Review of the Science and Practice of Stormwater Infiltration in Cold Climates





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Review of the Science and Practice of Stormwater Infiltration in Cold Climates

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EXECUTIVE SUMMARY

As the practice of stormwater management evolves to better address such issues as channel erosion and aquatic ecosystem protection, there is increasing interest in decentralized micro-controls at or near the source of drainage networks that supplement traditional detention facilities. Alternatively referred to as 'Low Impact Development', 'Sustainable Urban Drainage Systems', 'Water Sensitive Urban Design', or 'Stormwater Source Controls', these approaches attempt to reproduce the pre-development hydrologic regime through site planning and engineering techniques aimed at infiltrating, filtering, evaporating and detaining runoff, as well as preventing pollution.

Stormwater infiltration practices that direct runoff to pervious areas or engineered structures for storage and eventual infiltration are central to these approaches because the infiltration component of the water balance is substantially reduced under most urban development scenarios. These practices can provide multiple benefits where conditions are suitable. They reduce runoff volume and thereby minimize flood risk and prevent alterations to the natural channel and stream flow regime. They help to maintain groundwater levels and sustain stream flows during dry periods. They also reduce pollutant loading to receiving watercourses from runoff by retaining contaminants in the engineered structures and underlying soil.

This review provides an updated summary of the body of knowledge on infiltration based stormwater management. Particular emphasis is placed on peer reviewed journal articles and published reports from jurisdictions with climate and soil conditions similar to Ontario, including the northeastern United States, United Kingdom, France, Norway, Sweden, Denmark, Germany, Austria, Switzerland and Japan. The review begins with a comparison of guidelines regarding the suitability and siting of stormwater infiltration practices from selected cold climate jurisdictions, followed by descriptions of general types of infiltration practices, their typical application, and pretreatment requirements. Stormwater management issues specific to cold climates are then briefly summarized and overviews are provided of typical urban runoff contaminants and the physical, chemical and biological processes by which these may be treated as water percolates through soil. This more theoretical discussion is followed by a review of literature addressing the risk of soil and groundwater pollution from application of stormwater infiltration practices and associated management recommendations. Available information on the performance of each type of infiltration practice is summarized with regard to runoff reduction (*i.e*., hydrologic benefits), surface water quality (*i.e.,* effects on water quality in overflow or underdrain flows), groundwater quality (*i.e*., potential for groundwater contamination) and soil quality (*i.e*., accumulation of contaminants). The final section outlines typical inspection and maintenance requirements for each type of infiltration practice.

Comparison of guidelines on the suitability and siting of stormwater infiltration practices from selected cold climate jurisdictions reveals that while consistent direction is provided regarding the factors that should be considered, specific criteria vary considerably among jurisdictions. Of particular note are differences in direction regarding types of land uses considered to have potential to generate highly contaminated runoff (*i.e*., stormwater hot spots or pollution hot spots) and that are, as a result, unsuitable for application of stormwater infiltration practices. Current stormwater planning and design guidelines in Ontario can be interpreted as blanket restrictions on infiltration practices installed within any industrial or commercial land use, which leaves little flexibility for exceptions. Improving direction in this regard in the updated guideline would reduce a significant barrier to the application of infiltration practices in Ontario.

There are numerous studies documenting the performance of stormwater infiltration practices in cold climate regions. The vast majority of literature reports favorable performance for most parameters examined, suggesting that greater integration of infiltration practices into stormwater management system designs in cold climates could further reduce impacts of urbanization on receiving waters and their aquatic ecosystems. Few studies have examined the performance of bioretention and infiltration chambers after several years of operation. There is also insufficient information regarding the effects on receiving water quality of infiltrating deicing salt laden runoff in small areas distributed across the catchment versus discharging runoff to centralized end-of-pipe facilities. These are topics requiring further research.

A number of common concerns about the performance of stormwater infiltration practices have been addressed in the literature cited in this paper. Concern about the potential for clogging through the accumulation of fine sediments has been addressed through improvements to design, installation and maintenance, as indicated by recent performance monitoring studies. While longer term performance studies are needed, the research to date indicates that stormwater infiltration practices are effective at preserving the predevelopment hydrologic function of a site and removing pollutants from runoff.

Concerns about the effectiveness of infiltration practices in cold climates and on fine-textured soils have been topics addressed in several recent studies on stormwater infiltration technologies. Permeable pavement and bioretention facilities have been observed to function well in cold climates during winter months, even with frost in the ground, albeit at lower efficiencies than during warm weather. While guidelines in some jurisdictions discourage the application of infiltration practices on sites with fine-textured soils containing greater than 20% clay, recent studies have shown that substantial volumes of stormwater can be infiltrated in tight soils beneath permeable pavement installations.

The ability of infiltration practices to remove typical contaminants from urban stormwater is becoming well established, with a few exceptions. High reductions in the concentration (and loads) of suspended solids, metals, polycyclic aromatic hydrocarbons (PAH), and other organic compounds have been consistently observed in performance studies. Observations of effects on nutrient (dissolved nitrogen and phosphorus) concentrations and loads have been more variable. Adapting designs to utilize media with lower or slow-release phosphorus content, combined with pretreatment practices that help to retain nitrates and dissolved phosphorus (*e.g.*, vegetated filter strips, grassed swales), could improve net load reductions for these constituents.

Risk of groundwater contamination from stormwater infiltration practices is the most common concern due to the presence of a wide variety of pollutants in urban runoff. Most pollutants are well retained by infiltration technologies and soils and therefore, have a low to moderate potential for groundwater contamination. The most notable exceptions are de-icing salt constituents (typically sodium and chloride), which are not well attenuated in soil and can easily travel to shallow groundwater. Infiltration of de-icing salt constituents is also known to increase the mobility of certain heavy metals in soil (*e.g.*, lead, copper and cadmium), thereby raising the potential for elevated concentrations in underlying groundwater. However, very few studies that have sampled groundwater below infiltration facilities or roadside ditches receiving de-icing salt laden runoff have found concentrations of heavy metals that exceed drinking water standards. The few instances where this has been observed have received runoff from high traffic areas (*i.e*., large highways) with elevated levels of metals. Some jurisdictions (*e.g*., Maine and Minnesota) consider high traffic areas where large amounts of de-icing salt are used to be unsuitable for the application of stormwater infiltration practices.

With the exception of infiltration basins, most infiltration practices are well distributed across the landscape, rather than centralized in a small area. With a distributed approach there is less potential for runoff to accumulate large masses of pollutants and therefore, the occurrence of elevated, potentially toxic concentrations of pollutants in the soil and groundwater is less likely. Collecting and treating stormwater from high traffic areas and pollution hot spots in centralized detention facilities, while using infiltration practices to treat runoff from roofs and low traffic areas may provide a good margin of safety where groundwater contamination is a concern. While it is prudent to restrict infiltration practices in designated pollution hot spots, broader guidance regarding the suitability and siting of these practices should be provided through detailed technical studies at the watershed, subwatershed and local scales.

Landowners and municipalities have been concerned about soil quality below infiltration facilities and the potential need for future remediation and disposal. Available evidence indicates that small distributed infiltration controls such as permeable pavements do not contaminate underlying soils, even after more than 10 years of operation. Based on limited results it can be concluded that for large centralized infiltration facilities, removal and landfilling of at least the upper 5 centimetres of soil may be required when the facilities are decommissioned.

Based on comparison of stormwater management design guidelines from selected jurisdictions, and consideration of recent research on the performance of infiltration practices in cold climate regions, some recommendations can be made regarding on-going work to update the Ontario guidelines:

- Serious consideration should be given to providing revised criteria for evaluating site suitability for stormwater infiltration practices. This guidance should be provided in an up-front, easy to locate section.
- Evidence that significant runoff reduction can be achieved by infiltration practices on fine-textured soils and that such practices continue to function during much of the winter, suggests that minimum percolation rate of the native soil should not be used as a screening criterion, or that a much lower rate than the current 15 mm/hr should be used. Alternatively, different criteria could be recommended for facilities designed for partial infiltration (with an underdrain) than those designed for full infiltration (no underdrain).
- Consideration should be given to requiring underdrains with adjustable flow restrictors to be installed in facilities located on fine-textured soils with percolation rates less than 15 mm/hr in order to ensure complete drainage of water between rain events and reduce the potential for freezing during winter.
- Acknowledging that infiltration rates of newly built facilities will gradually decrease as they age and accumulate fine sediments, it is recommended that updated guidelines require that such reductions be factored into facility design.
- Clarification regarding the criterion for maximum drawdown time is needed, as the current Ontario guideline is vague in this regard. Such a criterion should consider the typical length of time between storm events in a given geographic location from long-term climate records, and the maximum

acceptable amount of time that standing open water should be allowed to occur in an urban area to prevent mosquito-borne illnesses.

- As previously noted, improved guidance is needed regarding what conditions constitute areas where infiltration practices should not be applied due to risk of groundwater contamination. While blanket restrictions may be appropriate in certain circumstances, broader application decisions should be based on a thorough understanding of present and future groundwater uses, contaminant types and loads and the attenuation capacity of native soils. Guidance regarding the suitability of infiltration practices in communities where water supply is derived from groundwater will also need to consider Ontario drinking water source protection requirements that may prohibit certain types of land use or human activities or require contaminant management plans be put in place within wellhead protection zones.
- Current guidance in the Ontario Stormwater Management Planning and Design Manual indicates that implementation of lot level and conveyance controls can only be used to reduce the active storage volume component of end-of-pipe facilities (*i.e.*, not the permanent pool volume). This criterion should be reviewed in light of the significant runoff volume and contaminant load reductions made feasible through the use of distributed micro controls upstream of the stormwater pond or wetland.

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1.0 INTRODUCTION

Urban development alters the local hydrologic cycle by increasing stormwater runoff and decreasing infiltration and evaporation, through the creation of impervious surfaces and enhanced drainage systems. These higher runoff volumes carry a wide variety of contaminants that when discharged to receiving waters degrade river ecosystems and contaminate swimming areas. While traditional detention facilities such as ponds and constructed wetlands successfully reduce peak flows and improve runoff quality, they do not fully address fundamental changes to the water balance brought about by urbanization and the negative physical and biological effects that stormwater has on aquatic ecosystems and human uses of water resources (Aquafor Beech Ltd., 2006).

As the practice of stormwater management evolves to better address such issues as channel erosion and aquatic ecosystem protection, there is increasing interest in decentralized micro-controls at or near the source of drainage networks that supplement traditional detention facilities. Alternatively referred to as ‘Low Impact Development’, ‘Sustainable Urban Drainage Systems’, ‘Water Sensitive Urban Design’, or ‘Stormwater Source Controls’, these approaches attempt to reproduce the pre-development hydrologic regime through site planning and engineering techniques aimed at infiltrating, filtering, evaporating and detaining runoff, as well as preventing pollution.

Stormwater infiltration practices that direct runoff to pervious areas or engineered structures for storage and eventual infiltration are central to Low Impact Development because the infiltration component of the water balance is substantially reduced under most urban development scenarios. These infiltration practices can provide multiple benefits where conditions are suitable. Chief among these is their ability to reduce the volume of runoff discharged to watercourses, thereby minimizing flood risk and preventing alterations to the natural channel and stream flow regime. They help to maintain groundwater levels and sustain stream flows during dry periods (baseflow). They also reduce pollutant loading to receiving watercourses from contaminated runoff by retaining contaminants in the engineered structures and underlying native soil.

Significant pollutant removal is possible with infiltration practices due to the “first flush” effect. This effect occurs when pollutants that have accumulated on impervious surfaces between storms are mobilized by the first runoff that occurs during each storm. Since a large proportion of the pollutant load is carried in the “first flush”, and this initial volume of runoff is a relatively small proportion of total runoff from any given storm, infiltration controls designed to treat the first flush could potentially achieve major reductions to loads of many types of pollutants.

Despite the multiple benefits that stormwater infiltration practices can provide, widespread application as part of urban stormwater management strategies has only occurred to a limited extent in North America, while in Europe, soakaways, infiltration basins and trenches have been in use for over thirty years. Widespread adoption of infiltration practices has been particularly slow in cold climate regions, where documented experience regarding their feasibility and effectiveness during the winter season has, until recently, remained limited.

The cautious adoption of stormwater infiltration practices in North America can largely be attributed to the low rate of success experienced by early adopters during the 1980s and early 1990s. Failures were often due to clogging from sediments during construction, excess compaction of the filter media, high groundwater tables and location of practices on unsuitable native soils. Field surveys of infiltration facilities constructed in Maryland between the mid to late 1980s revealed that half the facilities were not functioning as designed and that two-thirds of them needed maintenance after less than 6 years of operation (Lindsey *et al.*, 1992). The main reason noted for facility failure was inadequate buffer strips (*i.e.*, pretreatment) resulting in sediment accumulation and clogging. Many facilities were not being properly maintained, suggesting that more frequent inspections and modification of regulatory and institutional arrangements dealing with inspection, maintenance and enforcement were needed. Based on observations of very high failure rates of permeable pavement installations and infiltration basins, it was suggested that use of such practices be discouraged except in extraordinary circumstances (Lindsey *et al.*, 1992).

Building on past experiences, infiltration facility designs have since been improved, and guidelines now recommend careful testing of soil and groundwater conditions, protection of facilities during construction and pretreatment of runoff to prevent clogging of the filtration media and attenuate contaminants (*e.g.*, MDE, 2000). These measures have substantially improved success rates (see section 6 for further details).

With the resurgence of infiltration practices in the 1990s, came a number of comprehensive summaries of stormwater infiltration practices (Ferguson, 1994) and their potential to contaminate groundwater (Pitt *et al.*, 1996; Pitt *et al.*, 1995). These summaries provide an overview of the early literature on the subject and establish a framework for assessing the benefits, risks and maintenance requirements associated with some of the more common practices. The work on groundwater contamination potential was subsequently reproduced, updated and elaborated in shorter journal publications (Pitt *et al.,* 1999; Clark and Pitt, 2007). Other reviews (USEPA, 2000; Dietz, 2007) have focused on Low Impact Development approaches and practices more generally.

The intent of this review is to provide an updated summary of the body of knowledge on infiltration based stormwater management to inform the Ontario Ministry of the Environment process that is currently underway to update the Stormwater Management Planning and Design Manual (OMOE, 2003a). Particular emphasis is placed on peer reviewed journal articles and published reports from jurisdictions with climate and soil conditions similar to Ontario, including the northeastern United States, United Kingdom, France, Norway, Sweden, Denmark, Germany, Austria, Switzerland and Japan. Potential implications of findings from this summary on the existing Ontario Ministry of the Environment guidelines are noted and topics for future research are suggested.

2.0 OVERVIEW OF EXISTING GUIDELINES FOR EVALUATING SUITABILITY OF STORMWATER INFILTRATION PRACTICES

Technical guidance documents on stormwater management planning and design have been provided in many cold climate jurisdictions in North America and Europe. In Ontario, stormwater management has been practiced for flood control purposes since the 1970s, with the introduction of requirements for stormwater quality treatment in the early 1990s (OMOEE & OMNR, 1991). The Ontario Ministry of the Environment 2003 Stormwater Management Planning and Design Manual, updates a 1994 version. Evolution of this technical guidance is necessary as design approaches and practices evolve to better protect humans and aquatic ecosystems.

Table 2.1 compares guidelines from selected cold climate jurisdictions on the suitability and siting of stormwater infiltration practices. The comparison reveals that while consistent direction is provided regarding the factors that should be considered in evaluating site suitability, specific criteria vary considerably among jurisdictions. Of particular note are differences in direction regarding the types of land uses that are considered unsuitable for application of stormwater infiltration practices. Some guidelines, such as in Ontario and Alberta, caution against the use of infiltration practices in all commercial and industrial developments, whereas most others suggest restrictions only in stormwater 'hot spots', such as gas stations and loading yards, where spills or leaks of organic compounds are more likely. Although most guidelines provide set back distances from drinking water wells (typically 31 metres), only the guideline from Maine restricts infiltration practices in the capture zone of groundwater drinking wells. The Pennsylvania and Maryland guidelines are more flexible in allowing for infiltration practices in designated 'hot spots' if appropriate pre-treatment is provided or contaminant management plans are in place.

The guidelines also provide inconsistent criteria on minimum percolation rate (*i.e.*, infiltration rate) of native soil, the types of structures or features requiring minimum setbacks, and the maximum time for runoff to fully drain. Some regulatory agencies have taken the approach of not using a minimum percolation rate as a criterion of suitability (*e.g.*, BC MWLAP, 2002; MPCA, 2008; CIRIA, 2007), or have suggested a minimum rate that is much lower than in many other jurisdictions (*e.g.*, PDEP, 2006). The variation in criteria may reflect differences in the range of infiltration practice designs and benefits being considered under the guidelines. A system designed to infiltrate runoff without underdrains would normally require a minimum percolation rate of 13 millimetres per hour (mm/hr) or greater to ensure runoff is drained over the desired time period (usually 24 to 72 hours). The Ontario, Alberta, Maryland, Maine, Michigan and New York guidelines would be appropriate for this type of 'full infiltration' system. At lower percolation rates, underdrains would be needed and the amount of runoff infiltrated would normally be less, but there would still be some runoff, peak flow and water quality control. This type of 'partial infiltration' system would be permitted under the less restrictive British Columbia, Minnesota and United Kingdom guidelines.

In his review of the practice of stormwater infiltration, Ferguson cautions against the approach of evaluating suitability of infiltration practices based solely on hydraulic conductivity of the native soil (1994):

> “to say that it is not feasible to use a fine-textured soil for infiltration because it is slowly permeable is like saying that it is not feasible to use corrugated metal pipe for carrying water because corrugated metal has a high roughness factor and therefore carries less water”.

He goes on to note that low hydraulic conductivity by itself does not make a soil unsuitable for infiltration; rather, conductivity must be taken into account in the design. He suggests that there are only a few contexts in which infiltration practices should not be given consideration (1994):

- Soil as impermeable as the roofs and pavement that will be placed upon it;
- Highly contaminated soils (*e.g.*, toxic waste or saline deposits);
- Steep unstable slopes; and,
- Land in close proximity to water supply wells, septic tanks, basements or other sensitive structural foundations.

