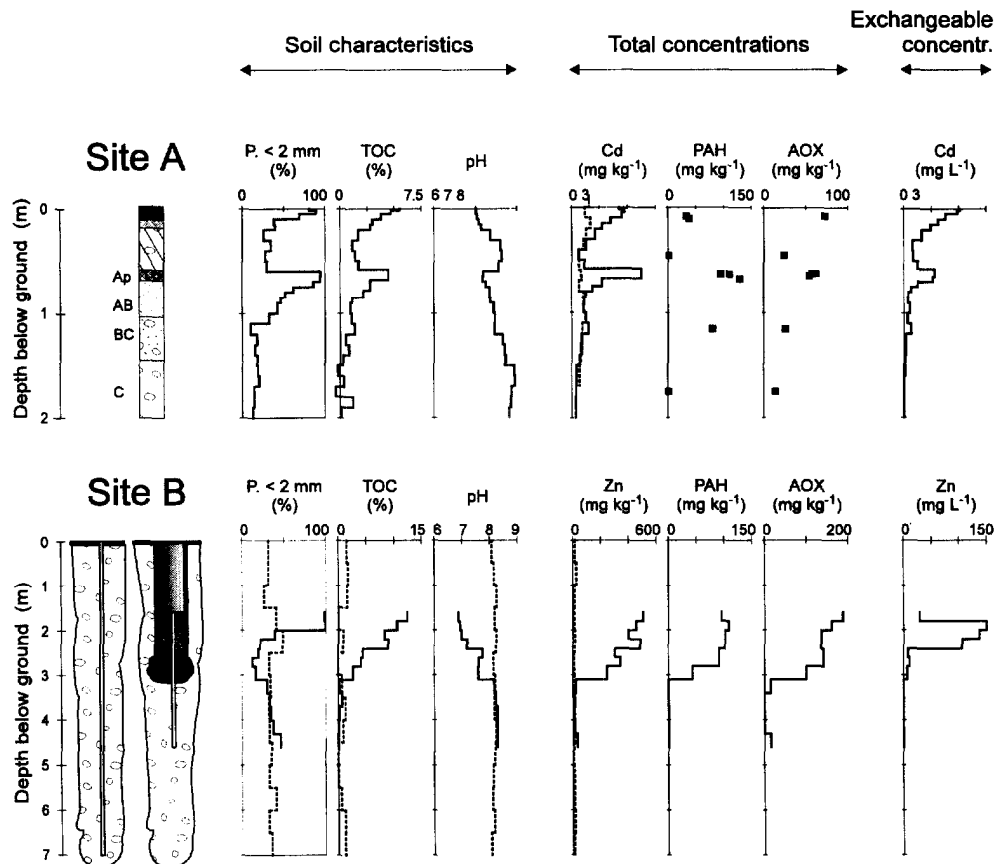


Figure 1 Vertical profiles of soil properties and pollutant concentrations (Mikkelsen et al. 1997).

It was concluded that the solids deposited by the runoff were a source of contaminants, but also were capable of some contaminant sorption. A top soil layer was found to contain high levels of pollutants, but concentrations decreased rapidly with increasing depth. Thus the authors suggest groundwater contamination is not a threat for the foreseeable future. Also, contaminants that do not sorb to soil solids (e.g. salts) may have passed through the media to the groundwater. The results are dependent on the areas geology and should not be assumed to hold true in other infiltration systems.

Bucheli et al. (1998) investigated pesticides in rainwater, roof runoff, and artificially infiltrated runoff. Pesticide concentrations in percolating groundwater at different depths and at different infiltration rates were found to be very similar to those in the runoff. Figure 2 shows this trend and the data suggest very quick infiltration of runoff and that pesticides were not retained by the soil media. As defined in the legend, the black dots indicate roof runoff samples and the lines are data from three different lysimeters. If not retained by the soil, these contaminants could potentially contaminate groundwater supplies.

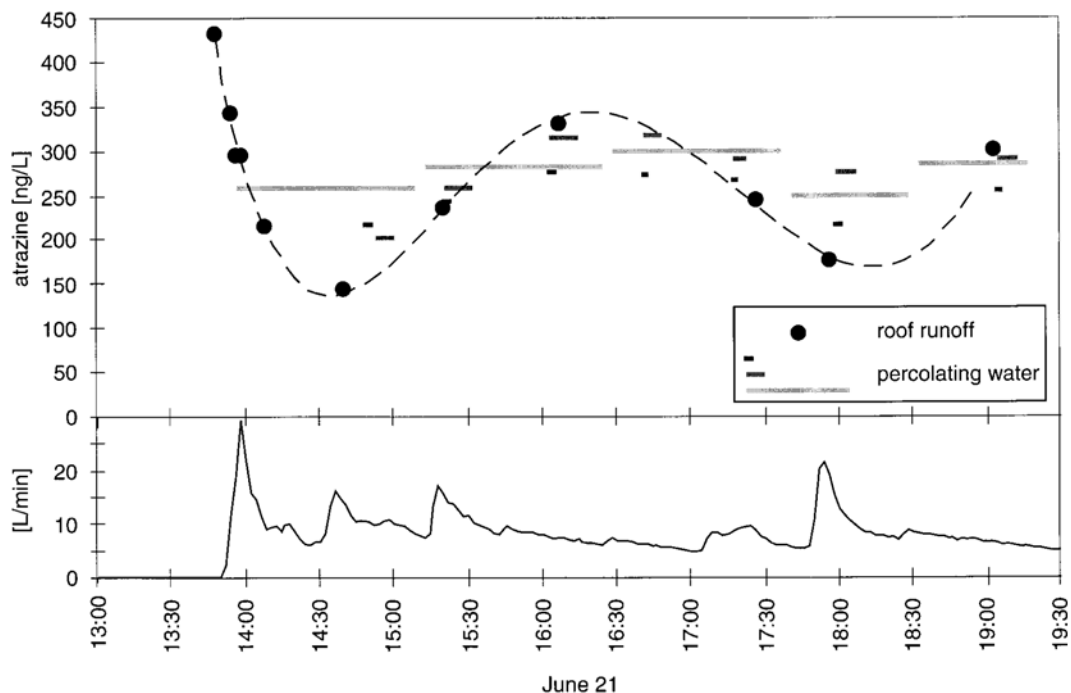


Figure 2 Atrazine concentrations in roof runoff and infiltrated water (upper graph) and runoff flow rate (lower graph) as a function of time (Bucheli et al. 1998).

