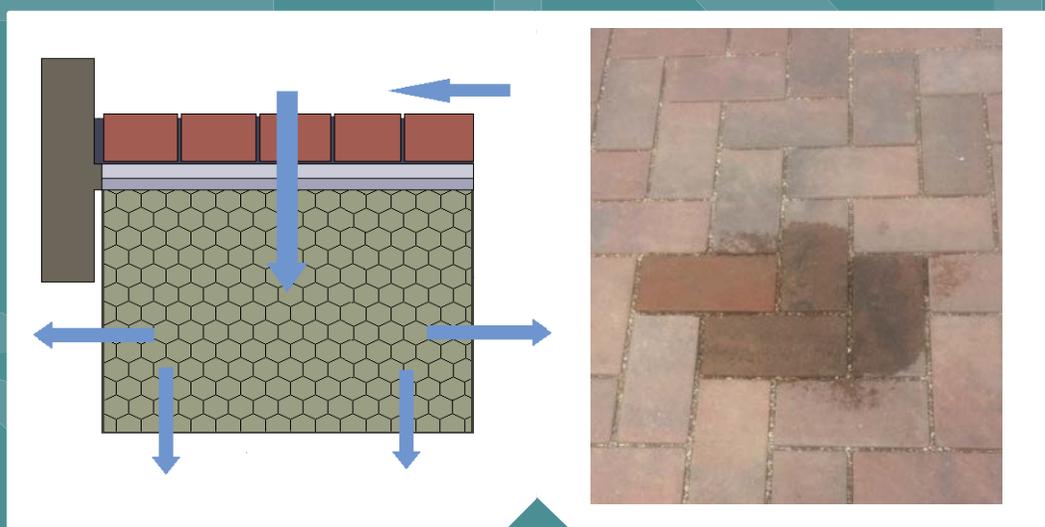


# The Influence of Green Infrastructure Practices on Groundwater Quality: The State of the Science



# **The Influence of Green Infrastructure Practices on Groundwater Quality: The State of the Science**

by

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# Abstract

Green infrastructure (GI) technologies applied to stormwater are developed to mimic natural infiltration and hydrologic processes. GI is a design strategy that enhances runoff storage volume, infiltrates runoff, and contributes to groundwater recharge. Urban development often leads to the removal of vegetation and soil, and replacing them with large stretches of impervious surfaces. This disturbance of the natural hydrologic cycle due to urbanization is closely connected to deteriorating urban water quality and enhanced flood risks.

When GI is used for urban runoff, there are concerns as to how the soils and subsurface geology/sediments interact with the stormwater runoff constituents, thus providing possible risks of groundwater quality impairment. Groundwater can be contaminated by many constituents: nutrients, metals, dissolved minerals, pesticides, other organics, and pathogens. This review provides insight into the current state of knowledge of the influence of GI on the subsurface environment and groundwater. All types of GI were assessed, both surface and subsurface infiltration infrastructures from peer-reviewed literature, published reports, and conference proceedings. Issues addressed include: 1) pollutant risks that need further research, 2) new infrastructure that has not been researched in depth, and 3) determining local considerations when planning for green infrastructure.

When managing water resources, the tendency for contaminants to move between the ground and surface water needs to be considered. This requires an understanding of the native soil characteristics in the unsaturated zone and saturated zone as well as the hydrology. The primary geochemical processes that need to be considered as stormwater infiltrates are dissolution and precipitation, redox, ion exchange, adsorption/desorption, complexation/chelation, kinetics, mixing relationships, and colloid-facilitated transport. Simulation models are a potentially affordable way to predict risk as well as provide a decision-making tool for implementing GI. While many models are used to assess surface water and groundwater transport, few integrate GI; those that do integrate GI do not address groundwater contaminant transport.

The biology of the system can have various impacts. Microorganisms such as bacteria, viruses, and parasites can be a contamination risk depending on the unsaturated and saturated zone conditions, incubation time, and native microbial populations. Macrobiological organisms can enhance or cause complications for green infrastructure, but research on these is limited. Riparian zones do not have any studies specific to urban GI, but previous studies on riparian zone restoration show they could restore denitrification to urban streams, induce recharge, and serve as a less manipulative approach for enhancing infiltration into alluvial groundwater.

Overall, a better understanding of the risks associated with GI is needed to recognize the implications of GI on a longer temporal scale and wider spatial scale. When implementing GI, the local geology, climate, hydrology, biology, geochemistry, type of infrastructure, and contaminant loads need to be carefully considered to reduce risks to groundwater.