Others indicate that infiltration practices such as permeable pavements that require soils to bear heavy loads, would not be suitable on clay soils of high plasticity (*e.g.*, Smith, 2006).

It is also notable that many stormwater management design guidelines recommend limiting the application of infiltration practices to coarse-textured soils even though these types of soils typically attenuate many contaminants, particularly metals, less effectively than fine-textured soils (see section 4 for further details).

Few jurisdictions in Ontario stipulate mandatory requirements for infiltration. The City of Toronto requires that all runoff from small rainfall events (up to 5 millimetres) be retained onsite through infiltration, evapotranspiration and rainwater reuse (City of Toronto, 2006). Most other jurisdictions promote infiltration as a preferred means of meeting stormwater management goals, but stop short at specifying a volume requirement for new or redevelopment projects.

Table 2.1: Guidelines for stormwater infiltration practice suitability and siting constraints from cold climate jurisdictions**

Jurisdiction	Reference	Groundwater contamination risk	Minimum native soil percolation rate (fully saturated)	Minimum depth between infiltration facility base and seasonal high water table or bedrock	Minimum separation distance	Other
Ontario	Ontario Ministry of the Environment (2003)	Not recommended for use in industrial developments, nor commercial parking areas.	15 mm/hr (60 mm/hr for infiltration basins)	1 metre	- 4 metres from building foundations. - Setbacks from wells as specified in the Building Code for leaching bed systems.	- Generally 24 to 48 hour drawdown.
British Columbia	British Columbia Ministry of Water, Land and Air Protection (2002)	Not recommended for use in any stormwater quality hot spot[1].	- No restrictions. - Underdrain recommended where soils are relatively impermeable.	0.5 metre	- 3 metres from building foundations (5 metres for heavy clay soils).	
Alberta	Alberta Environmental Protection (1999)	Not recommended for use in commercial or industrial developments.	15 mm/hr	1 metre	- Consideration should be given to proximity to septic fields.	- Should drain the system within 48 hours of the storm event. - Should not be located on non-native fill material.
Halifax Regional Municipality	Halifax Regional Municipality (2006)		15 mm/hr (60 mm/hr for infiltration basins)	1 metre	- 4 metres from building foundations. - 10 metres down-gradient from septic systems.	- Runoff from 25 mm rain event should drain in 24 to 48 hours. - Runoff from 2 year storm event should drain in 72 hours.
Maine	Maine Department of Environmental Protection (2006)	- Should not be used in manufacturing and industrial areas. - Should not receive construction site runoff. - Should not receive dry weather storm drain effluent. - Should not receive combined sewer overflows. - Should not receive snowmelt runoff from areas subject to or adjacent to road traffic or parking.	13 mm/hr (no greater than 61 mm/hr).	- 0.91 metre (3 feet) to water table. - 1.5 metre (5 feet) to bedrock for facilities serving 0.4 hectare (1 acre) or more of impervious area.	- 91 metres (300 feet) from any private water supply well. - Outside delineated contributing areas of public water supply wells. - 3 metres (10 feet) from any water supply conduit. - 15.2 metres (50 feet) from downhill slopes greater than 3:1. - 3 metres (10 feet) from a 10 year floodplain. - 7.6 metres (25 feet) from the property line.	- Shall not be located on slopes greater than 20%. - Must drain completely within 72 hours following the runoff event.

Jurisdiction	Reference	Groundwater contamination risk	Minimum native soil percolation rate (fully saturated)	Minimum depth between infiltration facility base and seasonal high water table or bedrock	Minimum separation distance	Other
Maryland	Maryland Department of the Environment (2000)	Runoff from designated hot spot land uses or activities[2] cannot be infiltrated without proper pretreatment to remove hydrocarbons, trace metals, or toxicants.	13 mm/hr (clay content of less than 20% and a silt/clay content of less than 40%).	1.2 metre (4 feet)	- 7.6 metres (25 feet) from structures. - 30.5 metres (100 feet) from any water supply well.	- Cannot be located on slopes greater than 15% or within fill soils. - May be prohibited within areas of karst topography. - Maximum contributing area should generally be less than 2 hectares (5 acres). - Must drain water quality volume within 48 hours of storm event.
Michigan	Michigan Department of Environmental Quality (1998)	-Should not be used in industrial parks, high density or heavy industrial areas, areas with chemical or pesticide storage, and fueling stations	- 13 mm/hr (clay content less than 30%) for basins and trenches - 7 mm/hr for permeable pavement.	1.2 metre (4 feet)	- 30.5 metres (100 feet) from drinking water wells. - 30.5 metres (100 feet) up-gradient from building foundations.	- Shall not be located on slopes greater than 20%. - Should drain completely within 72 hours following the storm event. - Should not be constructed in areas which have been filled in.
Minnesota	Minnesota Pollution Control Agency (2008)	- No infiltration from potential stormwater hot spots[3] or high traffic areas. - In potential stormwater hot-spots an impervious liner should be used.	- No restrictions. - Minimum of 5 mm/hr is highly recommended. - Underdrain highly recommended where less than 25 mm/hr	0.91 metre (3 feet)	- 3 metres (10 feet) from building foundations. - 15 metres (50 feet) from wells. - 10.7 metres (35 feet) from septic system beds.	- Should not be located on slopes greater than 15%. - Not recommended in active karst regions (karst feature within 15 metres of surface)
New York State	New York State Department of Environmental Conservation (2003)	- Should not receive runoff from stormwater hot spots[4].	13 mm/hr (clay content less than 20% and silt/clay content less than 40%).	0.91 metre (3 feet)	- 30.5 metres (100 feet) from water supply wells. - 7.6 metres (25 feet) down gradient from structures or septic systems (basins and trenches). - 3 metres (10 feet) from structures (dry wells)	- Cannot be located on areas with natural slopes greater than 15%. - Cannot be located in fill soils. - Must drain water quality volume within 48 hours of storm event.

Jurisdiction	Reference	Groundwater contamination risk	Minimum native soil percolation rate (fully saturated)	Minimum depth between infiltration facility base and seasonal high water table or bedrock	Minimum separation distance	Other
Pennsylvania	Pennsylvania Department of Environmental Protection (2006)	- Infiltration may occur in stormwater hot spots[5] provided pretreatment is suitable to address groundwater contamination concerns.	2.5 mm/hr (not greater than 254 mm/hr)	0.61 metre (2 feet)	- 15.2 metres (50 feet) from individual water supply wells. - 30.5 metres (100 feet) from community or municipal water supply wells. - 3 metres (10 feet) down gradient or 30.5 metres (100 feet) up gradient from building basement foundations. - 15.2 metres (50 feet) from septic system drain fields.	- Should not be placed on recent fill or compacted fill. - Should drain water quality volume within 72 hours of storm event.
United Kingdom	CIRIA (2007)	- Not recommended for pollution hot spots[6]. - Should not receive construction site runoff.	- No restrictions (soils with less than 5 mm/hr are considered poor infiltration media). - Appropriate factor of safety should be applied to measured rates for facility design.	1 metre	- 5 metres from foundations.	- Should not be used where soil or groundwater is contaminated. - Should drain to half full volume within 24 hours of storm event.

Notes:

1. In B.C. *stormwater quality hot spots* include gas stations, wrecking yards, fleet storage yards, or other sites that store hazardous materials.
2. In Maryland *designated stormwater hot spots* are vehicle salvage yards and recycling facilities, vehicle service and maintenance facilities, vehicle and equipment cleaning facilities, fleet storage areas (bus, truck, etc.), industrial sites, marinas (service and maintenance), outdoor liquid container storage, outdoor loading/unloading facilities, public works storage areas, facilities that generate or store hazardous materials, commercial container nursery, or other land uses and activities as designated by an appropriate review authority. Large highways (average daily traffic volume greater than 30,000) and retail gasoline outlet facilities are not designated as stormwater hot spots but require stormwater management plans to adequately protect groundwater.
3. In Minnesota *potential stormwater hot spots* include vehicle salvage yards and recycling facilities, vehicle service and maintenance facilities, vehicle and equipment cleaning facilities, fleet storage areas (bus, truck, etc.), industrial sites, marinas (service and maintenance), outdoor liquid container storage, outdoor loading/unloading facilities, public works storage areas, facilities that generate or store hazardous materials, commercial container nursery, large parking lots, transportation routes with a history of contaminated runoff, fueling areas, and large chemically managed turf areas.
4. In New York State *stormwater hot spots* are classified as vehicle salvage yards and recycling facilities, vehicle service and maintenance facilities, vehicle and equipment cleaning facilities, fleet storage areas (bus, truck, etc.), industrial sites, marinas (service and maintenance), outdoor liquid container storage, outdoor loading/unloading facilities, public works storage areas, facilities that generate or store hazardous materials, commercial container nursery, vehicle fueling stations, or other land uses and activities as designated by an appropriate review authority.
5. In Pennsylvania *stormwater hot spots* are generally considered to be vehicle fueling areas, vehicle service and maintenance areas, some industrial sites, high traffic roadways, public works storage areas, fast food parking lots, and trash dumpsters.
6. In the U.K. *pollution hot spots* are generally considered to be industrial development areas such as rubbish skip areas (dumpsters), areas where chemicals and oils may be spilled, delivery bays where there is a high risk of spillage, designated pressure washing areas, and fuelling areas.

3.0 GENERAL TYPES OF STORMWATER INFILTRATION PRACTICES

There are a wide variety of different types of stormwater infiltration practices. These can be grouped according to their location within the drainage network, and together with other practices, are often implemented as part of an overall "treatment train" for a given development. *Lot level* stormwater infiltration practices are applied at the scale of the individual property or site to treat runoff where it is being generated. They include such controls as permeable pavement, vegetated filter strips, bioretention, soakaways, and infiltration trenches and chambers. *Conveyance* stormwater infiltration practices are part of the system of stormwater infrastructure that transports urban drainage from individual lots to a treatment facility and ultimately, to the receiving watercourse or waterbody. Conveyance infiltration practices include such controls as grassed swales and pervious pipe and catchbasin systems (*e.g.*, exfiltration systems). *End-of-pipe* stormwater infiltration practices receive and treat drainage from a conveyance system and typically service multiple lots or entire subdivisions. End-of-pipe infiltration practices are generally limited to infiltration basins. It should be noted that there are also a variety of non-structural or preventative measures, such as alternative development site layout, impervious area disconnection and natural area preservation that can limit the reduction in pre-development infiltration potential of a site following development. Although very important, these measures lie beyond the scope of this review.

Table 3.1 provides descriptions of the various types of stormwater infiltration practices, the contexts in which they are typically applied, pretreatment requirements, and the typical ratio of impervious drainage area to treatment facility area. The ratio of impervious drainage area to treatment facility area provides an indication of the extent to which runoff is concentrated for each best management practice (BMP) type, with higher ratios implying higher levels and rates of contaminant accumulation. For example, end-of-pipe controls that are typically used to treat runoff from multiple lots or entire subdivisions have the highest ratios of impervious drainage area to facility area and highest contaminant accumulation rates. At the other end of the spectrum are permeable pavements, which have very low ratios and receive considerably less runoff because often they treat only the rainfall that falls on their surface.

Table 3.1: Overview of Stormwater Infiltration Best Management Practice (BMP) Types[1]

BMP Type	Description	Application	Pretreatment	Typical Ratio of Impervious Drainage Area to Treatment Facility Area
Lot Level				
Permeable pavement	A pervious pavement surface underlain by a uniformly graded stone bed reservoir. The surface course may consist of permeable asphalt, permeable concrete, permeable interlocking concrete pavers, concrete grid pavers and plastic grid pavers. Openings in permeable interlocking concrete pavers, concrete grid pavers and plastic grid pavers are typically filled with pea gravel, sand or top soil and grass. The stone bed includes an overflow control structure and may also include an underdrain if native soil percolation rates are very low. Also referred to as porous pavement or pervious pavement.	Most appropriate in low to medium traffic areas (*e.g.*, residential roads, parking lots, driveways, walkways, plazas, playgrounds, boat ramps etc.). Can be designed to treat limited amounts of roof runoff in addition to road, parking and walkway runoff. Should not receive runoff from pollution hot spots[2] areas or active construction sites. In cold climates, the base of the stone reservoir should be below the frost line to reduce the risk of frost heave.	The pavement itself acts as pretreatment to the stone reservoir bed below. Because of this, frequent maintenance of the surface, such as vacuum sweeping, is critical to prevent clogging. A layer of fine gravel can be laid atop the coarse gravel treatment reservoir to provide additional pretreatment.	1:1 to 5:1[4,5]
Vegetated filter strip	Gently sloping, densely vegetated areas that are designed to treat runoff as sheet flow from adjacent impervious surfaces. Filter strips function by slowing runoff velocities and filtering out sediment and other pollutants, and by providing some infiltration into underlying soils. Filter strips may be comprised of a variety of trees, shrubs, and native vegetation to add aesthetic value as well as water quality benefits. Level spreading devices or other measures may be required to provide uniform sheet flow conditions at the interface of the filter strip and the adjacent land cover. Also referred to as grassed filter strip or buffer strip.	Best suited to treating runoff from roads and highways, roof downspouts and small parking lots. They are also ideal as pretreatment to another structural practice. Filter strips are often impractical in ultra-urban areas[3] because they consume a large amount of space. Should not receive pollution hot spot[2] runoff. In cold climates, filter strips provide a convenient area for snow storage and treatment.	A pea gravel diaphragm (a small trench running along the top of the filter strip) should be used at the top of the slope which settles out sediment particles and acts as a level spreader, helping to maintain sheet flow.	5:1

BMP Type	Description	Application	Pretreatment	Typical Ratio of Impervious Drainage Area to Treatment Facility Area
Bioretention	A shallow excavated surface depression containing mulch and a prepared soil mix and planted with specially selected native vegetation that captures and treats runoff. During storms, runoff ponds in the depression and gradually filters through the mulch and prepared soil mix and root zone. The filtered runoff can either infiltrate into the native soil or be collected in a perforated underdrain and returned to the storm sewer system. They remove pollutants from runoff through filtration in the soil and uptake by plant roots and can help to reduce runoff volume through evapotranspiration and full or partial infiltration. They can also provide wildlife habitat and enhance local esthetics. Also referred to as rain garden, bioswale or biofilter.	Most suitable for treating roof, road, parking and walkway runoff from small drainage areas. Commonly located in parking lot islands or within small pockets of residential land uses. Ideally suited to many ultra-urban areas. Can be used to treat pollution hot spots[2] as long as an impermeable liner is used at the bottom of the filter bed. In cold climates, bioretention areas can be used as snow storage areas.	Often, runoff is directed to a vegetated filter strip or grass channel to filter out coarse materials before the runoff flows into the filter bed of the bioretention area. Other features may include a pea gravel diaphragm, which acts to spread flow evenly and drop out larger particles.	5:1 to 15:1
Soakaway	A square or circular excavation lined with geotextile filter cloth and/or other perforated storage structure, and filled with clean granular stone or other void forming material that receives runoff and allows it to infiltrate into the native soil (CIRIA, 2007; OMOE, 2003a). Also referred to as a dry well or infiltration trench.	In North America, a soakaway is typically considered to be an infiltration trench that serves an individual lot and only receives roof runoff (City of Toronto, 2002; OMOE, 2003a). They are most commonly used for residential roof runoff. In the U.K. "linear soakaways" or infiltration trenches are used for runoff from commercial developments and highways with suitable design and pretreatment (CIRIA, 2007). Can be suited to many ultra urban areas. Should not receive pollution hot spot [2] runoff. Can also be used to manage overflows from rain barrels or other rainwater harvesting systems.	If only roof runoff is received, which typically contains low pollutant loads, there is no need for pretreatment. A removable filter can be incorporated into the roof leader below the overflow pipe to prevent leaves and debris from entering the soakaway pit (OMOE, 2003a). In the U.K. oil and grit separators are used as pretreatment in commercial and highway applications (CIRIA, 2007).	5:1 to 10:1[4]
Infiltration trench	A stone aggregate filled trench that is lined with geotextile filter cloth and receives runoff through some combination of pretreatment measures. There, runoff is stored in the void space between the stones and infiltrates into the native soil. Also referred to as an infiltration gallery or linear soakaway. Infiltration trenches that contain perforated pipes and provide conveyance functions are referred to as pervious pipe or exfiltration systems.	Typically designed to treat roof, road, parking and walkway runoff from relatively small drainage areas with high impervious cover. Suitable for many ultra urban areas. Typically used to capture and treat runoff from small storm events. Should not receive pollution hot spot [2] runoff. In cold climates, if trenches receive runoff from surfaces subject to spreading of de-icing salt it may be desirable to divert flow around the trench in the winter to prevent infiltration of salt laden runoff.	If receiving road and walkway runoff, multiple pretreatment measures in series should be incorporated using practices such as grassed swales, vegetated filter strips, oil and grit separators, sand or organic filters, plunge pools and/or detention ponds.	5:1 to 10:1