Legret et al. (1999) investigated the infiltration of stormwater containing copper, cadmium, lead, and zinc into a porous pavement and underlying soil. Laboratory studies and mathematical modeling were conducted with concurrent field verification of a site in France. The concentrations in rainwater for lead, copper, cadmium, and zinc were 28-50, 17-24, 0.4-0.8, and 250-370 $\mu\text{g/l}$, respectively. In general, results from the different studies were in agreement and it was concluded that the increase of lead, copper, and zinc in soil that acts as filter media for stormwater appears slight after 50 years as soil concentrations were well below regulation threshold values. The metals were retained in the upper 20 cm of soil except for cadmium which showed migration beyond 30 cm.

Dierkes and Geiger (1999) also investigated the concentrations of zinc, copper, cadmium, polycyclic aromatic hydrocarbons, and mineral-oil-type hydrocarbons along the side of five major highways near Essen, Germany. Soil cores were taken and the soil was analyzed at depths of 0 cm to 5 cm, 5 cm to 10 cm, and 10 cm to 30 cm. They found that pollutant concentrations in the soil were a function of traffic density and the length of time the soil had been infiltrating runoff. Highest concentrations were found in the upper 5 cm of soil within a distance of 2 meters from the road. Dierkes and Geiger (1999) also found that concentrations of pollutants decreased rapidly with depth. Between 10 and 30 cm, in most cases, only 7% to 25% of the metal concentrations found in the upper 5 cm was detected. The decrease in metal concentration with depth was most pronounced for lead and copper and was weakest for cadmium. At a distance of 10 m from the road, the concentrations of copper decreased to 7% of that found close to the road. For lead, zinc, and cadmium the decrease was 30%, 30%, and 45%, respectively. The results of this portion of the study are summarized in Table 1. The study further found little impact to groundwater; nevertheless, an increase in heavy metals concentration was detected. It

was suggested that a minimum of 40 cm of unsaturated soil be in place to protect groundwater. The mobility of these metals once deposited, however, is mostly unknown (Perez et al., 2008).

Table 1 Concentrations of contaminants at five sites as a function of depth and distance from road (Dierkes and Gieger 1999).

Highway	Depth [cm]	Dis- tance [m]	Pb	Zn	Cu	Cd	PAH	MOTH
			[mg/kg]	[mg/kg]	[mg/kg]	[mg/kg]	[mg/kg]	[mg/kg]
A 2	0-5	0.5	239	527	413	3.9	6.7	150
	5-10	0.5	202	361	78	3.5	11.3	110
	10-30	0.5	34	99	31	2.7	5.3	57
	0-5	0.3	213	398	121	3.4	16.6	190
	0-5	2	220	336	95	3.0	9.4	74
	0.5	5	141	231	42	2.0	9.4	62
	0-5	10	65	155	27	1.8	2.1	36
A 3	0-5	2	81	174	25	2.0	5.3	200
	5-10	2	69	141	20	1.9	7.0	73
	10-30	2	67	114	11	1.1	5.0	23
A 31	0-5	0.75	276	759	268	4.3	< 2.1	28
	5-10	0.75	130	303	69	2.6	< 2.1	23
	10-30	0.75	54	112	24	2.5	< 1.6	12
A 42	0-5	2	290	1580	167	5.6	23.0	510
	5-10	2	348	1630	155	8.5	16.9	220
	10-30	2	27	138	23	3.1	< 1.6	60
B 224	0-5	0.75	71	187	40	2.2	2.5	160
	5-10	0.75	53	120	42	2.5	< 1.9	25
	10-30	0.75	18	69	24	-	< 1.7	21

Diekes and Gieger (1999) sampled water quality at three sites and a detention pond (A43). The detention pond was used as a runoff source for the laboratory experiments. Results are given in Table 2.

Table 2 Runoff quality as found by Dierkes and Gieger (1999).

	Total metals				Dissolved metals			
	Zn (mg/l)	Cu (µg/l)	Pb (µg/l)	Cd (µg/l)	Zn (mg/l)	Cu (µg/l)	Pb (µg/l)	Cd (µg/l)
A 43	0.8-3.0	80-130	10-20	0.8-3.6	0.3-1.3	17-56	n.d.-10	n.d.-0.7
A 31	-	40-150	4-60	0.5-1.0	-	20-80	n.d.	n.d.-0.6
A 42	0.7-41.0	60-70	10-40	1.7-3.3	0.2-23.1	30-60	n.d.	n.d.-2.4
B 224	0.2-0.8	60-160	n.d.-4	0.7-7.6	0.1-0.6	20-50	n.d.	n.d.-1.8

n.d. = not detectable

For the laboratory studies, a core of soil from each site was tested. Each core infiltrated water from the detention pond (A43) for six months. The mean concentration of pollutants in the source water and the effluent are shown in Table 3.

Table 3 Mean concentrations of source water (i.e. runoff) and effluent from soil samples (Dierkes and Gieger 1999).

	runoff	B 224	A 2	A 3	A 31	A 42
pH	7.3	7.7	7.4	7.5	7.6	7.6
Pb	[$\mu\text{g/l}$] 17	< 5	< 5	< 5	11	8
Zn	[$\mu\text{g/l}$] 1250	44	101	69	274	107
Cu	[$\mu\text{g/l}$] 140	31	50	38	40	50
Cd	[$\mu\text{g/l}$] 1.4	< 0.4	< 0.4	< 0.4	< 0.4	< 0.4

The authors concluded that cadmium exhibited the most significant downward mobility but that breakthrough of all metals (i.e. contamination of groundwater) is not expected in the near future. There is a limitation on the the amount of contaminants the soil can retain, however, and at some point in the future soil capacity will be exhausted and the soil will have to be removed. In the long run, however, the authors suggested that the soils may have to be removed. PAH, which biodegrades slowly, accumulated in the upper 10 cm of the soil whereas MOTHs decreased more rapidly with depth due to faster degradation.

Barraud et al. (1999) investigated the effectiveness of two different infiltration chambers and the potential impact on the soil and groundwater in France. One infiltration chamber was two years old and the other was 30 years old. The chambers received runoff of similar quality and the bottoms of both structures were 40 cm above the water table. Both soil solids and groundwater samples were analyzed for various pollutants.

Table 4 shows the quality of runoff water that entered the infiltration chamber (i.e. soakway) and the groundwater quality below the chamber during dry and rainfall conditions.

Table 4. Runoff and groundwater quality at 2 year old infiltration chamber (Barraud et al. 1999).