# Foreword

The U.S. Environmental Protection Agency (EPA) is charged by Congress with protecting the Nation's land, air, and water resources. Under a mandate of national environmental laws, the Agency strives to formulate and implement actions leading to a compatible balance between human activities and the ability of natural systems to support and nurture life. To meet this mandate, EPA's research program is providing data and technical support for solving environmental problems today and building a science knowledge base necessary to manage our ecological resources wisely, understand how contaminants affect our health, and prevent or reduce environmental risks in the future.

The National Risk Management Research Laboratory (NRMRL) within the Office of Research and Development (ORD) is the Agency's center for investigation of technological and management approaches for preventing and reducing risks from pollution that threaten human health and the environment. The focus of the Laboratory's research program is on methods and their cost-effectiveness for prevention and control of pollution to air, land, water, and subsurface resources; protection of water quality in public water systems; remediation of contaminated sites, sediments and ground water; prevention and control of indoor air pollution; and restoration of ecosystems. NRMRL collaborates with both public and private sector partners to foster technologies that reduce the cost of compliance and to anticipate emerging problems. NRMRL's research provides solutions to environmental problems by: developing and promoting technologies that protect and improve the environment; advancing scientific and engineering information to support regulatory and policy decisions; and providing the technical support and information transfer to ensure implementation of environmental regulations and strategies at the national, state, and community levels.

As part of the Safe and Sustainable Water Resources Research Program (SSWR) a report on the current understanding of the potential impacts to groundwater quality from the use of Green Infrastructure (GI) for stormwater management was initiated as part of the SSWR 5.02.B research effort. The goal of this report is to enhance scientific information on groundwater impacts and interactions from green infrastructure. In addition, the effort will provide the basis for long-term research on the effectiveness of GI as a suite of Best Management Practices for water resources. Potential impacts to groundwater quality are of at least three general types: 1) direct contamination of groundwater by infiltrated dissolved and suspended surface contaminants (e.g. microbials, oil and gas, pesticides); 2) indirect contamination through changing aquifer conditions that allow a potential contaminant to be mobilized (e.g., arsenic mobilization due to redox changes); 3) interaction of infiltrated water with existing subsurface contaminants (in either soil, subsoil, or groundwater) that could alter the spatial extent of existing contamination. The results of this report will be used to inform current and future GI research efforts on the potential pathways for changes in groundwater quality, as well as knowledge gaps requiring further investigation.

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# Acronyms and Abbreviations

1,1,1-TCA	1,1,1-Trichloroethane
AEC	Anion exchange capacity
AGI	Acute gastrointestinal illness
Al	Aluminum
AOX	Absorbed organically-bound halogens
As	Arsenic
ASR	Aquifer storage and recovery
AT	Alternative techniques
B	Boron
BMP	Best management practices
BOD	Biochemical Oxygen Demand
Ca	Calcium
CAFOs	Concentrated animal feeding operations
Cd	Cadmium
CEC	Cation exchange capacity
CFU	Colony-forming unit
Cl <sup>-</sup>	Chloride ion
CN <sup>-</sup>	Cyanide ion
Co	Cobalt
CO <sub>3</sub> <sup>2-</sup>	Carbonate ion
COD	Chemical oxygen demand
Cr	Chromium
Cs	Cesium
Cu	Copper
DDT	Dichlorodiphenyltrichloroethane
DO	Dissolved oxygen
DOC	Dissolved organic carbon
DR3M-QUAL	Distributed routing rainfall runoff model
Eh	Oxidation/reduction potential
F <sup>-</sup>	Fluoride ion
Fe <sup>2+</sup>	Iron (II) ion/ferrous ion
FEFLOW	Finite element subsurface FLOW system
F <sub>oc</sub>	Organic carbon content
GI/GSI	Green infrastructure/green stormwater infrastructure
HCO <sub>3</sub> <sup>-</sup>	Bicarbonate ion
Hg	Mercury
hr	Hour
HS <sup>-</sup>	Bisulfide ion
HSPF	Hydrologic stimulation program-Fortran
I <sup>-</sup>	Iodide ion
IS	Ionic strength
IUWM	Integrated urban water management
K	Potassium
K <sub>oc</sub>	Organic carbon/water partitioning coefficient
K <sub>ow</sub>	Octanol/water partitioning coefficient
LID	Low impact development