BMP Type	Description	Application	Pretreatment	Typical Ratio of Impervious Drainage Area to Treatment Facility Area
Infiltration chamber	Includes a range of proprietary manufactured modular structures installed subsurface typically under parking or landscaped areas that temporarily store stormwater runoff, allowing it to infiltrate into an optional granular stone reservoir and the underlying native soil. Structures typically have open bottoms and perforated side walls and can be installed individually or in series in trench or bed configurations. (*e.g.*, Cultec Inc., 2009). Also referred to as infiltration tank.	Well suited to commercial, industrial or institutional lots and ultra urban areas where lands available for other types of BMPs are limited. Can treat roof, road and walkway runoff with adequate pretreatment. Can be designed to capture and treat runoff from medium sized storm events. Should not receive pollution hot spot[2] runoff. In cold climates, if chambers receive runoff from surfaces subject to spreading of de-icing salt it may be desirable to divert flow around the chamber in the winter to prevent infiltration of salt laden runoff.	If receiving road and walkway runoff, multiple pretreatment measures in series should be incorporated using practices such as grassed swales, oil and grit separators, and/or other types of filters (sand, organic, sediment etc.).	20:1[6]
Conveyance				
Grassed swale	Vegetated, open channels designed to convey, treat and attenuate runoff. Vegetation in the swale slows the water to allow sedimentation, filtration through the soil matrix and root zone, and infiltration into the underlying native soil. Specific designs can vary but all improve upon traditional drainage ditches. Designs incorporate modified geometry and check dams that make grassed swales both a treatment and conveyance practice. Dry swales incorporate an engineered soil bed and perforated pipe underdrain. Also referred to as grassed channel, vegetated swale, dry swale, wet swale and bioswale.	Well suited for treating highway or residential road runoff because they are linear practices. Typically used to treat small drainage areas (less than 2 hectares). Swales are also useful as one of a series of BMPs or as part of a treatment train. May not be well suited to ultra-urban areas because they require a relatively large area of pervious surfaces. With the exception of the dry swale design, pollution hot spot[2] runoff can be directed to grassed swales. In cold climates, swales can also be used as snow storage areas.	Small forebay should be used at the front of the swale to trap incoming sediments. A pea gravel diaphragm (a small trench filled with river-run gravel) should be constructed along the length of the swale and used as pretreatment for runoff entering the sides of the swale.	5:1 to 10:1[4]
Perforated catchbasin	A catchbasin that is connected to a granular stone reservoir by a pervious pipe or where the catchbasin sump is perforated, allowing runoff to gradually infiltrate into the native soil (adapted from OMOE, 2003a and GVRD, 2005). Also referred to as soakaway manhole, pervious catchbasin, percolation drainage system and exfiltration system.	Suitable for treatment of runoff from residential, commercial, institutional and parkland areas (BC MWLAP, 2002). Not suited for treatment of runoff from parking lots and heavy traffic areas unless there has been adequate pretreatment to remove hydrocarbons and heavy metals (GVRD, 2005).	Large catchbasins with deep sumps or oil and grit separators help to pre-treat runoff before it is conveyed to the pervious catchbasin but other measures should also be used before the runoff enters the sewer system (*e.g.*, permeable pavement, bioretention, etc.).	5:1 to 10:1[7]

BMP Type	Description	Application	Pretreatment	Typical Ratio of Impervious Drainage Area to Treatment Facility Area
Perforated pipe	A stormwater conveyance pipe that is perforated along its length, installed in a granular stone bedding, allowing exfiltration of water into the native soil through the pipe wall as it is conveyed. Should be constructed with anti-seepage collars to discourage exfiltrated water from flowing along the bedding to the outlet. (OMOE, 2003a). Also referred to as pervious pipe, third pipe, percolation drainage system and exfiltration system.	Intended to treat drainage from low to medium traffic areas which can contain high levels of suspended sediment. Should not receive pollution hot spot[2] runoff. Should be located in areas of relatively flat or gentle slope (0.5%).	Pretreatment of road runoff is necessary before it reaches the pervious pipe system to reduce risk of clogging and potential for groundwater contamination. Should include multiple practices in series such as permeable pavement, bioretention, vegetative filter strips and grassed swales to treat runoff before it enters the sewer system (Adapted from OMOE, 2003a).	5:1 to 10:1[7]
End-of-pipe				
Infiltration basin	A shallow impoundment designed to infiltrate runoff into the native soil. Can have high pollutant removal efficiency and help recharge aquifers, thus maintaining baseflow to streams. Also referred to as rapid infiltration basin or infiltration pond.	Should be used to treat small drainage areas (less than 4 hectares). Rarely applied in ultra-urban areas due to the need for native soils with high percolation rates, potential for interference with underground infrastructure and the considerable land area required. Should only receive pollution hot spot [2] runoff if adequate pretreatment is provided. In cold climates, underdrains and level control valves can be used to drain the basin at the beginning of winter and to convert it to a detention facility to treat snowmelt during spring freshets. To prevent infiltration of runoff laden with de-icing salt the facility should be disconnected from the storm sewers during winter months.	Pretreatment of runoff is necessary before it reaches the infiltration basin to reduce risk of clogging and potential for groundwater contamination. Should include multiple practices in series such as permeable pavement, bioretention, vegetative filter strips, grassed swales, oil and grit separators, etc. to treat runoff before it enters the sewer system.	30:1 to 114:1[8]

Notes:
1. Adapted from USEPA, 2008 unless otherwise noted.
2. Pollution hot spots are stormwater source areas where certain land uses or activities have the potential to generate highly contaminated runoff (e.g., vehicle fuelling, servicing or demolition areas, outdoor storage or handling areas for hazardous materials, high traffic roads or parking lots, some heavy industry sites).
3. Ultra-urban areas are densely developed urban areas in which few pervious surfaces exist
4. Greater Vancouver Regional District, 2005.
5. Smith, 2006.
6. Estimated based on an infiltration chamber in Richmond Hill, Ontario (TRCA, 2009).
7. Estimated based on Etobicoke Exfiltration System and North York Exfiltration System in Toronto, Ontario (SWAMP, 2005).
8. Based on Nightingale, 1987b; Appleyard, 1993; Datry *et al.*, 2004.

4.0 TYPICAL CONTAMINANTS IN URBAN RUNOFF, COLD CLIMATE EFFECTS, AND STORMWATER INFILTRATION TREATMENT PROCESSES

Contaminants present in urban stormwater runoff vary depending on the specific land uses and activities occurring in the source area. Many studies have examined the quality of stormwater runoff from particular land uses and activities with most efforts focused on contaminants relevant to known pollution problems in receiving waters (*e.g.* CRSDCWP, 2008; Burton and Pitt, 2002).

Release of untreated urban stormwater runoff to receiving waters may cause physical, chemical, biological and combined effects that impair their quality and thereby affect aquatic ecosystems and their beneficial use by humans. These effects differ in various climatic regions, but seem to be particularly severe during snowmelt or rain-on-snow events occurring in alpine and some temperate climates (Marsalek *et al.*, 2003). During cold weather, precipitation accumulates on surfaces in the form of snow and ice (*i.e.*, snow pack), which stores water, chemicals, solids and other materials. In addition, rates of chemical and material accumulation on urban surfaces tend to be higher during cold weather due to higher releases from sources such as fossil fuel combustion for heating, enhanced corrosion, less efficient operation of vehicles, and application of road de-icing salt and sand or gravel as anti-skid agents (Malmquist, 1978). Sand or gravel spreading adds large particulate loads to snowmelt and can contribute to phosphorus and metal loads as well (Oberts, 1986). In dense urban areas, large volumes of snow are often transported to central snow dumps, thereby concentrating associated contaminants further.

Catchment conditions in winter are characterized by reduced infiltration due to the presence of frost in the uppermost soil horizon and corresponding increases in the total area generating runoff during a melt or rain-on-snow event. In cold climates, snowmelt or rain-on-snow events are often the largest runoff events of the year, in terms of both volume and pollutant loads and concentrations (Oberts, 1994). During such events, accumulated water and chemicals may be suddenly released and contribute to acute and chronic impacts on receiving waters (*e.g.*, Environment Canada and Health Canada, 2001). Dissolved pollutants that are preferentially eluted early in the melt event are usually more toxic and far more mobile than pollutants associated with particulates that tend to be left behind, thus, exerting a “first flush” of harmful contamination (Oberts *et al.*, 2000). The presence of high sodium and chloride levels and acidic pH conditions in the urban snow pack from the accumulation of road de-icing salt may shift the speciation of metals in runoff water from particulate to more mobile and toxic soluble phases. Acidic early melt waters leaving the snow pack can be toxic enough to stress or kill aquatic life in receiving waterbodies (Novotny *et al.*, 1999). After the snow pack is melted medium and coarse-grained particulates often remain on urban surfaces. Thereby, a second shock to receiving waters can occur when the accumulated particles and associated contaminants left behind are washed off during the first rain events of the year.

It is well recognized that the use of chloride salts as road de-icing chemicals is an environmental concern for a host of reasons including damage to vegetation, weakening of concrete structures, corrosion of vehicles, contamination of shallow groundwater and impacts on freshwater ecosystems (*e.g.*, Howard and Beck, 1993; EC & HC, 2001; Marsalek, 2003). Infiltration of de-icing salt constituents has also been observed to increase the mobility of metals in roadside soils (Amrhein *et al.*, 1992; Norrström and Bergstedt, 2001) and increase metal concentrations in underlying aquifers, (Granato *et al.*, 1995;

Norrström, 2005) although contamination of groundwater by metals is a relatively rare occurrence that appears to be limited to older, high traffic areas.

Recommendations for modified designs of stormwater management practices to address such factors related to the hydrology and quality of snowmelt and winter runoff have been provided by Caraco and Claytor (1997) and are reflected in many stormwater management design guideline documents (*e.g.*, MPCA, 2008). Generally stormwater infiltration practices are recommended for managing snowmelt and winter runoff from source areas that are not considered pollution hot spots[1] and where de-icing salts are not highly concentrated (*e.g.*, snow dumps). Stormwater infiltration practices utilize the physical, chemical and biological process occurring in soils to attenuate contaminants dissolved in or carried by urban runoff before they reach aquifers of surface waterbodies.

Stormwater controls that utilize filtration and infiltration treatment processes can be highly effective at reducing concentrations of fine suspended sediment in runoff (Li and Davis, 2008a; Roseen *et al.,* 2009). Small particles (*e.g*., clay and silt) have higher surface-to-volume ratios than larger particles and, consequently, adsorb contaminants more readily. This is particularly true of clays, which have crystalline structures characterized by plates or flakes with external and internal surfaces available for adsorption (Brady, 1984). Therefore, by removing the fine suspended sediment in runoff, filtration and infiltration practices can also be highly effective at removing contaminants that tend to be adsorbed to particles.

One of the main considerations in evaluating the suitability of stormwater infiltration practices relates to the risk of contaminating groundwater resources, particularly where they would impair human use of the affected groundwater and surface water resources and aquatic ecosystems that receive groundwater inputs. Roof runoff typically generates low contaminant loads that mainly originate from atmospheric deposition and leaching from, or decomposition of construction materials. In contrast, runoff from roads, parking areas and vehicle service areas generate relatively high contaminant loads and contain a wider range of constituents (Table 4.1). Therefore, runoff from these sources poses a greater risk of groundwater contamination if infiltration practices are applied. Serious concern remains regarding the suitability of infiltration practices for treating runoff from high traffic areas subject to de-icing salt applications during winter, and in pollution hot spots that can generally be defined as areas where land use or activities have the potential to generate highly contaminated runoff (MDE, 2000).

[1] Pollution hot spots are defined here as stormwater source areas where certain land uses or activities have the potential to generate highly contaminated runoff (e.g., vehicle fuelling, servicing and demolition areas, outdoor storage and handling areas for hazardous materials, high traffic roads and parking lots, some heavy industry sites).

Table 4.1: Highway Runoff Constituents and Their Primary Sources*

Constituents	Primary Sources
Particulate	Pavement wear, vehicles, atmosphere, road maintenance (sanding in winter), tire wear, tire tread deposits
Nitrogen, phosphorus	Atmosphere, roadside fertilizer application
Lead	Tire wear (lead oxide filler material), lubricating oil and grease, bearing wear, metal deterioration
Zinc	Tire wear (filler materials), motor oil (stabilizing additive), grease, metal deterioration
Iron	Auto body rust, steel highway structures (guard rails, etc.), moving engine parts, metal deterioration
Copper	Metal plating, bearing and bushing wear, moving engine parts, brake lining wear, fungicides and insecticides, metal deterioration
Cadmium	Tire wear (filler material), insecticides, metal deterioration
Chromium	Metal plating, moving engine parts, break lining wear, metal deterioration
Nickel	Diesel fuel and gasoline exhaust, lubricating oil, metal plating, bushing wear, brake lining wear, asphalt paving, metal deterioration
Manganese	Moving engine parts
Cyanide	Anti-caking compound (ferric ferrocyanide, sodium ferrocyanide, yellow prussiate of soda) used to keep de-icing salt granular
Sodium, calcium, chloride	De-icing salts
Sulfate	Roadway beds, fuel, de-icing salts
Petroleum, oil and grease	Spills, leaks or blow-by of motor lubricants, antifreeze and hydraulic fluids, asphalt surface leachate, fuel and oil spills and leaks
Polycyclic aromatic hydrocarbon (PAH)	Asphalt, fuel and oil spills and leaks

*Adapted from Burton and Pitt, 2002.

The following sections focus on typical urban runoff contaminants, and the physical, chemical and biological processes by which these may be trapped, transformed or mobilized as water percolates through soil. While particulate matter (any suspended solid material including trash, soil, stone, asphalt, metal, glass, rubber, plastic and organic litter) is the most common type of urban runoff contaminant, during infiltration almost all suspended solids are removed by filtration or adsorption in the initial few centimetres of soil, and pose no risk to groundwater quality.

Based on reviews by Pitt *et al.* (1996 and 1999), the general categories of contaminants typically found in urban stormwater that may affect groundwater quality include metals, nutrients, pathogens, dissolved minerals, pesticides and other organic compounds. Metals and petroleum hydrocarbons are generated primarily from roads whereas nutrients, pathogens and pesticides mostly originate from pervious areas (*e.g.*, farms, gardens, landscaped areas) and overflows from combined sewers. The following section provides a brief review of these general contaminant categories.

4.1 Metals

Metals in runoff typically originate from automobiles and the decomposition of construction materials and pavement (Table 4.1). Clark *et al.* (2002) identified roofs as significant contributors of copper, zinc, lead, and cadmium to urban runoff and treated wood as a contributor of copper. In groundwater, metals can cause human health problems and impair aquatic ecosystems at high levels and cause aesthetic problems for use as drinking water (*e.g.*, taste) at lower levels. Metals that are typically observed at elevated levels in urban runoff include aluminum, cadmium, chromium, copper, iron, lead, nickel and zinc.

The solubility of most metals in water is pH dependant, with increasing solubility as pH decreases. The majority of these compounds, with the exception of zinc, are mostly found associated with the particulate solids in stormwater and are thus, easily removed through sedimentation and filtration (*e.g.*, Ku and Simmons, 1986). Although zinc is a necessary element for life, when present in excessive amounts, it affects the taste of drinking water. Dissolved forms of chromium, copper and zinc are rarely observed in urban runoff at concentrations exceeding Ontario drinking water guidelines (CRSDCWP, 2008; OMOE, 2003b). Thus, these pollutants do not typically present a concern with regard to human uses of groundwater.

Dissolved metal ions can be removed from infiltrating runoff through adsorption onto soil particles, cation exchange reactions, organic complexation and reaction with other dissolved constituents to form precipitates (Crites, 1985). In addition, uptake by plants and soil organisms also contributes to the retention of dissolved metals to a lesser extent. Adsorption to natural organic matter and soil particles is the dominant process by which dissolved metal ions are removed from runoff. Immobilization reactions (*i.e.*, cationic exchange, precipitation) also occur and are more pronounced at higher pH and in aerobic soil conditions. Organic complexation of a metal ion may enhance the metal's ability to move through the vadose zone as organic complexes are often stable and uncharged or negatively charged and not attracted to negatively charged adsorption or ion exchange sites (Wilde, 1994).

In cold climates, the application of de-icing salts (most commonly, sodium chloride) for winter road maintenance can affect the adsorption of dissolved metal ions in the soil (Bäckström *et al.*, 2004). Sodium ions can exchange with metals already associated with adsorption sites on soil particles, releasing them back into solution (Bauske and Goetz, 1993; Löfgren, 2001). If conditions are favorable, chloride and metal ions can also form water soluble complexes thereby reducing the metal retention capacity of the soil (Lumsdon *et al.*, 1995). Furthermore, the presence of high sodium ion concentrations in the soil followed by large volumes of low electrolyte infiltrating runoff water could promote the dispersion and mobilization of colloids and subsequent transport of metals associated with the colloids through the vadose zone, particularly lead in coarse-textured soil (Amrhein *et al.*, 1992; Amrhein *et al.*, 1994; Norrström and Bergstedt, 2001; Norrström, 2005).

4.2 Nutrients

Nutrients are compounds or constituents that contain nitrogen, phosphorus and other elements that are essential for plant growth. In urban areas, nutrients can originate from many different sources including decomposition of natural organic matter, animal waste, combined sewer overflows, septic system leachate, detergents, fossil fuel combustion, and fertilizers used for landscaping. Nitrogen and phosphorus are cyclic elements in that their chemical forms may be changed by decomposition or metabolic activity of bacteria, plants and animals. Excessive concentrations of dissolved nutrients can lead to high biological and chemical oxygen demand and result in eutrophication in aquatic ecosystems. Dissolved phosphorus exists in the environment in the form of orthophosphate ions. Dissolved nitrogen can exist as nitrate, nitrite and ammonium ions or as urea. Phosphorus is the limiting nutrient for plant growth in many aquatic ecosystems and elevated levels of dissolved phosphorus (*i.e.*, orthophosphate) is a typical cause of eutrophication. Consuming water containing excessive nitrogen is a direct human health hazard. Nitrogen is a commonly encountered contaminant in groundwater while phosphorus is not. During infiltration, some nutrients are removed from runoff and concentrations are significantly reduced. However, nutrient rich soils may also contribute nutrients to infiltrating water through desportion or ion exchange.

Most phosphorus in urban runoff is associated with the surfaces of suspended particles and is readily filtered or adsorbed along with the particles when infiltrated into soil. Dissolved phosphorus in the form of orthophosphate may be taken up by plants, directly precipitated or chemically adsorbed onto soil particles through reactions with iron, aluminum or calcium (Crites, 1985). In cold climate regions, orthophosphate removed from urban runoff by plants may be seasonally released back to the soil through decomposition of foliage, particularly during fall and winter seasons. The orthophosphate is typically immobilized through adsorption to soil particles and remains available for plant growth in subsequent spring and summer seasons.

Dissolved nitrogen can be removed from runoff through nitrogen fixing organisms and plant uptake in aerobic conditions, and denitrifying organisms in anaerobic conditions through reduction to ammonia gas. Nitrogen in the form of nitrate is highly soluble and will stay in solution in percolating runoff water after leaving the root zone and reach the water table. In cold climates, dissolved nitrogen removed from urban runoff by plants may also be seasonally released back to the soil through decomposition of foliage. The dissolved nitrogen may be immobilized through interactions with soil particles, mineralized into ammonium, metabolized by nitrifying or denitrifying bacteria or transported to aquifers and surface water through the movement of groundwater.