	Runoff-water at soakway entry (01/10/96)	Groundwater under dry weather conditions (mean value)	Groundwater during the rainfall downstream from newer soakway	
			1.15 pm	3.20 pm
Conductivity ($\mu\text{S}/\text{cm}$)	54	675	179	123
pH	7	7.4	7.4	8.2
Nitrates NO ₃ (mg/l)	3	59	13	4
TKN (mg/l)	1.2	<1	1.2	1.4
Zn ($\mu\text{g}/\text{l}$)	5700	0 to 8	90	50
Pb ($\mu\text{g}/\text{l}$)	45	0 to 6	8	13
Cd ($\mu\text{g}/\text{l}$)	<5	<5	<5	<5
COD (mg/l)	63	0	42	27
TOC (mg/l)	8.7	0.9	21.2	3.5

Perforated PVC pipe 50 cm high and 100 mm in diameter were filled with fine sand and planted into the bottom of each chamber. The mass of pollutants that entered each chamber was estimated by multiplying pollutant concentrations by volumes of runoff. At the terminus of the experiment (after 10 months), the mass of pollutants retained by the sand cores were analyzed. Results for the new and old chamber are provided in Table 5 and Table 6, respectively.

Table 5 Pollutant loads and retention in 2 year old infiltration chamber (Barraud et al. 1999).

element	concentration in sand-cores (mg/kg)			trapped mass (g)	input mass (g)	% retention
	0/5 cm	5/10 cm	mean			
Zn	857	917	887	70	80-130	54-88
Pb	197	212	204.5	16	10-16.3	98- ?
Cd	2.2	2.5	2.35	0.18	<	?
Aromatic hydrocarbons	0.29	0.1	0.19	0.02	<	?
Mineral oils	1290	1600	1445	113	10	?

< : not calculable (concentration below the threshold level of detection) ? : not calculable

Table 6 Pollutant loads and retention in 30 year old infiltration chamber (Barraud et al. 1999).

element	concentration in sand-cores (mg/kg)	trapped mass (g)	input mass (g)	% retention
	0/10 cm			
Zn	143	118	380	31
Pb	82	67	46	?
Cd	0.9	0.7	2.37	29.5
Aromatic hydrocarbons	0	0	<	?
Mineral oils	866	710	55	?

< : not calculable (concentration below the threshold level of detection) ? : not calculable

For some pollutants, the mass of pollutant input into the chamber was less than that retained. Possible explanations provided by the authors include variable concentrations in runoff that were not fully captured by the sampling method, samples taken were not representative of the runoff (e.g. floating mineral oils not present at sampler intake), or uncontrolled discharges into the chambers.

The authors concluded that heavy metal and hydrocarbon concentrations were very high in the first few centimeters of soil but dropped rapidly with depth. Even so, long term pollution is possible as the contaminants spread but the concentrations of pollutants could remain low enough as to meet the Dutch standards. The impact on groundwater was classified as “low” although heavy metal contamination in the groundwater was detected. The impact could be reduced by providing more than 40 cm of separation between the bottom of the infiltration chambers and the water table.

The performance of a sand trap, presettling basin, infiltration basins, and oil separator system with regards to pollutant retention was investigated by Bardin et al. (2001) in a heavy industrial area in France. The site has been in operation as a stormwater infiltration device since 1975 and has undergone some modifications since it was constructed. Since 1988 however, it has consisted of two sand traps, a settling basin, an infiltration basin, and an oil separator. The land

use in the 380 ha watershed was 77% industrial, 20% agricultural, and 3% residential. The basin had a volume of 80,000 m³ and an area of over 2 hectares. Total event mean influent concentrations for the pre-settling basin influent and effluent along with the infiltration basin influent are given in Table 7 and results from the analysis of solids in the sand trap, settling basin, and oil separator are given in Table 8.

Table 7 Event mean concentrations for the settling basin inlet (I), settling basin outlet (SB), and infiltration inlet (IB) (Bardin et al. 2001).

	Event no. 1			Event no. 2			Event no. 3			Event no. 4			Event no. 5			Event no. 6			Event no. 7		
	I	SB	IB	I	SB	IB	I	SB	IB	I	SB	IB	I	SB	IB	I	SB	IB	I	SB	IB
SS mg/l	39	21	21	64	57	71	64	47	38	51	23	21	128	35	50	37	15	13	135	43	/
Zn µg/l	177	<5	162	327	329	294	282	282	258	267	266	209	681	433	451	303	268	265	126	180	/
Cu µg/l	16	11	13	36	27	29	21	15	14	15	13	12	49	23	23	9	9	10	7	8	/
Pb µg/l	17	13	11	34	17	0.0	24	14	12	<5	<5	65	90	38	42	9	<5	<5	<5	5	/
COD mg/l	41	22	24	38	43	13	101	52	36	43	40	42	126	63	63	47	45	41	/	28	/
oil µg/l	294	684	602	53	262	276	<50	63	<50	<50	<50	<50	1407	1080	858	597	1413	1257	92	92	/

(I: settling basin inlet; /: not analyzed; SB: settling basin outlet (separator inlet); IB: infiltration basin inlet (separator outlet))

Table 8 Concentrations of sorbed pollutants on solids in sand traps, settling basin, and oil separator (Bardin et al. 2001).

Cleaning date	Operation period of the system	T210 sand-trap		C1200 sand-trap		Settling basin		Oil separator
		13/10/95	13/10/95	23/11/94	12/18/95	23/02/95		
	6 months	6 months	6 months	12 months	11 months	4 months		
Zn	(g/t dry matter)	288	224	1148	1836	168		
Cu	(g/t dry matter)	39	219.5	188	230	237		
Ni	(g/t dry matter)	111.6	14.2	25	42.8	27.9		
Cr	(g/t dry matter)	96	41.6	57	88	61.9		
Cd	(g/t dry matter)	1.9	1.95	5	5.6	5.83		
Pb	(g/t dry matter)	38.5	120.8	296	338.5	342		
Hg	(g/t dry matter)	0.083	0.1	/	0.45	/		
Mineral oils	(g/t dry matter)	624	768	10000	6583	37660		

Based upon full results (see article for details), the authors concluded that the sediment in the infiltration basin tends to sorb pollutants and increase the removal efficiency of pollutants from the runoff and that removal is more of a biological and/or chemical process than it is physical filtration. The authors also concluded that the gravel and geotextile had a significant filtration effect but they did not make any conclusions with respect to the quantity of pollutants that pass through to the underlying groundwater. It was also concluded that heavy metal retention was not very high (25% to 60%, depending on the metal).

Indiati and Diana (2004) investigated the capacity of acidic soils to sorb phosphorus. Sorption capacities were found to range from 19.1 mmol P/kg soil to 284.3 mmol P/kg soil, with the degree of phosphorus saturation being time dependant. Time to full phosphorus saturation ranged from 10 to 100 days. The phosphorus sorption capacity was determined to have a strong correlation with the sum of aluminum and iron oxides/hydroxides extracted with an acid oxalate solution (Figure 3).