LIUDD	Low impact urban design and development
MAR	Managed aquifer recharge
MF	Membrane filtration
Mg	Magnesium
Mn	Manganese
Mo	Molybdenum
MODFLOW	Modular Flow Model
MOTH	Mineral oil type hydrocarbons
MOUSE	Hydroworks and Model of Urban Sewers
MPN	Most probable number
MTBE	Methyl tert-butyl ether
N	Nitrogen
Na	Sodium
NH <sub>3</sub>	Ammonia
NH <sub>4</sub> <sup>+</sup>	Ammonium ion
Ni	Nickel
NO <sub>2</sub> <sup>-</sup>	Nitrite ion
NO <sub>3</sub> <sup>-</sup>	Nitrate ion
NOM	Natural organic matter
NO <sub>x</sub>	Nitric oxide
NPDES	National Pollutant Discharge Elimination System
ON	Organic nitrogen
ORP	Oxidation/reduction potential
P	Phosphorus
PAHs	Polycyclic aromatic hydrocarbon
Pb	Lead
Pb <sup>2+</sup>	Lead (II) ion
PCB	Polychlorinated biphenyl
PCE	Perchloroethylene/tetrachloroethylene
PFAs	Perfluoroalkyl and polyfluoroalkyl substances
pH	measure of acidity or basicity; negative log of hydronium ion concentration
PO <sub>4</sub> <sup>3-</sup>	Phosphate ion
POC	Particulate organic carbon
PVC	Polyvinyl chloride
Rn	Radon
RZGI	Riparian zone green infrastructure
Sb	Antimony
SC	Source control
SCM	Stormwater control measures
Se	Selenium
SESOIL	Seasonal Soil Compartment Model
SO <sub>4</sub> <sup>2-</sup>	Sulfate ion
SOM	Soil organic matter
SPC	Specific conductivity
SQIDs/ SQUIDS	Stormwater quality improvement devices
STORM	Stormwater Treatment Overflow Runoff Model
SUDS/SuDS	Sustainable urban drainage systems/Sustainable drainage systems
SUTRA	Saturated-unsaturated transport
SUWM	Sustainable urban water management

SVOCs	Semi-volatile organic compounds
SWAP	Soil Water Atmosphere and Plant
SWMM	Stormwater Management Model
SWS	Stormwater sediment
TDS	Total dissolved solids
THC	Total hydrocarbons
TKN	Total kjeldahl nitrogen
TMDLs	Total maximum daily loads
TN	Total N
TOC	Total organic carbon
TOUGH	Transport of Unsaturated Groundwater and Heat
TP	Total P
TPH	Total petroleum hydrocarbons
TSS	Total suspended solids
TVS	Total volatile solids
U	Uranium
U.S.	United States
USDA	United States Department of Agriculture
US EPA	United States Environmental Protection Agency
USGS	United States Geological Survey
V	Vanadium
VOC	Volatile organic compounds
WBDOS	Waterborne disease outbreak surveillance system
WQ-COSM	Water Quality Capture Optimization and Statistics Model
WSUD	Water sensitive urban design
yr	Year
Zn	Zinc

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# 1.0 Introduction

The disturbance of the natural hydrologic cycle due to urbanization is closely connected to deteriorating urban water quality, often from nonpoint source pollution. This creates an increased risk to groundwater quality because of new pathways for contaminant introduction into groundwater, chemicals associated with the nonpoint source pollution from anthropogenic activities, and wastewater exposure (Schirmer et al., 2013). Due to risks to groundwater quality, it has been suggested that green infrastructure (GI) not be used in areas with potentially high contaminant loading, i.e., recycling centers, gas stations, and brownfields (Dietz, 2007). When infiltrating devices are installed and used for urban runoff, there are concerns as to how the soils interact with the stormwater runoff pollution while infiltrating into the subsurface, thus providing possible risks of groundwater quality impairment from areas with potentially high contaminant concentrations (Tedoldi et al., 2016). Few studies address whether green infrastructure technology can be a source or sink for stormwater contaminants, or whether they pose the risk of groundwater contamination.

The US EPA (2010) produced a report that discussed various case studies of green infrastructure implemented throughout the United States. Policies, goals, and incentives for using green infrastructure in flood mitigation scenarios were discussed. Although infiltration technology was promoted, the effects on groundwater quantity or quality were not addressed. Various reviews of green infrastructure have been done over the years and most focused on the hydrology, or surface and underdrain water contamination (Bedan and Clausen, 2009; Eckart et al., 2017). Also, most studies just focus on one type of GI, such as permeable pavements (Brattebo and Booth, 2003) or vegetative swales to name a few. There are few studies on GI designs and most of them rely on assumptions from surveys from local governments and models of GIs, and many do not incorporate monitoring GI after installation for groundwater quality (Bedan and Clausen, 2009). The purpose of this report is to give a review of the literature that covers green infrastructure and its impacts on groundwater quality. This includes an overview of the uses of GI for stormwater infiltration, the hydrology and geochemical aspects that should be considered, biological concerns, and models that have been used.