4.3 Pathogens

Microorganism pathogens in stormwater that can pose a threat to human health include viruses, protozoa and bacteria. Pathogen sources that potentially contaminate groundwater include waste decomposition, wildlife, livestock and pet droppings, combined sewer overflows and sanitary sewage spills. Factors that affect the survival of viruses and enteric bacteria 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. Filtration and adsorption are the dominant processes by which pathogens are removed from infiltrating runoff. Fine textured soils retain pathogens more effectively than coarse-textured soils, with greater removal rates possible as clay content increases. Viral adsorption to soil particles does not necessarily result in virus inactivation and can be reversed with a change in the ionic environment (Jansons *et al.*, 1989). Bacteria and viruses survive longer in the soil at low temperatures (Crites, 1985).

4.4 Dissolved minerals

The dissolved minerals of concern in urban runoff from a groundwater contamination perspective are salts. These salts include compounds containing combinations of sodium, calcium, potassium, magnesium, chloride, fluoride, sulfate or bicarbonate. Application of de-icing salt for winter road maintenance is a common practice in many cold climate areas. Most commonly, sodium chloride is used, but other salts such as calcium chloride or magnesium chloride may also be applied. High salt concentrations in groundwater are undesirable because of possible physiological effects, mineral taste and corrosion. High concentrations of chloride ions in water affects taste, accelerates corrosion of pipes and household appliances and can be toxic to freshwater aquatic organisms. A comprehensive scientific assessment by Environment Canada and Health Canada determined that in sufficient concentrations, road salts containing inorganic chloride salts pose a risk to plants, animals and the aquatic environment and are 'toxic' as defined in Section 64 of the *Canadian Environmental Protection Act, 1999* (EC&HC, 2001).

Excess sodium can cause health problems, especially for people on sodium-restricted diets. Sodium can also adversely affect some types of vegetation and crops, causing leaf burn. Sodium chloride is readily soluble in water with chloride ions being extremely mobile in the environment as they do not react with other chemicals nor do they adsorb significantly on mineral surfaces when infiltrated into soil. Dissolved sodium ions may replace calcium and magnesium ions in soil minerals, altering soil structure, hydraulic properties and fertility (Krauskopt, 1995), and can cause trace metals to be leached from the soil and into groundwater (Amrhein *et al.*, 1992; Amrhein *et al.*, 1994; Granato *et al.*, 1995; Norrström and Bergstedt, 2001; Norrström, 2005).

Soil is not effective at removing most salts and once contamination begins, the movement of salts into groundwater can be rapid (Pitt *et al.*, 1996). Several studies in Ontario have established links between de-icing salt use and groundwater contamination (*e.g.*, Howard and Beck, 1993; Labadia and Buttle, 1996; Williams *et al.*, 2000).

4.5 Pesticides

Pesticides encompass a wide variety of synthetic chemical compounds that can be generally classified into one of the following three groups depending on their targets: herbicides, fungicides or insecticides. Pesticides are used in urban areas, primarily for weed and insect control in landscaped areas, along roadsides, in parks and on golf courses. Pesticides have been linked to cancer, nervous system disorders, birth defects, and other systemic disorders through toxicological testing. Some commonly used pesticides in urban areas that have been rated with regard to their potential for groundwater contamination when infiltrated into the soil are diazinon, malathion, 2,4-D, lindane, atrazine and chlordane (Pitt *et al.*, 1996). The sale or use of diazinon, malathion and certain forms of 2,4-D is restricted by Ontario's Cosmetic Pesticide Ban (O.Reg. 63/09) and lindane, atrazine and chlordane are not classified for sale or use in Ontario (OMOE, 2009).

The dominant processes by which pesticides are removed from runoff infiltrating through soil are volatilization, adsorption onto soil particles and decomposition. Volatilization losses of soil applied pesticides can be a significant removal mechanism for some pesticides with high volatility, but negligible for low volatility compounds. Decomposition of pesticides in the soil depends upon many factors including pH, temperature, light, humidity, air movement, soil type, persistence (*i.e.*, half-life), and microbiological activity (Ku and Simmons, 1986). Decomposition half-lives of many pesticides have been determined but generally apply to surface soils and do not account for reduced microbial activity found deep in the vadose zone (Bouwer, 1987). Pesticide mobility in soil depends on patterns of use, solubility in water, persistence of the pesticide compound, and texture and organic carbon content of the soil. Leaching is enhanced in alluvial soils with the greatest mobility occurring in coarse-grained or sandy soils having low clay and organic matter content and high permeability (Domagalski and Dubrovsky, 1992). Fungicides must be mobile in the soil in order to reach targeted pests and generally have the highest potential to leach into groundwater (Pitt *et al.*, 1996).

4.6 Other organic compounds

Organic compounds are comprised mainly of carbon, hydrogen, oxygen and nitrogen and encompass a wide range of naturally occurring and synthetic compounds. The compounds of most concern for groundwater contamination are typically man-made (synthetic). The types and concentration of organic compounds in urban runoff are related to land use, geographic location, and vehicular traffic volume. The major sources of organic contaminants in urban areas are from the use or spilling of petroleum products such as lubrication oils and grease, fuels, solvents and combustion emissions but can also come from landfills and sanitary sewer leaks or overflows. Many synthetic organic compounds are considered hazardous to the heath of humans and other organisms.

Most organic compounds are reduced in concentration during infiltration through the soil, although they may still be detectable in groundwater. Removal of organics from the soil and infiltrating water can occur through volatilization, adsorption, and degradation (Crites, 1985). Adsorption is not always a permanent removal mechanism. Degradation of organic contaminants mostly occurs through microorganisms. Like pesticides, organic compound mobility in soil depends on solubility in water, persistence of the compound, and texture and organic content of the soil. Leaching of organic compounds into groundwater occurs more readily in coarse textured soils such as sand and gravel. Removal of organics from infiltrating water by soil increases with increasing clay and organic matter content. The degree of removal of nonhalogenated organic compounds is greater than that of halogenated organics (Bouwer *et al.* 1984).

5.0 RISK OF GROUNDWATER CONTAMINATION

Application of stormwater infiltration practices in cold climate jurisdictions raises concerns regarding the risk of soil and groundwater pollution (Mikkelsen *et al.*, 1994). These concerns are particularly relevant when considering the types of contaminants typically present in urban runoff (*e.g.*, road de-icing salt constituents), limitations to the effectiveness of removal processes in soil, and complications associated with the hydrology and quality of snowmelt noted previously. Pitt *et al.* (1999) addressed potential groundwater contamination problems associated with stormwater infiltration in a summary of findings from a multi-year research project sponsored by the US EPA. This potential was evaluated based on the influencing factors: typical abundance of pollutants in runoff; pollutant mobility in soil; treatability of the pollutants; and infiltration practice applied. Contamination potential was defined as the most critical rating of the influencing factors. It was noted that their evaluation approach is only appropriate for initial estimates of contamination potential because of the simplifying assumptions made (*e.g.*, assumes infiltration in sandy soil with low organic content as a worst case for the pollutant mobility factor).

Pollutants with the greatest potential for adverse impacts on groundwater include:

- nitrate (although urban runoff concentrations are typically low);
- pesticides (lindane and chlordane in particular, which are not classified for sale or use in Ontario) if no pretreatment is provided;
- certain types of polycyclic aromatic hydrocarbon (PAH) and halogenated hydrocarbons if no sedimentation pretreatment is provided;
- pathogens if no disinfection pretreatment is provided;
- metals (zinc, chromium, nickel and lead in particular) if no sedimentation pretreatment is provided; and,
- chloride irrespective of pretreatment.

As noted previously, dissolved forms of zinc and chromium are rarely observed in urban runoff at concentrations exceeding Ontario drinking water guidelines and are typically not a concern with regard to groundwater contamination.

Recommendations for the control of these pollutants include (Pitt *et al.*, 1999):

- pollution prevention, including reduced use of galvanized metals, pesticides and fertilizers;
- diversion of runoff from manufacturing and industrial areas from infiltration facilities because these areas can be sources of relatively high concentrations of soluble toxicants;
- diversion of flows from combined sewer overflows from infiltration facilities because of sanitary sewage contamination;
- diversion of winter snowmelt and early spring runoff containing high contaminant concentrations (particularly de-icing salt constituents) from infiltration facilities;
- pretreatment of all other runoff using sedimentation processes before infiltration to minimize groundwater contamination and prolong the life of the infiltration facility;
- use of surface infiltration practices (*e.g.*, vegetated filter strips, bioretention, grassed swales) instead of subsurface infiltration practices (*e.g.*, soakaways, infiltration trenches and chambers, perforated

pipes and catchbasins), unless runoff is known to be relatively free of pollutants, because surface practices are able to take greater advantage of natural soil pollutant removal processes;
- prioritize infiltration of runoff from source areas that are comparably less polluted, such as residential areas and roofs.

It should be noted that the recommendation about diversion of runoff from road or parking lot runoff to detention facilities during certain winter and early spring periods may not be feasible in certain contexts, particularly when the infiltration facilities are on private property.

In a review of road de-icing salt issues in urban stormwater management, Marsalek agreed that risk of groundwater contamination from infiltration practices would have to be mitigated by focusing on infiltration of runoff relatively free of de-icing salt constituents (*e.g.*, roof runoff). He also recommends controlling the use of de-icing salt in vulnerable areas and restricting the application of infiltration practices to areas with limited uses of the groundwater (Marsalek, 2003). Identifying and testing alternative de-icing compounds that are less toxic, less corrosive and less mobile in the environment than the salts in wide use today (*e.g.,* TRB, 1991; Hellstén *et al*., 2005; Shi *et al*., 2009), and employing road maintenance practices that reduce the quantity of de-icing compound used (*e.g.*, pre-wetting, advanced road weather information systems), are other management approaches that are receiving considerable attention (TAC, 2003).

A three step methodology to evaluate infiltration as a management option has been developed that utilizes locally derived data and the available body of research on stormwater quality and soils (Clark and Pitt, 2007). The first step involves evaluating pollutant loadings and chemical forms. Identifying the native soil characteristics that affect pollutant migration at the particular site is the second step. The third step uses information from the previous two steps to predict the potential for groundwater contamination. This can be accomplished through a simplified method based on worst-case assumptions or application of a computer model that uses actual or estimated field data to predict the depth of migration in a pre-specified time period (Clark and Pitt, 2007).

6.0 PERFORMANCE MONITORING STUDIES

The following sections summarize the findings from peer reviewed journal articles, books and recently published studies of the application of stormwater infiltration by the general type of best management practice applied. Available information on performance is summarized with regard to runoff reduction (*i.e.*, hydrologic benefits), surface water quality (*i.e.*, effects on water quality in overflow or underdrain flows), groundwater quality (*i.e.*, potential for groundwater contamination), and soil quality (*i.e.*, accumulation of contaminants). Implications on suitability, siting, design and maintenance criteria are also discussed.

6.1 Lot level practices

As noted previously, lot level infiltration practices can be grouped into the general BMP types of permeable pavement, vegetated filter strips, bioretention, soakaways, and infiltration trenches and chambers.

6.1.1 Permeable pavement

A variety of types of permeable pavement systems exist featuring several design variations. A thorough review of the various types of pavement systems and their specifications is available in a book devoted to the subject (Ferguson, 2005).

Runoff reduction
There are several studies that have demonstrated the effectiveness of permeable pavement in reducing runoff. In France, a typical street section was replaced with permeable asphalt over a 55 centimetre (cm) thick crushed stone reservoir, where it was reported to infiltrate an average of 97% of stormwater runoff (Legret and Colandini, 1999). Similarly, Booth and Leavitt (1999) reported virtually no surface runoff during the autumn and winter from planted and unplanted concrete block permeable pavement installed in an institutional parking lot in Renton, Washington. A repeat study conducted at the same site four years later revealed similar results. Among the 15 storms monitored in the second study, only one 44 mm rain event generated runoff, representing a mere 3% of the total precipitation (Brattebo and Booth, 2003). Extensive research on permeable interlocking concrete pavers (PICP) at the University of Guelph, Ontario, showed 90% reduction in surface runoff volume compared to traditional impervious pavements (James, 2002).

In Luleå, Sweden, Bäckström (1999) examined runoff from two different residential road sections, one permeable and one impermeable, and found that snowmelt runoff volume could be reduced by 50 to 60% by using permeable pavement. A PICP system installed on sandy loam soil was monitored in Connecticut, where a 72% reduction in runoff was observed, compared to a nearby impermeable asphalt system, over a 22 month study (Gilbert and Clausen, 2006). In North Carolina, Collins *et al.* (2008) reported similar results in a comparison of the hydrologic performance of four permeable pavements relative to an impermeable asphalt control. The plots in this study consisted of two types of interlocking concrete pavers, concrete grid pavers and pervious concrete. All were underlain by a 5 to 10 cm bedding layer and a 23 to 25 cm crushed stone base layer with perforated underdrains installed at the bottom of each base layer. Mean runoff reductions from rainfall depth ranged from 98.2% to 99.9% for the permeable pavements examined, compared to 34.7% for impermeable asphalt. Although differences in performance were subtle, the concrete grid paver installation yielded the greatest volume of surface runoff, likely due to the lower hydraulic conductivity of the sand fill media. Mean peak flow reductions ranged from 60.3% to 77%, with concrete grid pavers providing the greatest reduction. During large storm events (>50 mm) all pavers performed similarly. Based on this study, it was recommended that various permeable pavement types be treated similarly by regulators with respect to runoff reduction (Collins *et al.*, 2008).

There is a paucity of studies on permeable pavements installed over fine textured soils, as many stormwater BMP manuals do not consider them suitable for these soil types. In King City, Ontario, an 18 month study of a PICP system with a 60 cm granular reservoir over a clay loam soil showed almost no surface runoff but large rain events (greater than 20 mm) often required more than 72 hours to drain (TRCA, 2008). In Georgia, Dreelin *et al.* (2006) tested the effectiveness of a grassed, plastic grid pavement with a sand bedding layer, 25 cm gravel base and perforated underdrain constructed over soils with a clay content of 35 to 60%. Although the native soils were clay based, infiltration rates were very good, ranging from 48 to 167 mm/hr. During nine rain events between 0.3 and 18.5 mm, total runoff from the grid pavers was 93% less than from a nearby conventional asphalt pavement, suggesting permeable pavements can be applied effectively on clay soils for the control of small storm events and the retention of the “first flush” during larger storms (Dreelin *et al.*, 2006). A large pervious concrete plaza with underlying stone reservoir infiltration beds were installed at Villanova University in Pennsylvania that captures runoff from roofs, walkways and grassed areas. Kwiatkowski *et al.* (2007) report that all runoff from storms of 50 mm or less have been successfully captured and infiltrated by the pervious concrete infiltration system, which in an average year, represents 90% of all rain events in Pennsylvania. The infiltration rate of the silty sand native soil underlying the infiltration beds is, on average 3.6 mm/hr, which is lower than what many cold climate jurisdictions recommended for this type of BMP (*e.g.*, OMOE, 2003a; NYDEC, 2003). These results indicate that permeable pavement BMPs can be designed to function as infiltration systems, even on low permeability soils.

Runoff volumes are reduced in permeable pavement systems even when runoff is prevented from infiltrating into the native soil by an impermeable membrane, as is common in areas where the soils below the base course layer have low permeability. In the UK, Anderson *et al.* (1999) examined rainfall, runoff and evaporation from permeable pavers with different bedding materials (25 to 50 mm deep) on a full scale model parking lot underlain by a drainage collection system. The researchers found that for a one hour duration 15 mm simulated rainfall event, an average of 55% and 30% of rainfall was retained when the structure was initially air dried and wet, respectively. Daily evaporation rates from a fully drained structure averaged approximately 20% of that from an open water evaporation pan, with fine bedding materials producing the highest rates of evaporation. Higher evaporation rates would be expected under field conditions where water is stored in the base course layer for 24 to 48 hours after a rain event.

Allowing rainfall to infiltrate into the native soil rather than runoff over the surface helps to recharge groundwater. Depending upon site design and soil type, permeable pavement may allow as much as 70-80% of rainfall to recharge groundwater (Gburek and Urban, 1980). In theory, the increase in recharge should enhance groundwater discharge to streams. However, this effect has not been quantified because most field studies of permeable pavement are conducted on relatively small areas and seepage rates into streams are notoriously difficult to estimate with a high degree of accuracy.

During the winter, water will continue to infiltrate as temperatures permit, but at a slower rate. In a laboratory investigation of porous asphalt in Sweden, Bäckström and Bergstrom (2000) reported a 50% reduction in surface infiltration rates as temperatures declined from 20˚C to 0˚C. When the pavement was subjected to alternate freezing and melting over two days, the infiltration rate fell to 90% of the rate observed at 20˚C. Even at this rate, however, the pavement still infiltrated at a rate of between 60 and 300 mm/hr, which is similar to that of a relatively well drained agricultural soil. Favorable performance of a permeable interlocking concrete paver system during winter has also been observed in an installation in King City, Ontario. Even with above ground air temperatures as low as -25˚C, the stone reservoir continued to function as an effective storage unit (TRCA, 2008). Because the permeable pavement was able to infiltrate snowmelt, ponding of melt water and subsequent ice build-up upon the return of freezing temperatures, was reduced.

Bäckström (2000) also monitored winter temperatures of a porous asphalt and conventional asphalt in a residential area of Luleå, Sweden. The porous asphalt base (1.6 to 8 cm) was drained with a pervious pipe and was installed on silty moraine soils with high clay content. The two pavements were found to freeze in much the same way. However, mid-winter temperatures of the stone reservoir beneath the conventional asphalt were lower and the pavement thawed in the spring 3 to 4 weeks after the porous asphalt. The maximum depth of frost penetration tended to be slightly greater beneath the conventional asphalt. The shallower frost penetration beneath the porous asphalt was attributed to the heat insulating effect of air in the porous pavement and moisture in the base course, which increases the latent heat available. The porous asphalt surface thawed earlier because infiltration of melting snow and ice helped to warm the underlying stone reservoir. More rapid infiltration of melt water has the additional benefit of reducing the potential for slip hazards as there is less water on the surface that can freeze during cold nights. He concluded from these data that porous pavement is more resistant to freezing and has a lower risk of frost heave damage relative to conventional impervious pavements (Bäckström, 2000).