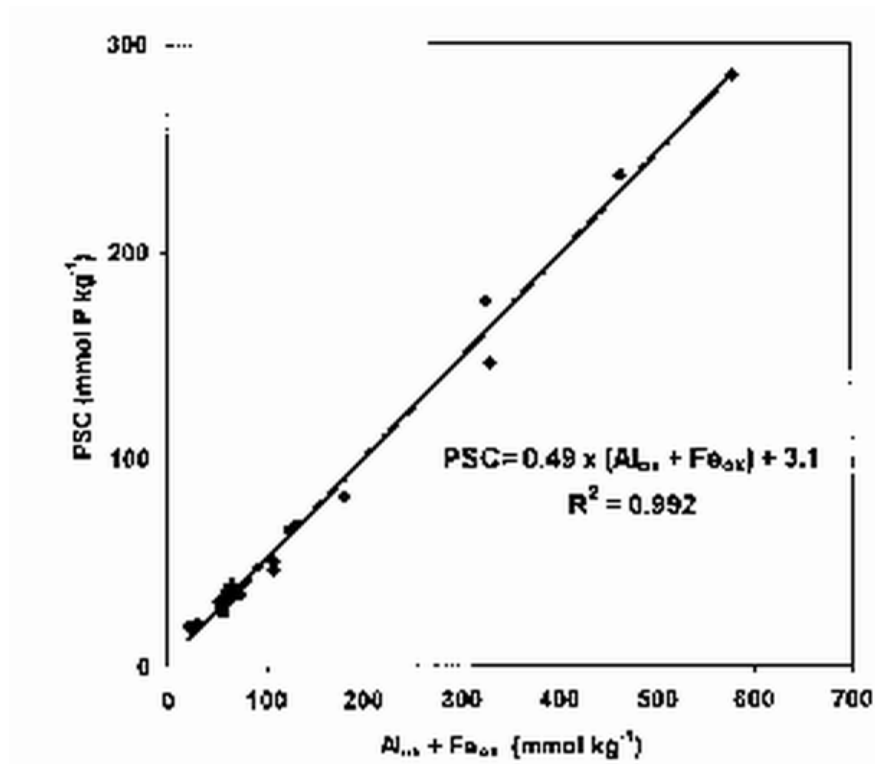


Figure 3 Phosphorus sorption capacity as a function of the sum of extracted aluminum and iron oxides/hydroxides (Indiati and Diana 2004).

An infiltration basin in France that had been operating for 14 years was investigated by Dechesne et al. (2004). The 2616 m³ basin infiltrated water from a 7 hectare truck parking lot and existed above highly permeable soils of calcareous sand, pebbles, and rocks. The water table was 4 meters below the basin and hydraulic conductivity values in the area ranged from 10⁻⁴ to 10⁻² m/s. At 10 locations soil samples were taken at the surface and at 30-40 cm, 60-70 cm, and 100-110 cm below the surface and were analyzed for a host of contaminants. Data for all ten sampling locations and all four depths are shown in Table 9

Table 9 Soil contaminant concentrations under an infiltration basin in France. 10 points, 4 depths. A (surface), B (30-40 cm), C (60-70 cm), D (100-110 cm) (Dechesne et al., 2004).

	pH	CEC mg/kg	Mineral matter %	TOC %	TN %	TP %	Cd mg/kg	Cr mg/kg	Cu mg/kg	Ni mg/kg	Pb mg/kg	Zn mg/kg	Hg mg/kg	As mg/kg	THC mg/kg
1A	7.9	98.8	86	10.78	0.33	0.27	4.00	60.94	142.4	29.97	222.3	1414	0.11	7.99	1565
1B	9.3	11.0	99.1	1.07	0.02	0.07	0.51	13.69	7.61	7.61	8.62	65.9	<0.1	3.04	29.75
1C	9.3	9.0	99.2	0.73	0.02	0.07	0.51	14.68	8.10	8.10	7.08	56.2	<0.1	3.04	22.48
1D	9.4	12.8	99.2	1.08	0.01	0.07	0.49	11.37	6.43	7.42	4.94	38.1	<0.1	2.47	13.29
2A	8.1	76.5	87.6	10.67	0.24	0.32	3.53	49.42	110.4	23.20	147.8	1145	<0.1	6.56	1745
2B	8.9	16.0	98.7	1.3	0.03	0.09	0.94	18.24	14.03	10.29	28.07	182.0	<0.1	4.21	138.1
2C	9.3	9.0	99.3	1.02	0.02	0.09	0.47	14.46	7.47	7.47	6.53	71.9	<0.1	3.27	18.87
2D	9.3	9.6	99.3	0.89	0.02	0.07	0.45	13.59	7.25	9.06	4.98	66.1	<0.1	3.62	9.97
3A	9.0	25.8	97.8	2.45	0.04	0.09	0.97	22.74	24.68	11.61	67.73	255.5	<0.1	4.84	151.1
3B	9.4	6.8	99.4	1.05	0.03	0.07	0.50	10.53	2.51	6.52	5.01	19.0	<0.1	3.01	<10
3C	9.4	6.6	99	1.05	0.01	0.05	0.49	12.76	3.93	7.36	4.91	23.6	<0.1	3.44	<10
3D	9.5	7.9	99.2	1.11	0.01	0.05	0.53	20.82	3.20	9.07	5.34	21.9	<0.1	4.27	<10
4A	8.2	83.9	86.7	11.13	0.25	0.25	3.55	56.33	121.8	27.40	176.6	1269	0.107	8.12	538
4B	9.0	18.6	98.2	1.8	0.03	0.07	1.04	23.85	20.74	11.41	52.89	247.8	<0.1	4.15	72.82
4C	9.3	13.5	98.9	1.24	0.02	0.07	0.51	19.54	13.37	11.83	10.29	85.9	<0.1	4.11	<10
4D	9.4	13.2	98.9	1.11	0.02	0.07	0.52	19.22	10.91	12.99	6.75	52.0	<0.1	3.64	<10
5A	8.7	37.6	95.7	2.78	0.09	0.11	1.54	29.32	38.06	14.40	84.87	458	0.1	4.63	95.17
5B	8.9	28.0	97.3	1.72	0.04	0.09	1.01	29.91	29.40	14.19	83.13	371.0	0.1	4.06	151.5
5C	9.3	11.1	98.9	0.74	0.02	0.05	0.51	13.24	7.64	7.64	17.82	73.3	0.1	3.56	63.26
5D	9.4	27.5	98.5	0.74	0.02	0.11	0.51	22.87	14.23	19.31	7.11	54.4	0.1	5.59	<10
6A	7.9	75.8	91.5	6.26	0.19	0.18	2.45	38.17	75.36	21.53	111.1	837	<0.1	8.81	811
6B	9.2	15.9	99.1	0.59	0.02	0.07	0.46	16.59	5.99	10.14	4.61	27.6	<0.1	5.53	<10
6C	9.5	7.4	99.3	0.5	0.01	0.07	0.46	16.03	5.04	8.70	4.58	22.4	<0.1	5.04	<10
6D	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--
7A	7.8	129.3	79	14.8	0.41	0.34	4.75	66.92	175.6	33.22	223.1	1827	0.1	12.34	1054
7B	9.2	7.6	99.3	0.46	0.01	0.05	0.48	21.59	7.68	9.60	4.80	45.6	<0.1	4.80	<10
7C	9.2	7.6	99.3	0.69	0.01	0.07	0.46	19.49	8.35	9.75	4.64	41.8	<0.1	5.11	<10
7D	9.2	11.0	99.2	0.3	0.02	0.07	0.48	17.82	8.67	9.63	4.82	55.4	<0.1	5.30	<10
8A	8.1	83.8	89.3	7.61	0.23	0.23	3.67	50.43	97.20	24.30	155.9	1151	<0.1	9.17	346.5
8B	8.8	15.4	98.6	0.84	0.03	0.09	0.96	22.03	17.72	10.54	28.73	197.3	<0.1	4.79	105.7
8C	9.2	8.7	99.2	0.54	0.02	0.07	0.48	15.43	9.16	8.68	8.20	89.7	<0.1	3.86	36.68
8D	9.3	9.5	99.2	0.43	0.02	0.07	0.49	14.62	9.26	8.77	6.33	75.0	<0.1	3.90	21.78
9A	8.0	103.8	85.2	9.97	0.31	0.27	4.11	66.64	152.0	31.04	224.6	1566	0.1	11.41	399.2
9B	8.8	12.8	98.8	0.65	0.03	0.07	0.93	17.28	14.01	8.87	19.14	142.0	<0.1	4.67	30.68
9C	9.1	8.7	99.3	0.35	0.02	0.07	0.52	28.32	9.97	9.44	8.92	89.7	<0.1	3.67	28.63
9D	9.2	12.8	99.1	0.36	0.02	0.07	0.48	21.28	13.54	12.09	5.80	83.2	<0.1	8.71	<10
10A	7.9	72.8	90.5	6.37	0.2	0.18	2.96	48.84	88.81	22.70	143.6	1066	<0.1	8.88	447.0
10B	8.9	15.5	98.7	0.7	0.03	0.07	0.52	21.78	14.52	10.37	19.70	156.1	<0.1	5.19	42.41
10C	9.2	9.7	99.2	0.48	0.02	0.07	0.52	14.95	9.28	9.28	7.73	67.0	<0.1	3.09	19.34
10D	9.3	8.7	99.1	0.37	0.02	0.05	0.48	17.43	8.71	9.20	4.84	49.4	<0.1	3.39	<10
Reference ^a	8.2	--	--	--	--	--	<0.5	35	10	13.5	6.5	51	<0.1	6.7	--
c _v ^b	2.5%	--	--	--	--	--	20.28%	24.14%	31.44%	--	19.11%	3.38%	--	--	--
DSI ^c	--	--	--	--	--	--	0.5	43	19	15	57	68	0.2	18	50
DSI ^d	--	--	--	--	--	--	7	162	100	90	354	347	7	34	5000