## 1.1 Stormwater Definitions

Stormwater runoff is defined as runoff from rainfall or snowmelt events over pervious or impervious surfaces (Minnesota Pollution Control Agency, 2018). As part of the Clean Water Act (33 U.S.C. 1251 et seq., 1972) and the 1987 Water Quality Act (33 U.S.C. 1251 et seq., 1987), the U.S. Environmental Protection Agency (US EPA) developed a stormwater permitting program (33 U.S.C. 1251 - 1376, 2002). Through the National Pollutant Discharge Elimination System (NPDES), permits are required for municipal, industrial, and construction site stormwater runoff. Stormwater is defined in the CFR 122.26(13) (NPDES) as "stormwater runoff, snow melt runoff, and surface runoff and drainage." Industrial stormwater runoff is defined as "discharge from any conveyance that is used for collecting and conveying storm water and that is directly related to manufacturing, processing or raw materials storage areas at an industrial plant" (CFR 122.26(14) (NPDES) (Protection of Environment, 1990). Stormwater runoff quantity and quality can reflect local geology and anthropogenic activities (Galfi et al., 2017).

## 1.2 Urban Stormwater

Urban stormwater runoff has been a documented issue for society dating back to 3000 B.C. and an engineering concern since the late 1800s in populous areas (Burian and Edwards, 2002; Dietz, 2007; Fletcher et al., 2015). Historically, the primary objective has been to move the stormwater quickly away

from urban areas, and is now the leading cause of impairments in our nations waterways (Dietz, 2007; US EPA, 2002; Fletcher et al., 2015). Today, many municipalities are seeing stormwater as an opportunity for additional water supply, increasing biodiversity, and improving microclimates, thus providing an opportunity for development of holistic approaches to urban stormwater management (Ashley et al., 2013; Fletcher et al., 2015). Stormwater reuse is triggering a paradigm shift from stormwater being considered a contaminant and a flood risk to a resource that can solve the pollutant and flood risk issue (Walsh et al., 2012; Marlow et al., 2013; Jiang et al., 2015).

Urban development often leads to the removal of vegetation and soil, replacing them with large stretches of impervious surfaces such as roads, buildings, parking lots, and driveways (Konrad, 2003; US EPA, 2010). A consequence of imperviousness is an increase in peak discharge and flood frequency on streams; these effects are more pronounced for moderate storms after dry periods. Urban development thus increases the chances of flooding (Konrad, 2003). Peak discharge and flooding are influenced by the intensity of the storm, storm duration, snowmelt, topography and geology of the stream basin, vegetation, and hydrologic conditions before the storm event. Human activities can influence peak discharge by modifying these factors (Konrad, 2003). Runoff from urban development leads to unnaturally high volumes of stormwater that erode stream banks; thus, large amounts of sediment can enter downstream water bodies (US EPA, 2010).

Traditional municipal stormwater management, referred to as gray infrastructure, includes pipes, sewers, and drainage networks developed alongside these impervious surfaces to either convey water to treatment facilities or rapidly move the water from the urban area and downstream into receiving waters (Berland et al., 2017; Konrad, 2003). This moves water out of the watershed, upsetting the hydrological balance by preventing streams and groundwater from being recharged (US EPA, 2010). For undeveloped areas, if storage capacity is reached in a natural area (forest/grassland) after a precipitation event, the stormwater runoff flows through the soil and becomes a subsurface flow; this would recharge aquifers or discharge into stream networks. The impermeable surfaces from urban areas create less storage capacity and accelerate the runoff where it flows faster than either overland and subsurface flow (Konrad, 2003).

Urban areas often use separate stormwater or combined stormwater management systems (Berland et al., 2017). Separate systems use separate pipes for stormwater and wastewater where stormwater is generally untreated and sent to surface waters. These are often in suburban areas or renovated urban areas. In combined systems stormwater and wastewater are carried through the same conveyance structure to a treatment facility. There is often limited storage capacity at the treatment facility and this increases the risk of overflowing and contaminating surface water bodies (Berland et al., 2017). Sometimes both these systems will lead to excessive soil moisture or raise shallow groundwater tables that flow into sewers and lead to overflows (Berland et al., 2017).

Not including major natural disasters such as hurricanes, flood damage in the United States averages \$6 billion annually (US EPA, 2010). More local governments are anticipating future flood risks, driving the development of additional green infrastructure systems to protect floodplains and prevent flood damage (US EPA, 2010). Various methods can be used for reducing flood hazards in urban areas including making flood prone areas into parks and playgrounds; making buildings and bridges elevated; use of floodwalls and levees; constructing buildings to withstand temporary inundation; use of drainage systems with increased capacity; using rooftops and parking lots to store water; and promoting infiltration and storage of water into the soil column through infiltration trenches; permeable pavements; soil amendments; and reducing the impermeable surface area (Konrad, 2003).