The pavement openings will clog over time as dust and dirt accumulate in the pavement openings and pore spaces of the underlying granular media. Rain and traffic further exacerbate the problem by breaking up soil aggregates into finer particles that block the pores and allow for further accumulation of fines. Eventually a hard crust forms, creating a seal that can drastically reduce infiltration through the surface openings (Balades *et al.*, 1995, Pratt *et al.*, 1995). As noted earlier, clogging has been a serious issue in some of the early permeable pavement installations (Lindsey *et al.*, 1992). Many of the early permeable pavement installations were constructed with sand as a bedding layer. Further, garden and grassed areas around the perimeter often drained onto the pavement, rather than away from it. These conditions tend to increase the potential for clogging. More recent installations use washed stone in the pavement openings and bedding layer because it resists breaking down into smaller particles with age, and the pore spaces are large enough to transmit fine particulate matter into the base course layers, thereby reducing the potential for surface sealing. At University of Guelph experimental plots, Gerrits (2001) reported considerably better infiltration on 8 year old permeable pavers constructed with a bedding layer of 7.5 cm of clear washed stone than those with a 10 cm mixture of clear washed stone and sand (both installations used 40 cm of granular ‘A’ as the sub-base). The pure washed stone bedding layer installation also responded much more effectively to maintenance efforts directed at restoring the original surface infiltration capacity.

In a study of long term performance of permeable concrete grid paver systems in Germany, Borgwardt (2006) hypothesizes that over the 10 to 20 year service life of the pavement, that infiltration rates can be expected to decrease to 10 to 25% of the original rates, due to accumulation of fines in the upper 20 mm of joint fillings. A correlation between infiltration performance and grain size of the aggregate material used in joint fillings was also observed, regardless of the age of the pavement, indicating that coarser materials consistently exhibit higher infiltration rates (Borgwardt, 2006). Bean *et al.* (2007b) examined surface infiltration rates on concrete grid (n = 16) and permeable interlocking concrete pavers (n = 11) located in North Carolina, Maryland, Virginia and Delaware. The pavements ranged in age from six months to 20 years. Infiltration tests conducted on the original condition of the pavement and after removal of 13 to 18 mm of surface residue indicated a 60% increase in infiltration rates. Although the infiltration capacity of permeable pavements may decrease as fine particles are loaded onto the surface, testing indicated that partially clogged pavements can still infiltrate large quantities of water, comparable to grassed sandy loam (Bean *et al.*, 2007b).

Surface water quality

Permeable pavements improve runoff water quality by filtering and trapping contaminants within pavement pores and the underlying stone reservoir or base course. A study conducted at the University of Guelph in Ontario reported improvements to water quality after infiltration of stormwater runoff through permeable pavers and a shallow base course, especially for zinc and iron (Shahin, 1994). The pollutant removal capacity of a permeable asphalt roadway in France was examined by comparing the quality of runoff at the reservoir structure outlets with a nearby impervious roadway (Legret and Collandini, 1999). After seven years of monitoring it was found that metallic pollutants were mainly retained within the permeable asphalt structure with minimal contaminants entering the underlying soil. Mean event pollutant load reductions of 59% for total suspended solids (TSS), 84% for lead, 73% for zinc and 77% for cadmium were observed. In a similar study in Nottingham, England, a parking lot surfaced with concrete grid permeable pavers filled with aggregate successfully trapped most suspended solids and metals (Pratt *et al.*, 1995).

Monitoring of a section of French highway before and after resurfacing the impermeable asphalt with a 30 mm thick layer of permeable asphalt indicated that runoff water quality is improved upon infiltration through the pavement. Heavy metal loads discharged into the environment were reduced from 20% for copper, up to 74% for lead (Pagotto *et al.*, 2000). Moreover, suspended solids loads were reduced by 87% and hydrocarbons by 90%. These high load reductions were believed to be achieved by retention of fine particulate pollutants in the pavement through filtration (Pagotto *et al.*, 2000).

In a study of a permeable interlocking concrete paver system in Connecticut, concentrations of all pollutants measured (suspended solids, nitrate, ammonia, TKN, total phosphorus, lead, zinc and copper) were significantly lower in permeable pavement runoff than in runoff from nearby impermeable asphalt driveways (Gilbert and Clausen, 2006). Sampling of water percolating through a porous asphalt pavement installation on the University of Rhode Island campus in Kingston also showed good removal of PAH within the base course, with observed concentrations near the detection limit (Boving *et al.*, 2008). Concentrations of zinc and copper in the base course remained below recommended drinking water limits

(5 mg/L for zinc; 1.3 mg/L for copper), with peak concentrations occurring during late winter and early spring.

In North Carolina, Bean *et al.* (2007a) compared the quality of water that had filtered through permeable interlocking concrete pavement and 275 mm gravel base with conventional asphalt runoff. They reported significantly lower concentrations of zinc, total phosphorus, ammonia and TKN in infiltrate at the base, but no significant differences in total nitrogen, nitrates, dissolved phosphorus, TSS and copper. Nitrate-nitrogen was the only contaminant for which higher concentrations were observed in infiltrate than in runoff from the conventional asphalt, likely due to nitrification of ammonia in the aerobic conditions present in the permeable pavement system. It was suggested that the use of permeable pavement in series with a secondary treatment down gradient, such as a vegetated buffer strip, would help to reduce nitrate-nitrogen loadings before reaching receiving water bodies (Bean *et al.*, 2007a).

Observations of good surface water quality performance of permeable pavement systems are supported by recent work by Scholes *et al.* (2008). Based on a methodology utilizing fundamental scientific principles, theoretical data and a risk-rating approach to rank stormwater BMPs according to pollutant removal potential, they predict such systems to perform equally well, if not better than constructed wetland ponds for removal of suspended solids, dissolved phosphorus and faecal coliform bacteria from stormwater (Scholes *et al.*, 2008).

Groundwater quality

As noted previously, typical contaminants that pose the greatest risk of contaminating groundwater through the practice of stormwater infiltration include some metals, nitrate, a few pesticides, some polycyclic aromatic hydrocarbons (PAHs), enteroviruses, and salts such as chloride (Pitt *et al.*, 1996). Paved areas are typically not significant sources of nitrogen, pesticides or enteroviruses. Due to deposition from vehicles and spreading of de-icing salt during winter, paved areas in cold climates can be significant sources of metals, PAHs and chloride.

Oils and hydrocarbons are relatively insoluble in water and tend to be adsorbed readily by soil particles and granular media. A growing body of research has demonstrated that naturally occurring microbial communities on pavement building materials help to retain and degrade hydrocarbons within the base course layer, even in cold climates (*e.g.* Newman *et al.*, 2006b). It is suggested that the key to successful biodegradation of hydrocarbons in permeable paving systems is the geotextile liner below the base course layer, where the physical and chemical properties of the geotextile reduce the velocity of water flow, immobilize the contaminants and provide an appropriate habitat for the growth of a microbial biofilm that degrades the hydrocarbons (Newman *et al.*, 2006a).

As with all stormwater infiltration practices, risk of groundwater contamination from infiltration of snowmelt or stormwater runoff that is laden with dissolved salt (*e.g.*, sodium, calcium or magnesium chloride) is a significant concern because chloride ions are extremely mobile in the soil and are transported by percolating water to underlying aquifers, unattenuated. In both Washington and California, observations have confirmed that dissolved chloride and nitrate ions can percolate through the pavement and underlying native soil and into the groundwater but most other contaminants are adsorbed within the upper 10 centimetres of native soil (Brattebo and Booth, 2003; Nightingale, 1987a).

Positive results were found for a study that examined four types of permeable pavement systems in an institutional parking area in Renton, Washington (Booth and Leavitt, 1999; Brattebo and Booth, 2003). The lot covered deep, well drained sandy soil. Initial results showed undetectable levels of diesel fuel and motor oil and low levels of lead, zinc and copper in the infiltrate from all permeable paving systems at 10 centimetres soil depth. Results following an additional five years of operation showed the permeable paving systems and underlying soils were still successfully filtering contaminants from stormwater. The researchers reported that 88 and 100% of asphalt runoff samples exceeded Washington receiving water standards for zinc and copper, respectively. By contrast, only 6 and 17% of permeable block pavement infiltrate samples exceeded the standards for copper and zinc, respectively (Brattebo and Booth, 2003). It is notable that the parking area was subject to very little application of de-icing salt, which may otherwise have affected the mobility of some metals (*e.g.*, lead, copper and cadmium) in the soil.

A pervious concrete infiltration system installed on the campus of Villanova University in Pennsylvania and subjected to routine applications of calcium chloride as a winter de-icing compound has been examined with regard to water quality performance (Kwiatkowski *et al.*, 2007). Dissolved copper was identified as a contaminant of concern in roof runoff at this site due to it being used as a roof and downspout building material. Based on approximately two years of monitoring the quality of groundwater below the permeable concrete infiltration bed it was concluded that the majority of copper in roof runoff was adsorbing to soil within the first 30 cm below the facility and that copper concentrations were below the Pennsylvania guideline for aquifer water quality of 1 mg/L. Elevated chloride concentrations were observed below the infiltration bed following de-icing salt spreading events with levels exceeding the USEPA secondary standard for drinking water of 250 mg/L (USEPA, 2009) on several occasions, while exceeding the acute toxicity threshold for aquatic ecosystem protection of 860 mg/L (USEPA, 1988) only once during the course of the study. Chloride levels in groundwater below the infiltration bed dropped quickly during the spring and eventually leveled out during the summer and fall months (Kwiatkowski *et al.,* 2007).

In King City, Ontario, a study of a permeable interlocking concrete paver system installed in a college parking lot examined the potential for infiltrated stormwater to contaminate groundwater (TRCA, 2008). The quality of runoff from impermeable asphalt was compared with water after infiltration through a 60 cm granular reservoir and one metre of soil below the permeable pavement. Relative to asphalt runoff the permeable pavement infiltrate was characterized by higher levels of pH, hardness (as calcium carbonate) and alkalinity. These properties help to buffer the effects of acid precipitation and reduce the aquatic toxicity of trace metals in surface water. Median concentrations of zinc, phosphorus, total suspended solids and oil and grease were significantly lower than those in asphalt runoff. PAHs were rarely detected, but concentrations were generally higher in runoff from the asphalt control. Chloride and sodium from spreading of de-icing salt during winter were the groundwater contaminants of greatest concern. Infiltrate concentrations of both constituents were frequently above the Ontario drinking water aesthetic objectives of 250 and 200 mg/L, respectively (OMOE, 2003b).

Soil quality

On porous asphalts, pollutants have been observed to accumulate mainly within the surface pores and, to a lesser extent, on the geotextile layer separating the base course layer from the underlying native soil (Legret *et al.*, 1996; Legret and Collandini, 1999). Copper, lead, zinc, and cadmium are retained near the surface in association with clogging particles (Legret *et al.*, 1999).

The simulation of heavy metal transfer into the soil below a porous asphalt pavement system was carried out using a mathematical model and predicted that increases in lead, copper and zinc content would be slight after 50 years of service and well below French regulation threshold values (Legret *et al.*, 1999). The model predicted migration of cadmium down to 30 cm depth, but that risk of groundwater contamination would be low (Legret *et al.*, 1999).

Gerrits (2001) collected and analyzed samples of material accumulated after eight years of service in the void spaces, bed and base of an interlocking concrete pavement system installed at the University of Guelph. Concentrations of heavy metals were found to be less than the Ontario Ministry of the Environment's Guideline Concentrations for Selected Metals in Soils (Gerrits, 2001).

Soil sampling of seven permeable interlocking concrete pavement installations in the Greater Toronto Area, ranging in age from 4 to 17 years, suggests that long term accumulation of contaminants in soils beneath the pavements was not a significant concern (TRCA, 2008). Contaminant levels at all sites were generally below Ontario soil background concentrations for non-agricultural land uses (OMOE, 2004). There were a few exceptions, but even in these cases, concentrations were still well below levels that would trigger the need for remediation or landfilling, even in comparison to proposed new Ontario site condition standards (Table 6.1).

6.1.2 Vegetated filter strips

Runoff reduction

Vegetated filter strips are typically used to filter and pretreat runoff as sheet flow, rather than infiltrate it. Hence, no studies have been found that report on their ability to reduce the rate and volume of runoff.

Surface water quality

Based on a synthesis of performance monitoring studies available as of 2000, it was reported that vegetated filter strip (*i.e.*, buffer strip) pollutant removal efficiencies can be expected to range from 20 to 80% for suspended solids, 20 to 60% for total nitrogen, 20 to 60% for total phosphorus and 20 to 80% for total heavy metals (ASCE, 2000). Because of the high variability of reported surface water quality performance, vegetated filter strips are generally considered by regulatory agencies as a pretreatment practice that should complement other water quality BMPs. However, Barrett *et al.* (1998) have noted that many of the studies in which low contaminant removal efficiencies have been observed were not well designed and that significant removal of pollutants had likely occurred before the runoff entered the test sections that were monitored.

In a study of eight vegetated filter strips (*i.e.*, buffer strips) receiving runoff from California highways, Barrett *et al.* (2004) found that they consistently reduced the concentration of suspended solids and total metals in road runoff. The strips were generally less effective at reducing concentrations of dissolved metals and essentially no changes in concentrations of nitrogen and phosphorus were observed. For stormwater constituents exhibiting decreases in concentration, steady state levels were generally achieved within 5 metres of the pavement edge when vegetation coverage exceeded 80%. Substantial reductions in contaminant loads were also observed for almost all constituents because of the large amount of infiltration that occurred at most sites (Barrett *et al.*, 2004).

Groundwater quality

In a study by Dierkes and Geiger (1999) of vegetated filter strips along high traffic highways (52,000 to 107,600 vehicles per day) in Germany, leaching to groundwater of contaminants accumulated in roadside soils was predicted to be limited, based on laboratory column tests, even for soils receiving highway runoff for greater than 20 years.

Soil quality

In a Swiss study, Mikkelsen *et al.* (1997) analyzed soil quality at various depths below a vegetated filter strip that had received runoff from a high traffic highway (average of 37,000 vehicles per day) for over thirty years. They found total concentrations of lead, copper, zinc and cadmium exceeded Swiss soil quality standards. However, none of the measured exchangeable metal concentrations exceeded threshold values for drinking water quality, suggesting that leaching of heavy metals into groundwater is likely limited (Mikkelsen *et al.*, 1997).

A German study investigating the impacts of runoff from a high traffic highway (52,000 to 107,600 vehicles per day) on roadside soils concluded that the age of roadside soils was positively correlated with the concentrations of several heavy metals and total polycyclic aromatic hydrocarbons (Dierkes and Geiger, 1999). The highest metals concentrations observed in the soils were for cadmium (up to 5.6 mg/kg), zinc (up to 1580 mg/kg) and lead (up to 290 mg/kg) in the first 5 cm depth at a 24 year old site (Table 6.1). Concentrations decreased rapidly with distance from the highway and with depth. Between 10 cm to 30 cm depth, in most cases, only 7% to 25% of metal concentrations found in the upper 5 cm of soil were recorded. The reduction was less for cadmium. The first two metres from the highway showed the highest metal concentrations. Within a distance of 10 m from the highway, concentrations of metals decreased to 7% for copper, approximately 30% for lead and zinc and 45% for cadmium. Concentrations of total PAH reached as high as 23 mg/kg in the upper 5 cm of soil at the 24 year old site. Soil characteristics such as organic content and pH were found to be important factors controlling the contaminant retention capacity of the soils (Dierkes and Geiger, 1999).

In residential areas the potential for soil contamination in roadside soils appears to be much lower than for highway roadside soils. Soil samples from two older residential areas in Toronto where roadside swales and ditches were preserved showed concentrations of copper, zinc, lead, cadmium, chromium and nickel below Ontario background soil concentrations (Table 6.1). PAHs were elevated above background levels between 14 and 21 cm below the surface, and generally higher than reference site samples, but overall concentrations were still below levels that would trigger the need for remediation (TRCA, 2008).

6.1.3 Bioretention

Runoff reduction

Concern over the winter performance of bioretention practices in cold climates has been very recently addressed by several researchers. In a study of a rain garden in Connecticut, sized to contain a 25 mm storm, a flow mass balance indicated that less than 1% of inflow water overflowed over the two year period of study, despite measurable frost being present in the bioretention media during winter months (Dietz and Clausen, 2006). Findings from studies of the performance of low impact development practices, including two types of bioretention systems at the University of New Hampshire indicate a high level of functionality during winter months and that frozen filter media has not been a concern (Roseen *et al.*, 2009). A study of a bioswale on a college campus in King City, Ontario also showed continuous infiltration throughout the winter, with soil temperatures at 50 cm below the soil surface never falling below freezing, despite air temperatures down to -25˚C (TRCA, 2008). Seasonal variation in infiltration rates through bioretention facilities have been observed, with reduced rates occurring in winter months, but differences between summer and winter are minimal (Emerson and Traver, 2008; Roseen *et al.,* 2009).

The hydrologic performance of four bioretention cells in Minnesota during cold climate conditions was examined by Davidson *et al.* (2008) over a three year period. The authors found that three of the four cells functioned for approximately 84% of the winter season. The fourth cell was constructed over poorly draining soils and did not function well even during warm weather. Soil temperature was found to be most highly correlated to hydrologic performance with infiltration ceasing at all cells on occasions when air temperatures were well below freezing. Recommendations for the design of bioretention cells to optimize performance in cold climates were made based on their observations, which include the use of engineered soils that are devoid of silt or clay particles, pool depths less than 1 foot deep that draw down to the frost line within 12 hours to minimize potential for freezing, and installation of an underdrain system with a valve at the outlet that permits operation of the cell as either an infiltration system or filtration system (Davidson *et al.*, 2008).