^a (Winiarski et al., 2001).

^b c_v stands for coefficient of variation (calculated from Bedell et al., 2001).

^c DSI stands for Dutch Standards target values (NMHSPE, 2000).

^d DSI stands for Dutch Standards intervention values (NMHSPE, 2000).

The authors concluded that pollutant concentrations decreased rapidly with depth while pH, mineral content, and grain size increased. Metals were concentrated in the top 30 cm of soil except for zinc which is more mobile. Also, hydrocarbon contamination was found to be deeper than most metals. Finally, the highest hydrocarbon concentrations were found near the influent location but the highest heavy metal concentrations were found at the low point of the basin. The authors also proposed a methodology for reducing the number of required samples while still characterizing the soil well.

Datry et al. (2004) also investigated sediment samples. The sediment samples revealed that nitrogen was mostly in organic form (not ammonium) and that zinc, lead, and copper comprised 95% of the heavy metals. The concentrations of metals, nutrients, and hydrocarbons dropped

significantly at a depth of 0.5 m below the bottom of the basin and concentrations in the sediment below the groundwater table were not statistically higher than a control site sediment samples.

Zimmerman et al. (2005) modeled metal (zinc, copper, lead) concentrations in soil and infiltrated water using batch tests and column tests. Breakthrough curves were modeled and compared to laboratory results (Figure 4).

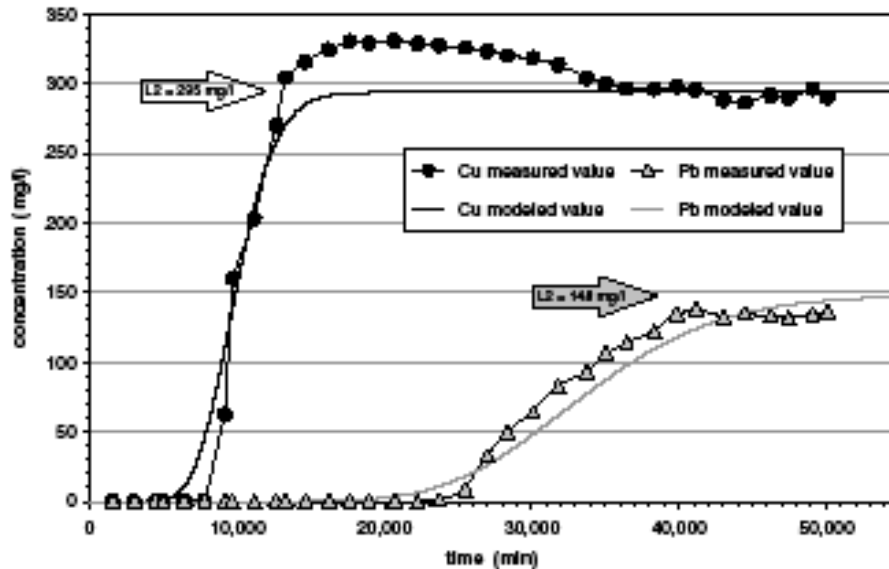


Figure 4 Comparison of model and experimental breakthrough curves (Zimmerman et al. 2005).

To prevent soil concentrations from exceeding German critical values, the authors recommended replacing the first 20 cm of soil after two or three years for highly adsorbing soil and after three or four years for low-adsorbing material, depending on the metal. According to the authors, metals may migrate down to the water body over several years or decades.