### **1.3 Green Infrastructure**

Green infrastructure (GI) has hydraulic and hydrologic benefits that are developed to more closely mimic natural infiltration and hydrologic processes. GI design strategy can retain storage volume, infiltrate runoff, and contribute to the groundwater recharge. GI can reduce the stress on the wastewater system, reduce combined sewer overflows to receiving water, reduce peak flow, restore impaired waters, and improve watershed health (Bedan and Clausen, 2009; US EPA, 2010; Tedoldi et al., 2016). The terminology used for green stormwater infrastructure varies from country, agency, and municipality. A summary of these terms can be seen in Table 1. While various terminology is used to describe these infrastructure designs, they all share two core principles: restore the natural hydrology, and reduce flooding risks from storm events.

**Table 1. Brief overview of the different terms used for sustainable stormwater management.**

Abbreviated Name	Full Name	History of Term	Reference
BMP	Best management practices	<ul style="list-style-type: none"> <li>• Used in North America</li> <li>• Term coined as part of the Clean Water Act</li> <li>• Used since 1949 for managing agricultural land</li> <li>• Now a universal term for pollution prevention (Pollution Prevention Act)</li> <li>• Must satisfy wastewater permit applications (NPDES)</li> <li>• Encompasses non-structural and structural practices</li> </ul>	Fletcher et al., 2015
SUDS/SuDS	Sustainable urban drainage systems/Sustainable drainage systems	<ul style="list-style-type: none"> <li>• Commonly used in the UK and Scotland</li> <li>• First used in the late 1980s-1992</li> <li>• Sustainable drainage triangle: quantity, quality, and habitat</li> <li>• SuDS are usually a train of technologies that work together</li> </ul>	Fletcher et al., 2015
AT	Alternative techniques	<ul style="list-style-type: none"> <li>• Also known as: Compensatory techniques or Techniques Alternative</li> <li>• French term since early 1980s</li> <li>• Solves drainage and pollution problems</li> <li>• Also used to improve quality of life</li> <li>• Aimed to counter the effect of urban expansion</li> </ul>	Fletcher et al., 2015

**Table 1. Brief overview of the different terms used for sustainable stormwater management.**

Abbreviated Name	Full Name	History of Term	Reference
SCM	Stormwater control measures	<ul style="list-style-type: none"> <li>• A term the National Research review suggested should replace the term “best management practices”</li> <li>• BMP is vague and practices were not always the best.</li> <li>• Refers to structural and non-structural control.</li> <li>• Now used by the U.S. Federal Highway Administration and many state departments of transportation</li> </ul>	Fletcher et al., 2015
LID	Low impact development	<ul style="list-style-type: none"> <li>• Common in North America and New Zealand</li> <li>• Minimizes cost of stormwater management</li> <li>• “Design with nature approach”</li> </ul>	Fletcher et al., 2015
LIUDD	Low impact urban design and development	<ul style="list-style-type: none"> <li>• Term primarily used in New Zealand</li> <li>• Less about managing flow regimes and</li> <li>• More focused on pollution prevention</li> <li>• Focus on ecosystem health</li> <li>• Merges with indigenous Maori perspectives</li> </ul>	Fletcher et al., 2015
WSUD	Water sensitive urban design	<ul style="list-style-type: none"> <li>• Started in Australia in the 1990s</li> <li>• Manage the water balance</li> <li>• Maintain and/or enhance water quality</li> <li>• Encourage water conservation</li> <li>• Maintain water related environmental and recreational opportunities</li> <li>• Stormwater management is a subset of WSUD</li> <li>• Encompasses all aspects of urban water cycle management</li> </ul>	Fletcher et al., 2015

**Table 1. Brief overview of the different terms used for sustainable stormwater management.**

Abbreviated Name	Full Name	History of Term	Reference
IUWM	Integrated urban water management	<ul style="list-style-type: none"> <li>• First used in the 1990s</li> <li>• Used as a broad term</li> <li>• Relates to managing all parts of the water cycle in a catchment</li> <li>• Combines management of water supply, groundwater, wastewater and stormwater</li> </ul>	Fletcher et al., 2015
SC	Source control	<ul style="list-style-type: none"> <li>• Initially term for on-site stormwater systems.</li> <li>• 1980s, defined as a subset of onsite detention techniques</li> <li>• Term used in “Urban Drainage Design Guidelines” in Ontario and Vancouver</li> <li>• As LID term became more common, SC was known as small scale practices used throughout a watershed.</li> <li>• Reproduce or maintain pre-development hydrological conditions</li> </ul>	MetroVancouver, 2012; Fletcher et al., 2015
GI/GSI	Green infrastructure/Green stormwater infrastructure	<ul style="list-style-type: none"> <li>• 1990s, United States</li> <li>• Concept and process that goes beyond stormwater</li> <li>• Originated in landscape architecture and ecology</li> <li>• Realized by the USEPA to use for stormwater management</li> <li>• Used interchangeably with BMPs and LIDs</li> <li>• Network of decentralized stormwater management practices that reduces runoff and improves health of surrounding waterways through capturing and infiltrating stormwater</li> </ul>	Fletcher et al., 2015