Table 6.1: Metal concentrations in soil below infiltration facilities over 10 years old in comparison to proposed Ontario standards

Study Reference	BMP Type and Location	Age of Facility (years)	Depth below base (cm)	Concentration (μg/g)					
				Cadmium	Chromium	Copper	Lead	Nickel	Zinc
TRCA, 2008	Permeable pavement, Humberwood Centre, Toronto, Ontario	12	0 – 7	0.3	23	23	13	29	54
			7 – 14	0.3	21	23	10	28	59
			14 – 21	0.2	23	25	8	28	59
			21 – 28	0.4	24	25	12	28	71
TRCA, 2008	Permeable pavement, Jerrett's Funeral Home, Vaughan, Ontario	10	0 – 7	0.2	26	24	8	24	51
			7 – 14	0.2	23	15	6	22	42
			14 – 21	0.2	21	14	5	20	39
			21 – 28	0.2	21	14	7	21	40
			28 - 35	0.2	21	13	6	24	36
TRCA, 2008	Permeable pavement, University of Guelph, Guelph, Ontario	13	0 – 7	0.5	8	16	53	12	290
			7 – 14	0.7	11	14	92	13	310
			14 – 21	0.5	12	11	30	16	270
			21 – 28	0.4	10	12	29	26	290
			28 - 35	0.2	7	8	41	39	260
Dierkes and Geiger, 1999	Vegetated filter strip, Highway A2, Essen, Germany	16	0 – 5	3.9	NT	413	239	NT	527
			5 - 10	3.5	NT	78	202	NT	361
			10 - 30	2.7	NT	31	34	NT	99
Dierkes and Geiger, 1999	Vegetated filter strip, Highway A31, Essen, Germany	11	0 – 5	4.3	NT	268	276	NT	759
			5 - 10	2.6	NT	69	130	NT	303
			10 - 30	2.5	NT	24	54	NT	112
Dierkes and Geiger, 1999	Vegetated filter strip, Highway A42, Essen, Germany	24	0 – 5	5.6	NT	167	290	NT	1580
			5 - 10	8.5	NT	155	348	NT	1630
			10 - 30	3.1	NT	23	27	NT	138
TRCA, 2008	Vegetated swale, TRCA Head Office parking lot, Toronto, Ontario	11	0 – 7	0.5	34	34	22	34	130
			7 – 14	0.5	27	29	15	35	67
			14 – 21	0.2	26	23	9	30	57
			21 – 28	0.4	31	21	10	34	58
			28 - 35	0.2	29	25	12	35	72
TRCA, 2008	Vegetated swale, Residential road (Royal York Area) Toronto, Ontario	<18	0 – 7	0.4	13	12	17	10	75
			7 – 14	0.4	8	8	9	11	27
			14 – 21	0.4	8	12	39	13	51
			21 – 28	0.4	7	15	39	19	49
			28 - 35	0.4	9	14	18	21	43
TRCA, 2008	Vegetated swale, Residential road (DeVere Gardens), Toronto, Ontario	>18	0 – 7	0.3	24	29	18	20	95
			7 – 14	0.4	26	20	40	21	100
			14 – 21	0.5	31	25	48	28	110
			21 – 28	0.7	25	29	49	21	97
			28 - 35	0.2	27	21	23	24	75
J.F. Sabourin & Assoc., 2008	Vegetated swale, Residential road (Heart's Desire Site SH9) Nepean, Ontario	13	0 - 15	NT	NT	0.9	20	NT	257

Study Reference	BMP Type and Location	Age of Facility (years)	Depth below base (cm)	Concentration (μg/g) Cadmium	Chromium	Copper	Lead	Nickel	Zinc
J.F. Sabourin & Assoc., 2008	Vegetated swale, Residential road (MacFarlane Site SM2) Nepean, Ontario	13	0 - 15	NT	NT	<0.2	19	NT	238
J.F. Sabourin & Assoc., 2008	Vegetated swale, Residential road (MacFarlane Site SM7) Nepean, Ontario	13	0 - 15	NT	NT	<0.2	43	NT	209
Barraud *et al.*, 1999	Soakaway, Valence, France	> 30	0 – 5[4]	~ 1.8	NT	NT	~ 125	NT	~225
			5 – 20	~ 2.5	NT	NT	~ 300	NT	~225
			20 – 65	~ 1.4	NT	NT	~100	NT	~100
Dechesne *et al.*, 2005	Infiltration basin, Centre Routier, Lyon, France	15	0 – 5	3.53	NT	110.4	147.8	NT	1145
			30 - 40	0.94	NT	14.0	28.1	NT	182
			60 - 70	0.47	NT	7.5	6.5	NT	71.9
Dechesne *et al.*, 2005	Infiltration basin, Homme, Lyon, France	21	0 – 5	3.01	NT	355.7	335.6	NT	1156
			30 - 40	1.06	NT	54.4	96.6	NT	254
			60 - 70	1.52	NT	45.6	58.7	NT	132
Dechesne *et al.*, 2005	Infiltration basin, Chene, Lyon, France	12	0 – 5	2.05	NT	256	191	NT	2605
			30 - 40	1.55	NT	173	177	NT	1725
			60 - 70	0.52	NT	19.2	12.4	NT	200
Dechesne *et al.*, 2005	Infiltration basin, Pivolles, Lyon, France	10	0 – 5	1.97	NT	173	930	NT	1033
			30 - 40	0.99	NT	85.5	428	NT	538
			60 - 70	0.47	NT	36.1	180	NT	221
Ontario proposed background standard, soil[1]				1.2	70	92	120	82	290
Ontario proposed site condition standard, surface soil[2]				1.2	160	140	120	100	340
Ontario proposed site condition standard, subsurface soil[3]				7.9	240000	5600	1000	510	47000

Notes:
1. Proposed full-depth background site condition standards for non-agricultural property uses (OMOE, 2008).
2. Proposed stratified site condition standards for residential, parkland, institutional property use, surface, coarse-textured soils in a potable groundwater condition (OMOE, 2008).
3. Proposed stratified site condition standards for residential, parkland, institutional property use, subsurface, coarse-textured soils in a potable groundwater condition (OMOE, 2008).
4. Soil at 0 – 5 cm depth below a soakaway, or any other subsurface infiltration practice, would likely be considered a subsurface soil and that the Ontario site condition standards for subsurface soils would apply.

NT = Not tested.

(grey) = exceeds proposed Ontario soil background standard and is considered an elevated concentration.

(magenta) = exceeds proposed Ontario soil background standard and site condition standard for residential/parkland/institutional property use, *surface*, coarse-textured soils in both potable and non-potable groundwater conditions, and would require removal and landfilling.

(dark red) = exceeds proposed Ontario soil background standard and site condition standard for residential/parkland/institutional property use, *subsurface*, coarse-textured soils in both potable and non-potable groundwater conditions, and would require removal and landfilling.

Preliminary results from monitoring the performance of a newly installed rain garden in a residential community in North Carolina indicates that they can be effective infiltration practices, even on soils with high clay content (Estes, 2009). The rain gardens were located on sandy clay soil where infiltration rates ranged from 29-38 mm/hr, with an average rate of 33 mm/hr, and were designed to retain and infiltrate the two-year design storm (a 79 mm event). After 4.5 months of monitoring, including 37 storm events of up to 38 mm in size, the average infiltration rate through the facility was 7 mm/hr, with the rate increasing to 25 mm/hr in the underlying native soil, once water levels were past the bottom of the installed soil mixture and filter fabric (Estes, 2009).

Recent studies clearly indicate that bioretention systems can be effective at controlling peak discharge rates and reducing runoff volume, thereby helping to achieve the Low Impact Development objective of maintaining predevelopment hydrology. Typical peak flow reductions of 44 to 64% were observed from two underdrained facilities at the University of Maryland after two years of monitoring, and flow peaks were significantly delayed, usually by a factor of 2 or more (Davis, 2008). Investigations of the hydrologic performance of six underdrained bioretention cells in Maryland and North Carolina indicate that substantial delays in peak flow and decreases in runoff volume can be achieved (Li *et al*., 2009). Annual water budget analysis by Li *et al.* suggests that approximately 20-50% of runoff entering the bioretention cells was either infiltrated into the native soil or lost through evapotranspiration. Some facilities reduced runoff volume by greater than 90% over the monitoring period, based on median ratios of influent to effluent volume over a 24 hour period (Li *et al.,* 2009).

Surface water quality

Performance results from both laboratory and field studies are promising and suggest that bioretention systems have the potential to be one of the most effective BMPs for pollutant removal. In laboratory studies of bioretention system prototypes (Davis *et al.*, 2001) reductions in metal concentrations (lead, zinc and copper) were greater than 90%. Plant uptake accounted for approximately 5% removal by mass. Total Kjeldahl nitrogen (TKN) retention was 68% and ammonia nitrogen retention was 87%. The only nutrient not well retained was nitrate nitrogen which had a retention rate of 24%.

Several field investigations of bioretention have been performed. In the Ontario study of a bioretention swale cited previously (TRCA, 2008), the effluent from the underdrain at one metre below the swale surface contained significantly lower concentrations of zinc than surface runoff from the asphalt, and other common roadway contaminants such as lead and PAH were detected much less frequently. In Maryland, synthetic runoff was applied to two different bioretention areas (Davis *et al.*, 2003). Removal of lead, zinc and copper was greater than 95% at one site, with lower removal rates observed at the second site (70% for lead, 64% for zinc and 43% for copper). High retention of metals has also been observed in facilities in New Hampshire (Roseen *et al.*, 2006), where 99% of zinc in runoff was retained, and in North Carolina (Hunt *et al.*, 2006) where retention rates of 81% for lead, 98% for zinc and 99% for copper were observed.

Improvements in parking lot runoff quality were documented by Davis (2007) for two bioretention cells at the University of Maryland. Overall composite median percent removals based on event mean concentrations for the two cells were 83% for lead, 62% for zinc, 57% for copper, 47% for total suspended solids and 76% for total phosphorus (Davis, 2007). Mass contaminant removal rates were higher than concentration based removal rates due to the attenuation of flow volume by the bioretention media. Much higher removal rates for total suspended solids, between 97-99%, have been documented through field tests at the University of New Hampshire Stormwater Center (Roseen *et al*., 2009). The University of Maryland bioretention cell was also effective at removing polycyclic aromatic hydrocarbon (PAH) pollutants from parking lot runoff. Event mean concentration reductions ranging from 31 to 99% were observed, with an average mass load reduction to the receiving waterbody of 87% (Diblasi *et al.,* 2009).

In an evaluation of metal retention and the fate of chloride in bioretention facilities receiving snow melt runoff from different types of urban roads in Norway, it was found that the facilities achieved excellent reductions in mass of metal contaminants from the snow to the outflowing melt water (Muthanna *et al.*, 2007). Mass reductions from 89% (for total copper) to 99% (for total lead) were observed, clearly demonstrating that bioretention can be used successfully to treat snowmelt from urban roads. The top mulch layer was responsible for the most significant metal retention (up to 74% for zinc). Uptake of dissolved metals by plants was estimated to be in the range of 2% to 8% (Muthanna *et al.*, 2007). However, concentrations of bioavailable (dissolved) copper and zinc in outflows from the bioretention cells were higher than in the input snowmelt, which requires further investigation to determine means of achieving better retention (Muthanna *et al.*, 2007).

Field investigations of nutrient retention have produced more variable results. In a Connecticut study, an increase in total phosphorus was observed in infiltrate (Dietz and Clausen, 2006). The export of total phosphorus from bioretention systems has been observed in other studies as well (Hunt *et al.*, 2006; TRCA, 2008). These findings have been attributed to high phosphorus content in the soil (Hunt *et al.*, 2006) and leaching of phosphorus from the mulch and organic soil used as the planting media in these systems (TRCA, 2008). Dietz (2007) notes that the combination of phosphorus export and an underdrain that is directly connected to the storm sewer system could cause more harm than good if a sensitive water body were downstream. To avoid such problems he suggests that the phosphorus content of the soil media used in a bioretention area should be examined, and if it is very high, an alternative media should be used. With the exception of the Connecticut study (Dietz and Clausen, 2006), nitrate nitrogen retention in bioretention systems has consistently been observed to be low, likely due to low adsorption of negatively charged nitrate ions to soil particles. Recent evidence suggests improvements to nitrogen removal can be achieved by designing facilities so that the bioretention media remains saturated for a significant period, which creates anerobic conditions under which denitrification by bacteria is possible (Kim et al., 2003; Dietz and Clausen, 2006; Hunt et al., 2006).

Little data exists on the ability of bioretention areas to reduce bacteria concentrations, but preliminary results of a laboratory study report an average removal rate of 88% of fecal coliform bacteria in simulated bioretention columns (Rusicano and Obropta, 2005). In the King City study in Ontario, mean concentrations in the bioswale underdrain were only 35 CFU/100 mL, compared to 302 CFU/100 mL in asphalt runoff (TRCA, 2008). Both the mean and median concentrations of bioswale effluent were below the Provincial Water Quality Objective for swimming areas (100 CFU/100 mL). Initial studies of an underdrained bioretention cell treating parking lot runoff in Charlotte, North Carolina show significant reductions in event mean concentrations of fecal coliform and *E. coli*, in the order of 70% (Hunt *et al*., 2008).

Few studies have investigated the effect of bioretention facilities on runoff temperature. In Connecticut, no temperature difference was found between inflow and underdrain flow from a rain garden (Dietz and Clausen, 2005), while a North Carolina study found significant reductions in both maximum and median water temperatures between the inlet and outlet of two bioretention areas (Jones and Hunt, 2009). It was noted by Jones and Hunt (2009) that reductions in runoff volume that are achieved by bioretention facilities also effectively reduce thermal impacts to receiving waters.

Groundwater quality and soil quality

There is a paucity of research on the effects of bioretention practices on groundwater quality and soil quality. This is of particular interest in cold climate applications where bioretention facilities may be used for snow storage and receive snow melt containing de-icing salt constituents, which could reduce the retention of some metals (*e.g.*, lead, copper and cadmium) in the soil and potentially increase metal concentrations in shallow groundwater.

Soil cores extracted from three bioretention facilities in the Greater Toronto Area ranging in age between 2 and 5 years showed metal and PAH levels comparable to nearby reference sites unimpacted by runoff (TRCA, 2008). All concentrations were below Ontario background concentrations. A repeat survey of one facility after two years showed no change in contamination. Depth profiles showed no consistent variation in contamination with depth (TRCA, 2008).

6.1.4 Soakaways, infiltration trenches and chambers

Runoff reduction

The performance of soakaways, infiltration trenches and chambers on commercial or residential lots would be expected to reduce runoff in a manner similar to perforated pipe systems installed as part of the conveyance system (see next section). Preliminary data from hydrologic monitoring of an infiltration chamber installed on silty sand soils that receives runoff from the roofs of two large retail outlets reported an 87% reduction in runoff volume over a 6 month period. Based on drawdown times, the rate of soil infiltration was calculated to range between 1 and 5 mm/hr., which is considerably lower than the estimate of 14 mm/hr. made for the native soil prior to installation of the chamber (TRCA, 2009).

In a New Hampshire field study examining seasonal performance variations for various types of stormwater BMPs in cold climate conditions, it was observed that an infiltration chamber system showed the least variability, attributable to its location well below the frost line (Roseen *et al.,* 2009). All infiltration systems examined (surface and subsurface) exhibited similar peak flow reduction and contaminant removal performance between summer and winter seasons.

Surface water quality

Based on a synthesis of performance monitoring studies available as of 2000, it was reported that infiltration trench pollutant removal efficiencies can be expected to range from 70 to 90% for suspended solids, 40-70% for total nitrogen, 50-70% for total phosphorus and 70-90% for total heavy metals (ASCE, 2000). In an infiltration trench designed for partial infiltration, Guo *et al.* (2006) observed removal rates for ionic (dissolved) forms of metals of 51% for lead, 95% for zinc, 91% for copper and 49% for cadmium. They also observed removal rates for organically bound forms of metals of 85% for lead, 98% for zinc, 97% for copper and 82% for cadmium (Guo *et al.*, 2006).

Groundwater quality
Barraud *et al.* observed that concentrations of lead, zinc and cadmium in groundwater below a 30 year old soakaway installed on alluvial soils were well below French drinking water standard thresholds, with mean removal rate estimated to be 74% for zinc and 98.5% for lead (Barraud *et al.*, 1999).

In Sweden, Norrström (2005) examined groundwater quality below a 32 year old infiltration trench that receives runoff from a high traffic (average of 50,000 vehicles per day) highway and is subject to routine application of de-icing salt (sodium chloride) during winter. Groundwater sampled below the infiltration trench was found to contain high levels of lead, with one sample taken in June exceeding the Swedish limit for drinking water quality at a depth of 2.5 metres below the base of the trench, while concentrations of cadmium and zinc did not exceed drinking water limits (Norrström, 2005). This finding supports results from laboratory testing of the effects of alternately infiltrating sodium chloride laden water and deionized water through soil columns extracted from the infiltration trench, which indicated that lead was likely being mobilized from the soil and into the groundwater through colloid-facilitated transport (Norrström, 2005). However, it should be noted that the total amount of lead leached from the columns was very low with only 0.06% to 0.15% of the total lead content in the soil being leached out, confirming that lead is highly immobile in soils.

Soil quality
Concentrations of metals and hydrocarbons in an alluvial soil below a 30 year old soakaway receiving runoff from a medium traffic road (7000 vehicles per day) showed high levels of lead, cadmium, zinc and hydrocarbons in the first 10 cm of underlying soil (**Table 6.1**) with levels falling off sharply thereafter with depth, with levels meeting Dutch standards for non-polluted soils at 20 to 30 centimetres depth (Barraud *et al.*, 1999).

6.2 Conveyance practices

Conveyance infiltration practices have been grouped for the purposes of this review into the general BMP types of grassed swale, perforated pipe and perforated catchbasin.

6.2.1 Grassed swale

Runoff reduction
Pitt and McLean (1986) monitored a residential area in Toronto served by both grassed swales and concrete curb and gutters. Stormwater flows in the portion served by swales were about 25% less than the portion served by curbs and gutters and very little flow was discharged from the swales for storm events less than 13 mm. Similarly, in a study of a 275 metre swale with two check dams that receives runoff from a Virginia highway, complete infiltration of runoff was observed for storms with less than 12.7 mm total precipitation (Yu *et al.*, 2001). Long term performance of 20 year old grassed swales was confirmed through testing of infiltration rates which showed that, while rates have declined since their installation, they remain within the range typically assumed for permeable grassed surfaces (J.F. Sabourin and Assc., 2008).