Winiarski et al. (2006) investigated the effect of about 20 years of stormwater infiltration on the receiving soil of an infiltration basin. The basin had a surface area of over 7,400 m², a volume of over 30,800 m³, and a depth of about 5.5 m. Soils samples were taken so as to obtain a vertical profile of soil characteristics and pollutant concentrations at three different locations: 1) Near the basin inlet, 2) In the middle of the basin, and 3) At the southern end of the basin. Results are shown in Figure 5 through Figure 10.

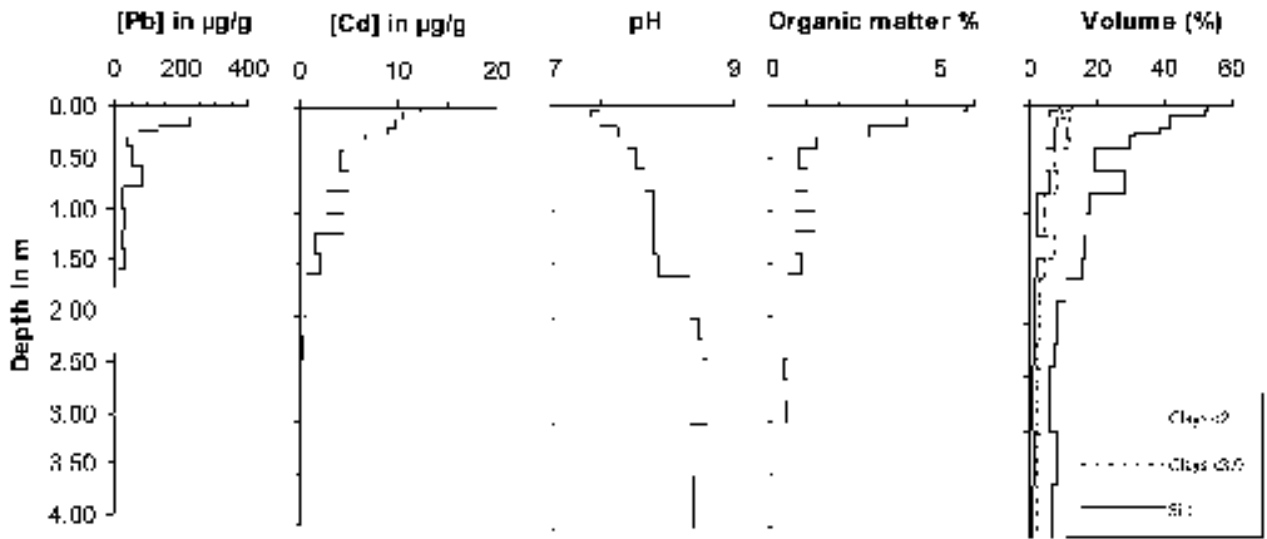


Figure 5 Concentration of metals, pH, clays, silts, and organics at point 1 (Winiarski et al. 2006).

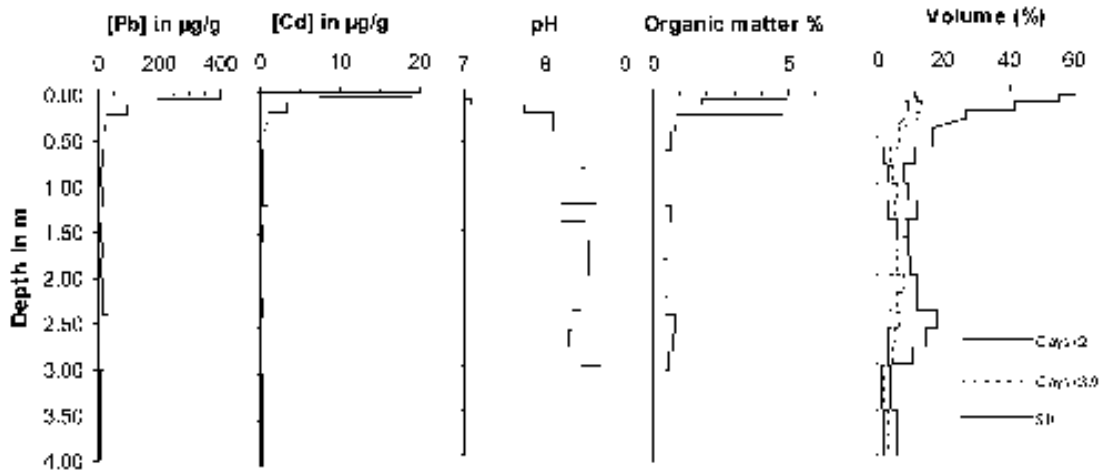


Figure 6 Concentration of metals, pH, clays, silts, and organics at point 2 (Winiarski et al. 2006).

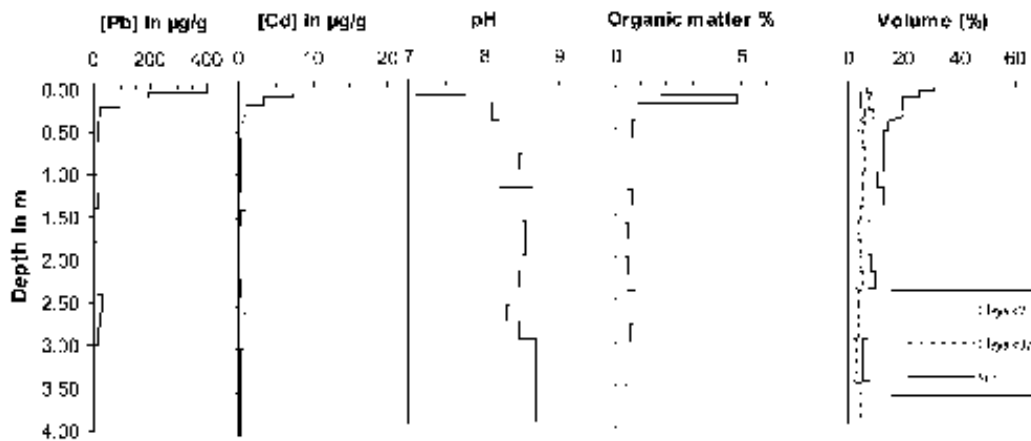


Figure 7 Concentration of metals, pH, clays, silts, and organics at point 3 (Winiarski et al. 2006).