**Table 1. Brief overview of the different terms used for sustainable stormwater management.**

Abbreviated Name	Full Name	History of Term	Reference
SQIDs/SQUIDs	Stormwater quality improvement devices	<ul style="list-style-type: none"> <li>• Significant local use in Australia</li> <li>• Coined by the Brisbane City Council.</li> <li>• Used primarily in conference communications and in relation to Australian studies.</li> <li>• Use has diminished in recent years</li> <li>• "Quality" implies one goal is being achieved (water quality)</li> </ul>	Fletcher et al., 2015
SUWM	Sustainable urban water management	<ul style="list-style-type: none"> <li>• An internationally used term, in conjunction with the UN's push to address sustainable development for the future</li> <li>• Followed the World Commission on Environment and Development.</li> <li>• Goal is to manage the urban water cycle</li> <li>• Produce more benefit than more traditional, centralized approaches.</li> <li>• A more integrated approach for water supply, sewerage, and stormwater management.</li> </ul>	Larsen and Gujer 1997; Marlow et al., 2013

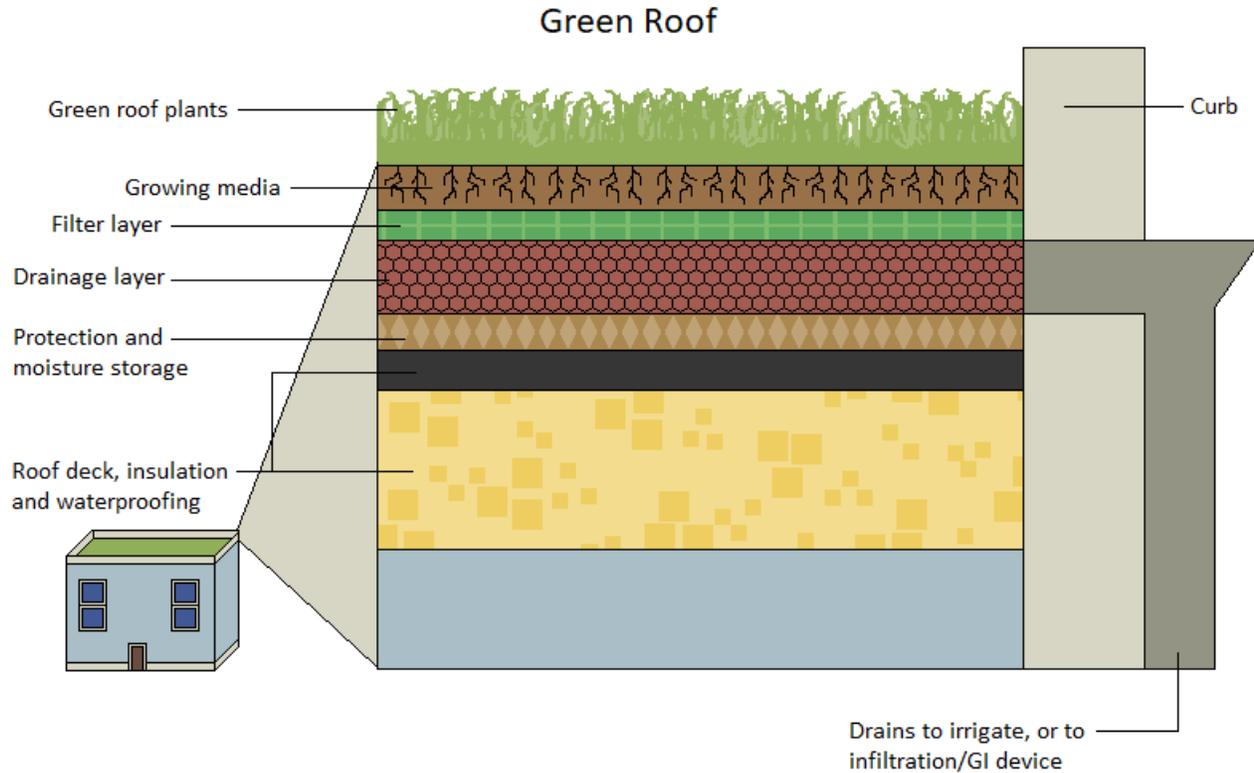
Eckart et al. (2017) defined LID as practices that retain water onsite and promote infiltration to protect surface water quality. Thus, predevelopment volume runoff is maintained through ground storage, infiltration, evapotranspiration, and reducing the peak flows (Eckart et al., 2017). There are two categories of stormwater management technologies that can reduce the impervious area of the watershed or area that is connected to the stormwater systems: retention and infiltration based technologies (Fletcher et al., 2013; Eckart et al., 2017).