Surface water quality

Wang *et al.* (1980) monitored the effectiveness of grassed swales at several freeway sites in the state of Washington and found lead concentrations in runoff were typically reduced by 80% or more, while copper was reduced by about 60% and zinc by about 70%. A grassed swale receiving runoff from a commercial parking lot in New Hampshire was observed to reduce concentrations of metals (lead, zinc, cadmium and copper) by about 50% and nitrate-nitrogen and ammonia nitrogen by about 25%, with no significant reductions found for organic nitrogen, phosphorus and bacteria (USEPA, 1983). Yousef *et al.* (1987) found swales adjacent to a highway in Florida produced total phosphorus removal efficiencies of between 25 and 30%, based on concentrations, with mass removal efficiencies that were much higher due to reductions in runoff volume from infiltration.

In a study of vegetated channels (highway medians) designed for stormwater conveyance in Texas, Barrett *et al.* (1998) found them to be effective as filtration systems for reducing the concentrations and loads of contaminants in highway runoff. The percent reduction in contaminant mass transport to receiving waters was above 85% for total suspended solids at both sites, which is comparable to removal efficiencies of other controls such as extended detention ponds. It was suggested that vegetated controls such as grassed swales and vegetated filter strips should be accepted by regulatory agencies as effective primary controls for treatment of highway and urban runoff (Barrett *et al.*, 1998). Similarly, in a study of a grassed swale receiving runoff from a Virginia highway, pollutant mass removal rates of 94% for TSS and 99% for total phosphorus were observed (Yu *et al.*, 2001).

A recent study of a roadside grassed swale in Sweden found that stormwater pollutant retention rates were variable with influent pollutant concentrations (Bäckström *et al.*, 2006). During high pollutant loading rates, the swale retained significant amounts of pollutants, but negative removal efficiencies were observed when the swale received runoff with low pollutant concentrations. These findings indicate that once pollutants are trapped in the swale, they are not permanently bound to vegetation or soil. They conclude that a roadside grassed swale may be regarded as a stormwater treatment facility that attenuates peaks in pollutant loads without being capable of producing consistently high removal rates (Bäckström *et al.*, 2006). Bäckström *et al.* (2006) suggest that grassed swales designed for pollution control should have a swale area equal to, or larger than the contributing impervious area.

As reported by Deletic and Fletcher (2006), median pollutant removal rates of swales from available performance studies are 76% for TSS, 55% for total phosphorus and 50% for total nitrogen. In their own field studies of the performance of grassed swales they observed variable TSS removal rates ranging from 61-86%. They concluded that TSS removal from runoff is primarily a physical process, reflecting the balance between flow and particle settling velocity and that removal performance is a function of flow rate, grass density and particle size and density (Deletic and Fletcher, 2006).

Groundwater quality

Since most of the runoff flowing through swales is conveyed rather than infiltrated, most studies examining effects on groundwater quality have not been conducted on this type of practice.

Soil quality

Zinc concentrations in soil above 40 cm depth under a five year old grassed swale receiving runoff from a zinc roof were observed to exceed German critical values (Zimmermann *et al.*, 2005). In a similar study the accumulation of contaminants in the soils of infiltration swales receiving runoff from supermarket parking lots in Austria was investigated through sampling (Achleitner *et al.*, 2007). Swales examined ranged from 2 to 10 years of age, with mean daily traffic loads at the parking lots ranging from 620 to 800 vehicles per day. Observed concentrations of hydrocarbons and heavy metals (lead, copper, zinc and cadmium) did not exceed Austrian guidelines for landfilling. No distinct correlations between observed contaminant concentrations and traffic load, nor age of the swale were found (Achleitner *et al.*, 2007). Similar conclusions were made following examination of surface soil quality in two grassed swale systems receiving residential road runoff in Nepean, Ontario that had been in operation for 15 to 20 years. Observed concentrations of lead, copper, zinc and mercury were below Ontario soil background standards for nonagricultural land uses (J.F. Sabourin and Assoc., 2008).

6.2.2 Perforated pipe and catchbasin

Runoff reduction

A series of interconnected perforated catchbasins in Long Island, New York, were found to recharge more than 99% of stormwater flow (USEPA, 1983). In a study examining two newly constructed residential neighborhoods with perforated pipe systems in Nepean, Ontario, runoff volumes were observed to be 99% and 86% less than a similar conventional pipe system (Paul Wisner and Assoc., 1994). The difference in runoff coefficients between the two perforated pipe system sites was attributed to a high groundwater table affecting one of the sites. Follow-up studies in 1999 (J.F. Sabourin and Assoc., 1999) and 2006 (J.F. Sabourin and Assoc., 2008) showed that the systems continued to exfiltrate similar volumes of runoff. In 1998, peak flows were 90% less than those observed for the conventional system and runoff volumes were 94% and 70% of the conventional system flows (J.F. Sabourin and Associates, 1999). In 2006, peak flows were between 47% and 86% less than those from the conventional system and runoff volumes were 86% and 73% of the conventional system flows (J.F. Sabourin and Assoc., 2008).

Performance of two perforated pipe systems installed in Etobicoke and North York, Ontario, that receive roof and road runoff from low density residential areas was examined with regard to effects on runoff quantity (SWAMP, 2002). Soils at the Etobicoke exfiltration system were clay to clayey-silt till over silty sand (infiltration rate between 0.004 to 36 mm/hr). Soils at the North York system were silty sand (infiltration rate 72 to 288 mm/hr). A year after installation, the systems were found to be effective in exfiltrating most of the runoff directed into the perforated pipes, exceeding their design criteria. The Etobicoke and North York exfiltration systems were observed to exfiltrate 95% and 89% of all runoff from storms greater than 5 mm, respectively over the two years of monitoring (SWAMP, 2005; SWAMP, 2002). High exfiltration rates to soils under the Etobicoke system were attributed to the presence of local sand lenses or fissures in the native clay soil matrix (SWAMP, 2005).

A pilot study of another exfiltration system installed in sandy silt soils in Vaughan, Ontario demonstrated 100% infiltration of roof drainage over the 15 month study period, during which the highest volume rainfall event was 45 mm over 2 days (Clarifica Inc. and Schaeffers, 2005).

Surface water quality

Seasonal reductions in contaminant loads achieved by exfiltration systems installed in silty sand soils in Toronto, Ontario exceeded 80% for most constituents in runoff from a low density residential area, with the exception of chloride, and were due primarily to reductions in runoff volume (SWAMP, 2005). In Ottawa, monitoring of residential roof and road runoff treatment through a perforated pipe system with pretreatment through a grassed swale indicated higher average concentrations of chloride, *E.Coli* and chromium than in a conventional pipe system (Paul Wisner and Assoc., 1994; J.F. Sabourin and Assoc., 1999; J.F. Sabourin and Assoc., 2008). However, because of much lower runoff volumes, the perforated pipe systems were shown to release significantly less pollutants than the conventional system, even after 20 years of operation (J.F. Sabourin and Assoc., 2008). Loadings of sediment, phosphorus, nitrogen, copper, lead and zinc in runoff flowing from the grassed swale/perforated pipe systems were between 1% and 25% of loadings from a similar catchment with conventional catchbasins and storm sewer pipes (J.F. Sabourin and Assoc., 1999; J.F. Sabourin and Assoc., 2008).

Groundwater quality and soil quality

There is a paucity of research on the effects of perforated pipe and catchbasin practices on groundwater quality and soil quality. This is of particular interest in cold climate applications where such facilities may receive snow melt containing de-icing salt constituents, which could reduce the retention of some metals (*e.g.*, lead, copper and cadmium) in the soil and potentially increase metal concentrations in shallow groundwater.

6.3 End-of-pipe practices

The infiltration practice most commonly used at the end-of-pipe is an infiltration basin. These are relatively rare in Ontario but several studies of these facilities have been conducted in Europe and the United States.

6.3.1 Infiltration basin

Runoff reduction

In Long Island, New York, infiltration basins in use since the 1930s to infiltrate urban runoff have maintained predevelopment groundwater levels (Ku and Simmons, 1986). Four infiltration basins in France, installed on gravelly soils, were observed to have good infiltration capacities after 10 to 21 years of service (Dechesne *et al.*, 2005).

Surface water quality

Infiltration basins typically rely primarily on infiltration to reduce runoff pollutant loads to watercourses. Therefore, studies of their performance have focused largely on their effects on groundwater and soil quality.

Groundwater quality

Concentrations of metals, nutrients pesticides and phenolic compounds in groundwater near three infiltration basins studied in Perth, Australia, receiving runoff from a mixture of light industrial, medium density residential and a high traffic road, and located on predominantly sandy soil with some clay and limestone, were low and well within drinking water guidelines (Appleyard, 1993). No significant contamination of groundwater was observed under five infiltration basins in California, ranging in age from 2 to 20 years that received runoff from predominantly residential land uses and were located on sandy alluvial soil (Nightingale, 1987b). Despite the highly permeable soils typical of most of Long Island, New York, Ku and Simmons (1986) found infiltration basins effective in removing bacteria and metals from stormwater before it reached the water table.

Salo et al (1986) monitored groundwater quality below five groundwater recharge basins, two of which had been in operation for more than 20 years at the time of the study. Several organic compounds were monitored including chlorinated pesticides, organo-phosphorus pesticides, chlorophenoxy herbicides and phenolic compounds. Examination of these samples revealed no adverse effects on groundwater as a result of infiltrating stormwater. Citing several studies of infiltration systems conducted in western European countries, Mikkelsen *et al.* (1994) reached a similar conclusion about the potential for groundwater contamination associated with stormwater infiltration.

At three stormwater infiltration facilities in Maryland, the nearby use of de-icing salt and subsequent infiltration to the groundwater shifted the major-ion chemistry of the groundwater to a chloride-dominated solution (Wilde, 1994). As expected, sodium and calcium ion concentration were also elevated in groundwater beneath the infiltration devices (Wilde, 1994). In a New Jersey study, groundwater quality beneath 16 infiltration basins was compared to ambient groundwater in the study area (Fisher *et al.*, 2003). Groundwater samples collected from wells installed in the basins exhibited lower levels of dissolved oxygen and greater detection frequency of petroleum hydrocarbons such as benzene and toluene. Pesticides used to control weeds along roads were also detected in greater frequency in groundwater beneath the infiltration basins (Fisher *et al.*, 2003).

Considering that infiltration basin practices represent the highest concentration of urban runoff and the highest contaminant accumulation rate, the relative lack of observations of significant contamination of underlying shallow aquifers, even after greater than 20 years of service, suggests that risk of groundwater contamination from infiltration practices can be properly managed through appropriate screening of suitability, siting and design.

Soil quality

Concentrations of metals (lead, zinc, cadmium, and copper) observed in infiltration basin soils receiving runoff from most residential developments in Fresno, California, after 2 to 12 years of operation did not constitute a hazard to the use of the basins for recreational or groundwater recharge purposes (Nightingale, 1987a). Sampling of soils in the basins showed large amounts of lead accumulating in the first 5 cm of soil (leaded gasoline was still in use at the time of the study). The amount of lead decreased in the 5 to 15 cm depth interval and reached natural background levels in the 15 to 30 cm interval. Wigington *et al.* (1983) confirmed that metals such as lead accumulate in the top few centimetres of soil and that movement downward through the soil is limited. Salo et al (1986) reported sharp declines in soil concentrations of lead, arsenic, nickel, and copper in the first 1 metre below five groundwater recharge basins, two of which had been in operation for more than 20 years at the time of the study. In France, studies of soil quality below four infiltration basins ranging in age between 10 and 21 years (Table 6.1), reported that soil contamination (metals, PAHs, hydrocarbons, nutrients) was limited to less than 50 cm below the basin bed (Barraud *et al.*, 2005; Deschesne *et al.*, 2005). The basins were constructed over gravelly calcerous soils with high infiltration rates (greater than 360 mm/hr).

Two 85 to 100 year old infiltration basins were discovered on the campus of Villanova University in Pennsylvania. Soils below the basins were tested for copper, as it was found to be a contaminant of concern in other Villanova BMP sites. Copper concentrations were observed to peak at a depth of 46 cm with a maximum value of 364 mg/kg, representing elevated levels, but not in excess of the Pennsylvania standard for residential soils (Welker *et al.*, 2006).

7.0 INSPECTION AND MAINTENANCE

Maintaining the performance of any stormwater BMP over the lifespan of the facility requires that the facilities be inspected and maintained at appropriate intervals. Recommendations regarding appropriate inspection and maintenance activities for different types of BMPs are provided in most stormwater management design guidelines. Table 7.1 summarizes guidance regarding inspection and maintenance requirements for stormwater infiltration practices, drawing on manuals from selected cold climate jurisdictions.

As shown in Table 7.1, most guidelines recommend that maximum drawdown time be used to trigger maintenance or rehabilitation activities in older facilities where clogging is becoming an issue. In general, it is recommended that facilities undergo such maintenance work when flows require more than 72 hours to fully drain following a storm event.

Among infiltration BMPs, permeable pavement stands out as requiring more frequent inspection and maintenance, likely due to the absence of sedimentation pretreatment in most permeable pavement systems. Infiltration BMPs that include a vegetation component also require frequent inspection and maintenance in the initial years of operation in order to establish and maintain dense, healthy vegetation cover. Subsurface infiltration BMPs generally require less maintenance than surface infiltration systems. A study in Ottawa of a perforated pipe infiltration trench with pretreatment in a swale showed no significant reduction in performance even after 20 years of little or no maintenance (J.F. Sabourin and Assc., 2008).

If infiltration practices are installed on private property it is critical that a maintenance agreement between the review authority and facility owner is established and enforced. Covenants must also be instituted on these lands to ensure that the infiltration practice is replaced with a similar practice at the end of its useful life. This is one of the most important challenges of a Low Impact Development approach to stormwater management as it requires a relatively significant effort on the part of public agencies to monitor and enforce these agreements and covenants.

Table 7.1: Summary of Typical Operation, Inspection and Maintenance Requirements for Stormwater Infiltration Practices[1]

BMP Type	Operation[2, 3, 4]	Inspection[5]	Maintenance or Repair
Lot Level			
Permeable pavement	- Large trucks and other heavy equipment should be prevented from tracking or spilling dirt onto the pavement. - All construction equipment or hazardous material carriers should be prohibited from entering a site with permeable pavement. - Do not allow construction staging or soil/mulch storage on unprotected pavement surface. - Plowed snow should not be stored on permeable pavements.	- Inspect pavement, inlet and overflow structures **monthly** to ensure they are clear of sediment, trash and other debris and that the pavement reservoir draws down completely between storm events. - Inspect pavement, inlet and overflow structures **annually** in spring for structural damage.	**Biannually or as needed:** - Remove accumulated trash and other debris from the pavement surface and inlet structures (PDEP, 2006). - Vacuum sweep pavement surface 2 to 4 times a year with a commercial cleaning unit (Smith, 2006). - Routine snow clearing and moderate use of de-icing compounds; Non-toxic organic de-icers applied as blended magnesium or calcium chloride-based liquid products or as pretreated salt are preferable over regular salts; Abrasives such as sand or cinders should not be applied on or adjacent to the pervious pavement (PDEP, 2006). - Re-establish vegetation on barren upland pervious areas. - Grass pavers should be mowed with clippings removed and may require watering and moderate fertilizer application, like other turf areas (MPCA, 2008). - When properly constructed and installed, replacement of pavement is generally not required for 20 to 25 years (Smith, 2006).
Vegetated filter strip	- If used for sediment control during construction, it should be regraded and reseeded after construction has finished (PDEP, 2006). - If used as a storage area for plowed snow where de-icing salt is used, the area should be planted with salt-tolerant, non-woody plant species.	- Inspect **after every major storm event** or **quarterly** for the first two years, and **biannually** thereafter for vegetation density, damage by foot or vehicular traffic, channelization, accumulation of sediment, trash and other debris, and structural damage to flow dispersion devices (PDEP, 2006).	**Annually or as needed:** - While vegetation is becoming established, regular watering may be required (PDEP, 2006). - Remove trash and other debris from the filter strip, particularly following the spring melt event. - Remove accumulated sediment from pretreatment and flow spreading devices. - Maintain vigorous vegetative cover (85%), grass 4-6 inches in height; replace plantings if original species not established within reasonable time frame or if damage >50% occurs (PDEP, 2006). - Replace dead vegetation, remove invasive growth, dethatch, remove thatching and aerate (PDEP, 2006). - Remove accumulated sediment on filter strip surface or bottom of slope when dry and exceeds 25 mm depth (PDEP, 2006). - Repair eroded or sparsely vegetated areas by improving flow dispersion structures, filling with topsoil or stabilizing with erosion control matting, and seeding (PDEP, 2006). - If pools of standing water are observed along the slope, regrading and revegetating may be required.