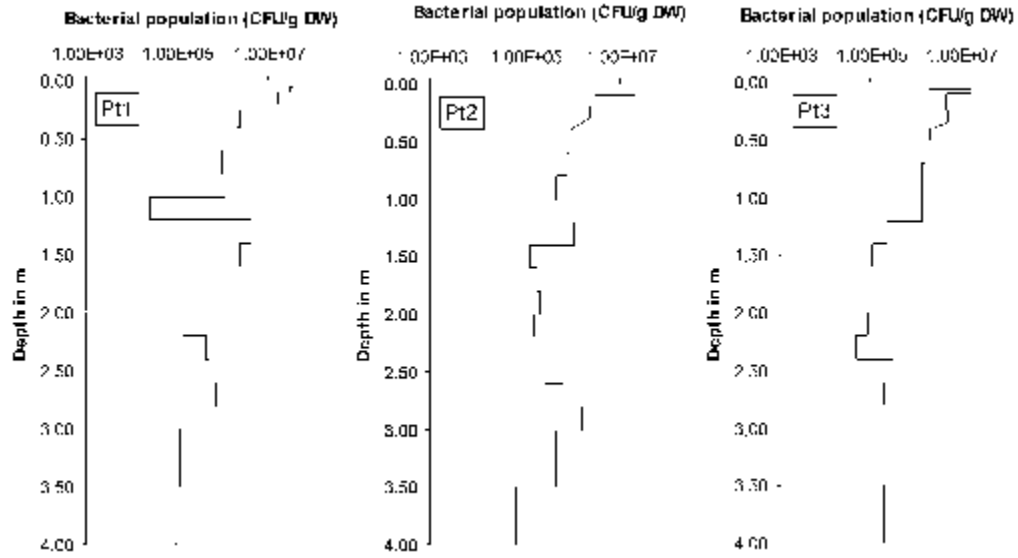


Figure 8 Density of viable heterotrophic bacteria along vertical profiles at points 1, 2, 3 (Winiarski et al. 2006).

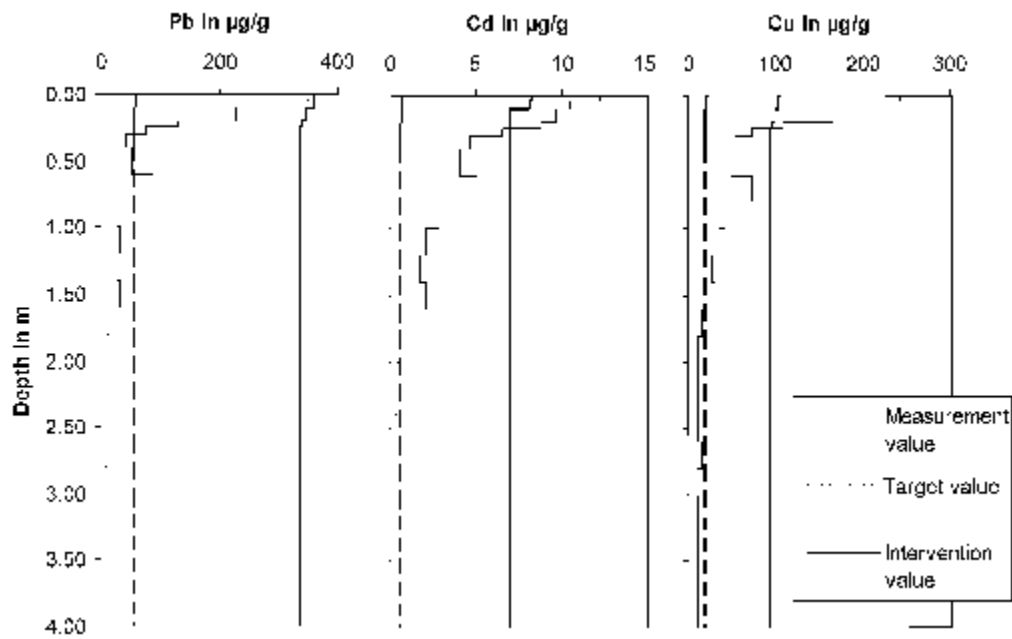


Figure 9 Heavy metal concentrations at point 1 (Winiarski et al. 2006).

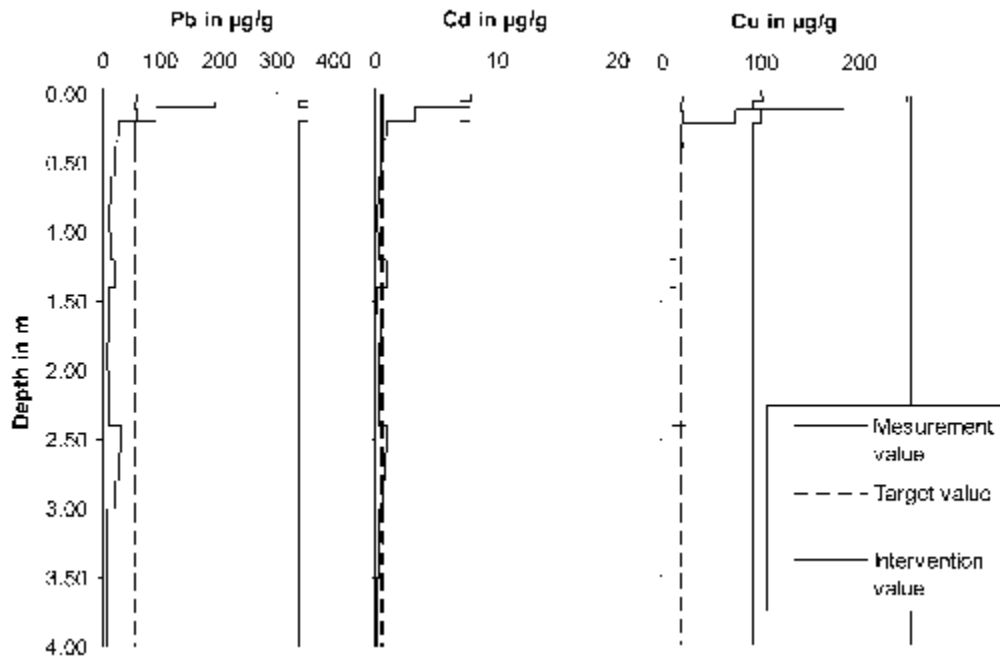


Figure 10 Heavy metal concentrations at point 2 (Winiarski et al. 2006).

The concentrations of metals in the upper soil layers were much higher than control soils. Possible reasons include the presence of carbonates that can retain metals through cation exchange, chemical precipitation with carbonates, or the filtration of particles to which metals are sorbed. The soils also exhibited low pH in the top 1.5 meters which could be due to microorganisms and organic matter. The authors speculate that at point 2 there may be preferential flow paths because the soil has a large capacity to retain pollutants and yet pollutant concentrations increase at a depth of 2.4 to 3.0 meters. If so, preferential flow paths could lead to contamination of groundwater. Also, the relatively high pollutant levels down to a depth of 1.5 meters indicates that the practice of removing a thin layer of top soil during maintenance may need to be expanded to include more soil.