Historically, GI was considered the green spaces in an urban area, now the terminology is used to refer to BMPs for wastewater/stormwater management (Berland et al., 2017). Green infrastructure can be used over a wide variety of watersheds including residential, commercial, industrial, car parks, roads and highways (Tedoldi et al., 2016). Green infrastructure generally has a more spatially distributed approach to stormwater treatment that treats the concentrations of contaminants in smaller hydrologic quantities before reaching end-of-pipe distribution, such as a stream, river, or reservoir. (Dietz, 2007) Traditional end-of-pipe stormwater often accumulates large masses of contaminants which can be prevented with more distributed green infrastructure.

Green stormwater infrastructure that encourages infiltration falls into two categories: surface infiltration and subsurface infiltration. The purpose of either category is to redirect any stormwater runoff from the surface into subsurface environments (Pitt et al., 1999). In surface infiltration, water is infiltrated from the surface thus mimicking natural ecosystem processes. Surface infiltration structures include bioretention, grass filters, constructed wetlands, and bioswales. Subsurface infiltration is when water is directly infiltrated into the vadose zone (subsurface), with or without pretreatment. Examples of subsurface infiltration are permeable pavement, french drains, soak-a-ways, and dry wells. Green roofs are not considered infiltration technologies, but their use for stormwater management can be used in conjunction with infiltration devices. The efficiency of these infiltration technologies is dependent on the local hydrogeological conditions and brief descriptions of some of the most common GI technologies are discussed below.

### **1.3.1 Green Roofs**

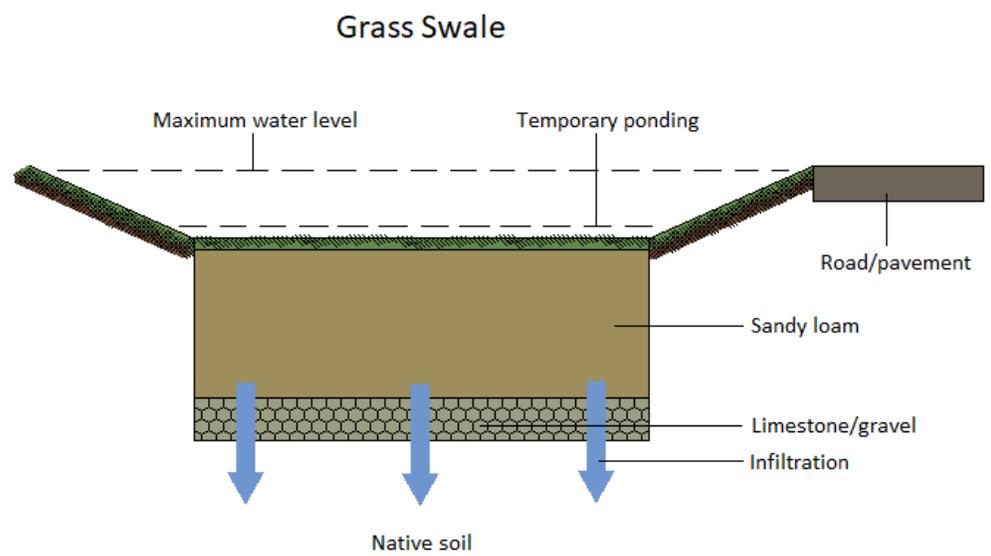
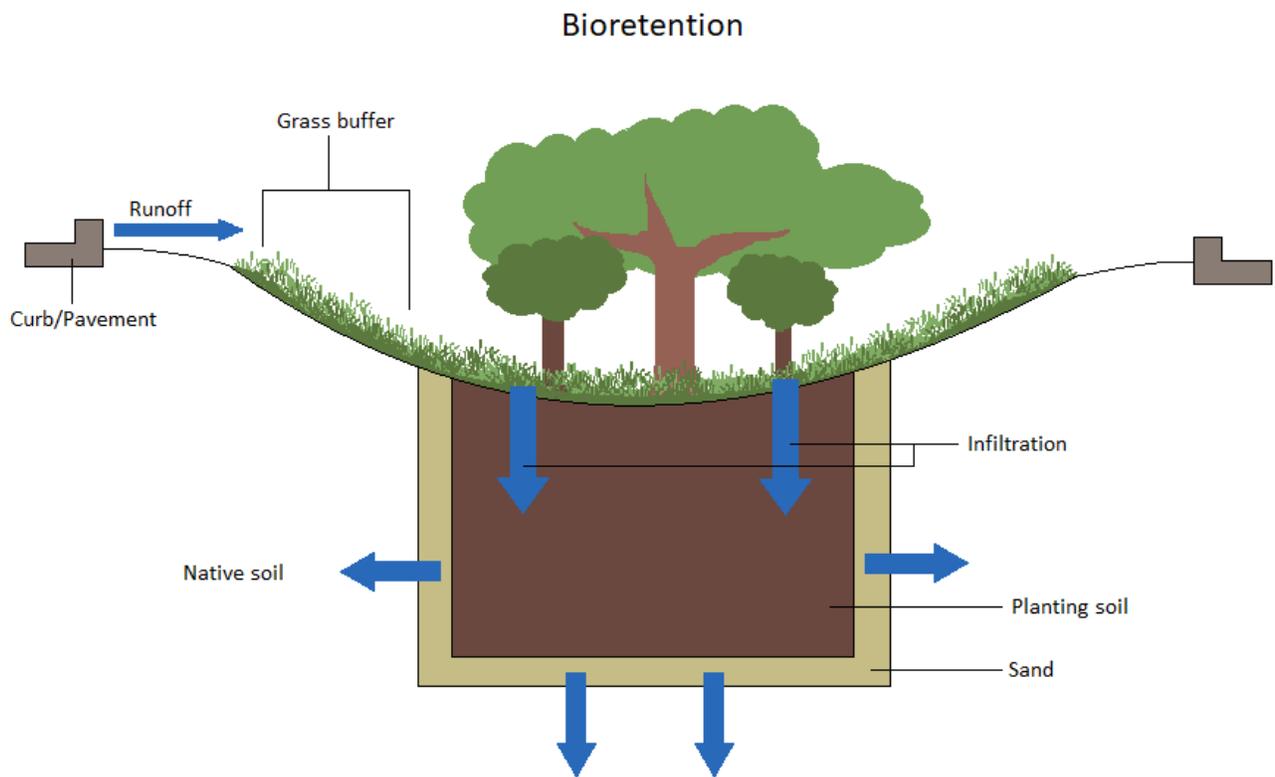
Green roofs are designed to collect the rainwater and retain it, thus slowing down the runoff from roofs (Figure 1). While they are designed to retain water and pollution, sometimes they can contribute to stormwater pollution through leaching from the materials and substrates used. Although green roofs are not designed to encourage infiltration into groundwater, the runoff from the roofs can become a part of the surface stormwater, thus contributing to the surface stormwater pollution. If groundwater infiltration systems are used alongside the green roofs, this could be a source of contamination. The contaminants of concern are nutrients and metals. Nutrient retention and leaching can come from the flushing of the substrates used on the roofs, from fertilizers, and from atmospheric deposition. Metals can come from leaching of roof materials and atmospheric deposition (Wang et al., 2017).