BMP Type	Operation[2, 3, 4]	Inspection[5]	Maintenance or Repair
Bioretention	- If used as a storage area for plowed snow where de-icing salt is used, the area should be planted with salt-tolerant, non-woody plant species.	- Inspect after **every major storm event or quarterly** during the first two years and **biannually** thereafter for vegetation density, invasive species, standing water, clogging, erosion, and structural damage to inlet, outlet, and overflow structures (MPCA, 2008). **Annually:** - Test pH of planting bed soil (Dayton & Knight Ltd. *et al*., 1999).	**Annually or as needed;** - While vegetation is becoming established and during periods of extended drought, regular watering may be required (PDEP, 2006). - Remove trash and other debris from bioretention area, particularly following the spring melt event. - Replace dead vegetation and remove invasive growth (PDEP, 2006). - Remove accumulated sediment, trash, and other debris from any pretreatment devices and diversion, inlet and overflow structures (MPCA, 2008). - Re-spread mulch when erosion is evident and replenish as needed; Mulch replacement may be required every 2 to 3 years (PDEP, 2006). - Adjust pH of planting bed soil if pH <5.2 or >8.0 (Dayton & Knight Ltd. *et al*., 1999; MPCA, 2008). **Every 5 years:** - Rake or replace the top 5 – 20 cm of the media bed (Li and Davis, 2008b)
Soakaway	- A removable filter should be installed in the roof leader below the surcharge pipe to screen out leaves and debris (PDEP, 2006).	**Biannually:** - Inspect to ensure the facility draws down completely within 72 hours of a storm event. - Inspect pretreatment devices and inlet and overflow structures for accumulation of sediment, trash or other debris and structural damage.	**Annually or as needed:** - Remove accumulated sediment, trash, and other debris from any pretreatment devices and inlet and overflow structures (PDEP, 2006). - Clean out eaves troughs and ensure proper connection to the facility (PDEP, 2006). -Trim any roots that may be blocking pipes (CIRIA, 2007). - If the time required to fully drain exceeds 72 hours, drain via pumping and clean out perforated piping, if present; If slow drainage persists, the system may need removal and replacement of granular material and/or geotextile liner (PDEP, 2006).
Infiltration trench	- If used to treat road runoff it may be desirable to divert flow from the facility during the winter and spring snowmelt event to prevent infiltration of runoff laden with de-icing compound constituents.	**Biannually:** - Inspect to ensure the facility draws down completely within 72 hours of a storm event. - Inspect pretreatment devices and diversion, inlet, and overflow structures for accumulation of sediment, trash or other debris and structural damage.	**Annually or as needed:** - Remove accumulated sediment, trash, and other debris from any pretreatment devices and diversion, inlet and overflow structures. -Trim any roots that may be blocking pipes (CIRIA, 2007). - Re-establish vegetation on barren upland pervious areas. - If the time required to fully drain exceeds 72 hours, drain via pumping and clean out perforated piping, if present; If slow drainage persists, the system may need removal and replacement of granular material and/or geotextile liner (PDEP, 2006).

BMP Type	Operation[2, 3, 4]	Inspection[5]	Maintenance or Repair
Infiltration chamber	- If used to treat road runoff it may be desirable to divert flow from the facility during the winter and spring melt event to prevent infiltration of runoff laden with de-icing compound constituents.	**Biannually:** - Inspect to ensure the facility draws down completely within 72 hours of a storm event. - Inspect pretreatment devices and diversion, inlet, and overflow structures for accumulation of sediment, trash or other debris and structural damage.	**Annually or as needed:** - Remove accumulated sediment, trash, and other debris from any pretreatment devices and diversion, inlet and overflow structures. -Trim any roots that may be blocking pipes (CIRIA, 2007). - Re-establish vegetation on barren upland pervious areas. - If the time required to fully drain exceeds 72 hours, drain via pumping and clean out; If slow drainage persists, the system may need removal and replacement.
Conveyance			
Grassed swale	- Vehicles should not be parked or driven on a grassed swale designed for infiltration, and care should be taken to avoid excessive compaction by mowers (PDEP, 2006). - If used to treat runoff from areas subject to de-icing salt spreading, the swale should be planted with salt-tolerant plant species.	- Inspect **after every major storm event** or **quarterly** for the first two years, and **biannually** thereafter, all pretreatment devices and inlet, check dam, outlet and overflow structures for accumulation of sediment, trash and other debris and structural damage; inspect swale for vegetation density, erosion and formation of rills or gullies, pools of standing water and bank stability; inspect to ensure the facility draws down completely within 72 hours of a storm event (PDEP, 2006).	**Annually or as needed:** - While vegetation is becoming established, regular watering may be required (PDEP, 2006). - Remove accumulated sediment, trash and other debris from pretreatment devices and diversion, inlet, check dam, outlet and overflow structures. - Mow grass when swale is dry to maintain a height of 4-6 inches with removal of clippings (PDEP, 2006). - Repair eroded or sparsely vegetated areas by improving check dams or other flow dispersion structures, filling with topsoil or stabilizing with erosion control matting, and seeding or sodding (PDEP, 2006). - Dethatch swale bottom and slopes, remove thatching and aerate (PDEP, 2006). - Re-establish vegetation on barren upland pervious areas. - Remove accumulated sediment on swale surface when dry and exceeds 25 mm depth (PDEP, 2006). - If the swale is designed for infiltration and does not fully drain within 72 hours, regrade and revegetate the swale (PDEP, 2006).
Perforated pipe and catchbasin	- Should be located below pervious boulevards or grassed swales where they can be readily excavated for servicing. - If used to treat road runoff it may be desirable to divert flow from the facility during the winter and spring melt event to prevent infiltration of runoff laden with de-icing compound constituents.	**Biannually:** - Inspect all pretreatment devices and diversion, inlet, outlet and overflow structures for accumulation of sediment, trash and other debris and structural damage (PDEP, 2006).	**Annually or as needed:** - Remove accumulated sediment, trash, and other debris from any pretreatment devices and diversion, inlet and overflow structures. -Trim any roots that may be blocking pipes (CIRIA, 2007). - Re-establish vegetation on barren upland pervious areas. - If the time required to fully drain exceeds 72 hours, drain via pumping and clean out; If slow drainage persists, the system may need removal and replacement.

BMP Type	Operation[2, 3, 4]	Inspection[5]	Maintenance or Repair
End-of-pipe			
Infiltration basin	- If used to treat road runoff it may be desirable to divert flow from the facility during the winter and spring melt event to prevent infiltration of runoff laden with de-icing compound constituents. - Vehicles should not be parked or driven on an infiltration basin, and excessive compaction by mowers should be avoided (PDEP, 2006).	- Inspect **after every major storm event** or **quarterly** for the first two years, and **biannually** thereafter, all pretreatment devices and diversion, inlet and overflow structures for accumulation of sediment, trash and other debris and structural damage; inspect basin for vegetation density, erosion and formation of pools of standing water and bank stability; inspect to ensure the facility draws down completely within 72 hours of a storm event (PDEP, 2006).	**Annually or as needed:** - While vegetation is becoming established, regular watering may be required (PDEP, 2006). - Remove accumulated sediment, trash and other debris from pretreatment devices and diversion, inlet and overflow structures. - Mow grass when basin is dry with removal of clippings (PDEP, 2006). - Repair eroded or sparsely vegetated areas by improving check dams or other flow dispersion structures, filling with topsoil or stabilizing with erosion control matting, and seeding or sodding (PDEP, 2006). - Dethatch basin bottom and slopes, remove thatching and aerate (PDEP, 2006). - Re-establish vegetation on barren upland pervious areas. - If basin does not fully drain within 72 hours or **every 5 years**, remove accumulated sediment in the basin, regrade and revegetate.

Notes:

1. Adapted from WMI, 1997 unless otherwise noted.
2. A maintenance agreement between the review authority and facility owner is recommended for all infiltration BMP types on private property (PDEP, 2006).
3. Ensuring that the contributing drainage area is stabilized prior to bringing the practice on-line is recommended for all infiltration BMP types.
4. Roads and parking areas draining to infiltration BMPs should be regularly swept, particularly following the spring melt event.
5. Inspection during construction to ensure the facility meets the design standards and specifications is recommended for all BMP types (MPCA, 2008).

8.0 CONCLUSIONS AND FUTURE RESEARCH NEEDS

An encouraging finding from this review is that there are numerous studies documenting the performance of stormwater infiltration practices in cold climate regions. The vast majority of literature reports favorable performance for most parameters examined, suggesting that greater integration of infiltration practices into stormwater management system designs in cold climates could further reduce impacts of urbanization on receiving waters and their aquatic ecosystems. As the practice of stormwater infiltration in bioretention facilities and infiltration chambers is relatively recent, there are few studies that have examined performance of these practices after several years of operation. There is also insufficient information regarding effects on receiving water quality of infiltrating deicing salt laden runoff in small areas distributed across the catchment versus discharging runoff to centralized end-of-pipe facilities. These are topics requiring further research.

Comparison of guidelines on the suitability and siting of stormwater infiltration practices from selected cold climate jurisdictions reveals that while consistent direction is provided regarding the factors that should be considered, specific criteria vary considerably among jurisdictions. Of particular note are differences in direction regarding types of land uses considered to have potential to generate highly contaminated runoff (*i.e.*, stormwater hot spots or pollution hot spots) and unsuitable for application of stormwater infiltration practices. Current stormwater planning and design guidelines in Ontario can be interpreted as blanket restrictions on infiltration practices in any industrial or commercial land use, which leaves little flexibility for exceptions. Improving direction in this regard in the updated guideline would reduce a significant barrier to the application of infiltration practices in Ontario.

Guidelines reviewed consistently recommend that infiltration practices should not be applied in certain contexts. Areas with contaminated soil, areas of shallow depth (< 1 metre) to seasonally high water table or bedrock, particularly where the shallow aquifer is used as a drinking water source, and steep, unstable slopes are typically considered places where stormwater detention, evapo-transpiration and harvesting practices are more appropriate management strategies. Other areas where restrictions on infiltration practices may apply need to be identified based on a more thorough understanding of present and future groundwater uses, contaminant types and loads and the attenuation capacity of native soils (*e.g.*, Clark and Pitt, 2007).

A number of common concerns about the performance of stormwater infiltration practices have been addressed in the literature cited in this paper. Concern about the potential for clogging through the accumulation of fine sediments, which was a common occurrence in facilities built during the 1980s and early 1990s (Lindsey *et al.*, 1992), has been addressed through improvements to design, installation and maintenance, as indicated by recent performance monitoring studies, particularly for permeable pavement systems. While longer term performance studies are needed, the research to date indicates that with proper siting, design, installation and maintenance, stormwater infiltration practices are effective at preserving the predevelopment hydrologic function of a site and removing pollutants from runoff.

Concern about the effectiveness of infiltration practices in cold climates and on fine-textured soils have been topics addressed in several recent studies on stormwater infiltration technologies. Permeable pavement and bioretention facilities have been observed to function well in cold climates during winter months, even with frost in the ground, albeit at lower efficiencies than during warm weather. While guidelines in some jurisdictions discourage the application of infiltration practices on sites with fine-textured soils containing greater than 20% clay, recent studies have shown that substantial volumes of stormwater can be infiltrated in tight soils beneath permeable pavement installations, particularly when the drainage area is small relative to the footprint of the facility. Test applications and performance studies of infiltration practices on tight soils such as those prevalent around many of the rapidly growing urban areas in southern Ontario are needed to quantify runoff reduction and determine how infiltration rates may decline over time so that downstream treatment train controls can be designed accordingly.

The ability of infiltration practices to remove typical contaminants from urban stormwater runoff is becoming well established, with a few exceptions. High reductions in concentration (and loads) of suspended solids, metals, polycyclic aromatic hydrocarbons (PAH), and other organic compounds have been consistently observed in performance studies. Observations of effects on nutrient concentrations and loads have been more variable. Low retention rates for nitrates has been observed for both permeable pavement and bioretention systems, however concentrations and loads in urban runoff are typically below Canadian surface water and drinking water quality guidelines. A common observation across studies of bioretention and grassed swales is the export of dissolved phosphorus, likely due to high levels in the growing media. Adapting designs to utilize media with lower or slow-release phosphorus content, combined with pretreatment practices that help to retain nitrates and dissolved phosphorus (*e.g.*, vegetated filter strips, grassed swales), could improve net load reductions for these constituents.

A new horizon for research is on design adaptations and chemically reactive additives to infiltration technologies which could increase the removal of soluble contaminants such as metals, phosphorus and nitrogen from infiltrating stormwater. Very little research has been done on the ability of infiltration practices to remove pathogens (bacteria and viruses) from urban stormwater runoff, which is also a topic requiring further study. The suitability of stormwater infiltrated and stored in permeable pavement or soakaway reservoirs for reuse for non-potable purposes is another avenue of research being pursued (Pratt, 1999).

Risk of groundwater contamination from stormwater infiltration practices is the most common concern due to the presence of a wide variety of pollutants in urban runoff. Typical pollutants of concern include metals, nutrients, pathogens, dissolved minerals (*e.g.*, de-icing compound constituents), pesticides and other organic compounds (*e.g.*, PAH). Most of these pollutants are well retained by infiltration technologies and soils and therefore, have a low to moderate potential for groundwater contamination (Pitt *et al.*, 1999). Notable exceptions include some pathogens (*e.g.*, viruses) and de-icing salt constituents (*e.g.*, sodium chloride). Disinfection systems typically associated with drinking water supply wells destroy pathogens rendering the water safe for human consumption. Chloride and sodium, however, are not well attenuated in soil and can easily travel to shallow groundwater. Infiltration of de-icing salt constituents is also known to increase the mobility of certain heavy metals in soil (*e.g.*, lead, copper and cadmium), thereby raising the potential for elevated concentrations in underlying groundwater. While the processes by which this may occur are well documented (*e.g.*, Amrhein *et al.*, 1992; Bauske and Goetz, 1993; Norrström and Bergstedt, 2001; Bäckström *et al.*, 2004; Norrström, 2005), very few studies that have sampled groundwater below infiltration facilities or roadside ditches receiving de-icing salt laden runoff have found concentrations of heavy metals that exceed drinking water standards (Howard and Beck, 1993; Granato *et al.*, 1995). The few instances where this has been observed have received runoff from high traffic areas (*i.e.,* large highways) with elevated levels of metals. Some jurisdictions (*e.g.*, Maine and Minnesota) consider high traffic areas where large amounts of de-icing salt are used to be unsuitable for the application of stormwater infiltration practices..

It should be noted that, with the exception of infiltration basins, most infiltration practices are well distributed across the landscape, rather than centralized in a small area. With a distributed approach there is less potential for runoff to accumulate large masses of pollutants and therefore, the occurrence of elevated, potentially toxic concentrations of pollutants in the soil and groundwater is less likely. Collecting and treating stormwater from high traffic areas and pollution hot spots in centralized detention facilities, while using infiltration practices to treat runoff from roofs and low traffic areas may provide a good margin of safety where groundwater contamination is a concern (Marsalek, 2003; Dietz, 2007). While it is prudent to restrict infiltration practices in designated pollution hot spots, broader guidance regarding the suitability and siting of these practices should be provided through studies of hydrogeologic and hydrologic contexts at the watershed, subwatershed and local scales.

De-icing salts are extremely mobile in the environment and inevitably find their way into surface and groundwater systems wherever they are applied. Currently, the only feasible means of reducing the quantity of de-icing compounds released into the environment is to establish a framework of incentives and regulations that will ensure that practices designed to minimize their use are in place, and that less mobile and less toxic alternatives to salt are tested and used wherever possible. To this end, Environment Canada has initiated a voluntary process through which municipalities and road authorities are to develop salt management plans and reduce salt use through established best management practices (Transportation Association of Canada, 2003). The benefits and success of this approach are currently being evaluated and, depending on the outcome, further incentives or regulations may be instituted.

Landowners and municipalities have been concerned about soil quality below infiltration facilities and the potential need for future remediation and disposal. As expected, the degree of contamination varies depending on facility age and the size of the drainage area relative to the footprint of the facility. Available evidence indicates that small distributed infiltration controls such as permeable pavements do not contaminate underlying soils, even after more than 10 years of operation (TRCA, 2008). In France, large infiltration basins in operation for 10 to 21 years, show contaminant profiles in the upper 70 cm of soil (Table 6.1), however, concentrations of contaminants in this case were below Dutch remediation standards at a depth of 30 cm (Dechesne *et al.*, 2005). Based on these limited results it can be concluded that removal and landfilling of at least the upper 5 cm of soil below large centralized infiltration facilities may be required when the facilities are decommissioned.

Based on comparison of stormwater management design guidelines from selected jurisdictions, and consideration of recent research on the performance of infiltration practices in cold climate regions, some recommendations can be made regarding on-going work to update the Ontario guidelines:

- Consideration should be given to providing revised criteria for evaluating site suitability for stormwater infiltration practices. This guidance should be provided in an up-front, easy to locate section.
- Evidence that significant runoff reduction can be achieved by infiltration practices on fine-textured soils and that such practices continue to function during much of the winter, suggests that minimum

percolation rate of the native soil should not be used as a screening criterion (*e.g.*, BC MWLAP, 2002; CIRIA, 2007; MPCA, 2008), or that a much lower rate than the current 15 mm/hr should be used (*e.g.*, PDEP, 2006). Alternatively, different criteria could be recommended for facilities designed for partial infiltration (with an underdrain) than those designed for full infiltration (no underdrain).

- Consideration should be given to requiring underdrains with adjustable flow restrictors to be installed in facilities located on fine-textured soils with percolation rates less than 15 mm/hr in order to ensure complete drainage of water between rain events and reduce the potential for freezing during winter.
- Acknowledging that infiltration rates of newly built facilities will gradually decrease as they age and accumulate fine sediments, it is recommended that updated guidelines require that such reductions be factored into facility design.
- Clarification regarding the criterion for maximum drawdown time is needed, as the current Ontario guideline is vague in this regard. Such a criterion should consider the typical length of time between storm events in a given geographic location from long-term climate records, and the maximum acceptable amount of time that standing open water should be allowed to occur in an urban area to prevent mosquito-borne illnesses.
- As previously noted, improved guidance is needed regarding what conditions constitute areas where infiltration practices should not be applied due to risk of groundwater contamination. Examples of detailed guidance in this regard can be found in guidelines from other jurisdictions reviewed in this paper (*e.g.*, MDE, 2000; MPCA, 2008). While blanket restrictions may be appropriate in certain circumstances, broader application decisions should be based on a thorough understanding of present and future groundwater uses, contaminant types and loads and the attenuation capacity of native soils. Guidance regarding the suitability of infiltration practices in communities where water supply is derived from groundwater will also need to consider Ontario drinking water source protection requirements that may prohibit certain types of land use or human activities or require contaminant management plans be put in place within wellhead protection zones.
- Current guidance in the Ontario Stormwater Management Planning and Design Manual indicates that implementation of lot level and conveyance controls can only be used to reduce the active storage volume component of end-of-pipe facilities (*i.e.*, not the permanent pool volume). This criterion should be reviewed in light of the significant runoff volume and contaminant load reductions made feasible through the use of distributed micro controls upstream of the stormwater pond or wetland.

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