An investigation of 20-year old infiltration facilities in Tokyo, Japan was conducted by Aryal et al. (2006). Based on the high heavy metal content of road dust and sediment in the inlets to the infiltration facilities, the authors concluded that road dust was a major source of heavy metals to the infiltration facilities. After determining the heavy metal profile in the sediment, it was concluded that there was probably leaching of heavy metals into the underlying soils. The concentration of heavy metals found were not above values that typically are considered to be a serious threat, but the authors stated that the leaching of heavy metals to the underlying soil could have serious ramifications.

One study that investigated whether or not plants in a bioretention system uptake heavy metals and, if so, to what extent found that less than 3% of retained metals accumulated in plant tissue, the rest remained in the soil media (Dietz and Clausen, 2006; Sun and Davis, 2007).

Based upon results of column studies, adsorption capacities of soil and mulch were estimated for lead, copper, and zinc (Davis et al., 2001), and breakthrough characteristics of the soil media

were observed. Using loading and capacity estimates, it was estimated that after 20 years concentration levels for cadmium, lead, and zinc reach or exceed levels permitted as per EPA biosolids land application regulations (Davis et al., 2003).

3.C. Model Studies and Literature Reviews

Stephenson and Beck (1995) performed a thorough review of literature related to highway stormwater runoff and the potential threat to groundwater. The literature reviewed in the paper generally agreed that contaminants typically found in highway runoff (and urban runoff, due to similar compositions) can be, but may not always be, removed from infiltrating stormwater by the soil media. The potential for removal is higher in areas with thick layers of soil. The paper goes on to review work that investigated the potential threat of highway runoff to groundwater in karst areas. As may be expected, due to more direct flow paths, groundwater in karst areas is much more susceptible to contamination from runoff, particularly where soils may be very thin or nonexistent. Documented cases of groundwater contamination in karst areas are readily available (Stephenson and Beck, 1995). For example, after a heavy rain event in West Virginia, silt and clay from a construction site was washed into a cavernous aquifer and emerged at a spring. This greatly increased the turbidity of the stream and killed more than 150,000 trout during a single storm event. Spills of diesel fuels at the site killed additional fish.

Paschka et al. (1999) investigated the potential effects of anticaking agents used in road salt on water quality. The major pollutant of concern in anticaking agents is cyanide which may take the form of HCN. Although HCN is usually assumed to leave the water surface quickly because of its volatility, there is not sufficient data in the literature to confirm this assumption. Of all the studies reviewed in this paper, however, none addressed the issue of groundwater contamination by anticaking agents.

Dietz (2007) wrote a review of studies related to stormwater infiltration systems and also discussed the potential of groundwater contamination. This review stated that, for residential and light commercial developments, the common pollutants (i.e. nutrients, petroleum residue, heavy metals, and possibly pesticides) are usually found in low concentrations and are retained by soil so that groundwater contamination is not a concern. Two exceptions to this statement were pathogens and salts. Fecal coliform, it was stated, is often found in high concentrations and may not be retained well by soil media. Also, salts are highly mobile and can easily travel to shallow groundwater. Some studies reviewed by Dietz (2007) indicated that salt concentrations have been increasing in some waterways in the US and, if this trend continues, salt levels will reach levels that are dangerous and could damage the health of the river. Dietz concluded that certain areas may not be good choices for infiltration or other LID technologies. Areas with high contaminant loads such as gas stations or recycling centers, for example, may not be good candidates for LID. Also, locations with steep slopes, shallow depth (< 3 feet) to bedrock, or seasonal high water tables also may not be appropriate for LID.

Clark and Pitt (2007) discussed factors that can influence groundwater contamination by infiltration practices and proposed a means to evaluate contamination potential. The evaluation method contains the following three steps: 1) Determine concentrations and forms of the pollutants entering and leaving the infiltration device, 2) Determine characteristics of the soil that

affect water quality, and 3) Determine required pretreatment actions. The paper also reviews previous articles that address groundwater contamination due to the infiltration of stormwater. This review found that nitrates are believed to be present at concentrations sufficiently low, such that they are not a threat to groundwater quality. Pesticides and other organics, however, may contaminate groundwater if the crops are irrigated and the soil is sandy. Because salts are not removed by soils, groundwater contamination can occur rapidly.

Documented pathogen contamination of groundwater due to infiltration practices was also reviewed and discussed. Contamination potential depends on the soil chemical properties, soil adsorption capability, and the ability of the soil to physically strain the pathogens.

Metals, as in other studies, were found to be mostly removed in the soil within the infiltration practice or in the vadose zone below the structure. Metal removal may occur by soil surface association, precipitation, inclusion with other precipitates, diffusion into soil solids, biological action, or filtration of particle-bound metals.

Clark and Pitt (2007) further state that important soil parameters which affect contaminant transport are organic content, microorganism activity in the vadose zone, porosity, and infiltration capacity. The authors present two methods for predicting groundwater contamination potential; a simple method which involves reading a series of 3 tables and a more complicated computer model. Examples are given for each model.

Wolf et al. (2007) used a detailed computer model analysis and a Monte Carlo simulation to determine the impact of leaky sewers and urban drainage systems on the city of Rastatt in southwest Germany. This study linked separate computer models to exchange data and provide a more holistic approach to subsurface water transport and potential groundwater contamination. The authors concluded that the soil system alone is not sufficient to completely protect urban groundwater from contamination. Based on the results of their study, Wolf et al. (2007) claim that the urban water cycle can be managed in a sustainable manner. For the city of study, Rastatt, Germany, the investigators concluded that the urban drainage system was too small to induce systemic groundwater contamination but that local areas could be contaminated if maintenance was inadequate.

4. Conclusion

An increasing proportion of modern stormwater management practices rely upon infiltration as a method of controlling runoff. The purpose of this literature review is to examine the current state of research regarding possible soil and groundwater pollution caused from stormwater infiltration practices. Research has shown that many of the priority pollutants in urban stormwater runoff have some potential to compromise groundwater supplies. Furthermore, concentrations of the pollutants in the receiving soil may become elevated above acceptable levels. Further research is necessary to determine important management and risk analysis decisions, such as heavy metal breakthrough times or establishment of a media exchange regime. Most important, optimizing pollutant minimization to protect the human and environmental healthy requires consideration of the ultimate fate of stormwater pollutants. Certain pollution

risks are associated with infiltration, but many pollution risks are also associated with the status-quo methods (i.e. discharging to surface water bodies). This review provides an informative reference regarding infiltration practices and the consequential possibilities of pollution, as well as a cornerstone for future and much-needed research in this growing field.

5. References

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