**Figure 1. A potential green roof system for the capture of precipitation.**

### **1.3.2 Bioretention or Swales**

Bioretention or swales are an urban stormwater practice where native ecosystems and landscape processes are used to enhance stormwater quality by capturing stormwater runoff from impervious surfaces (Figure 2). The processes are a combination of microbial processes, infiltration, evapotranspiration, and plants (Schueler and Holland 2000). The runoff is directed to a bioretention area from the impervious surface or occasionally a grass filter strip. These grass filter strips act as buffers that reduce velocities and filter particulates. After the grass filter strip, runoff is directed to a sand trench. This separates the planting bed from the impervious surface, thus augmenting the infiltration capacity of the planting bed, slowing the velocity, evenly distributing the incoming runoff, and facilitating flushing of the contaminants from the surrounding soil (Schueler and Holland 2000). The ponded water will eventually infiltrate through the organic top soil and to the groundwater or evaporate. This organic topsoil provides a medium for microorganisms and facilitates plant growth. Plant growth helps infiltrate runoff and absorb heavy metals, nutrients and hydrocarbons. Considerations when designing a bioretention system include the size of the drainage area, location, sizing guidelines, water budgets, nutrient removal capabilities, grading, elevations, soil amendments, organic layers and mulch amendments, planting concept used, design, number and size of plant material, and species, the plant growth and fertility, the surrounding land use and cover, local hydrogeology, and system maintenance (Schueler and Holland, 2000).



**Figure 2. Diagram demonstrating the features and water movement within a typical bioretention system and a typical grass swale system.**

### 1.3.3 Permeable Pavement

Permeable pavement can come in the form of pervious concrete, porous asphalt, or interlocking concrete pavers which can allow stormwater runoff to infiltrate into a temporary storage gallery (Figure 3). These systems can be used as parking lots, driveways, sidewalks, and low speed roads. The amount of rainfall that infiltrates through is dependent on the volume of storage below the system and how quickly the water infiltrates into the soil and groundwater below. These systems have the added benefit of converting typically impervious urban surfaces to permeable surfaces to better mimic natural hydrology. When designed properly, these systems can reduce runoff quantity, reduce total suspended solids (TSS) and total phosphorus (TP) from surface water runoff. Contaminant risks with permeable pavements can be from metals and oils from vehicular traffic.

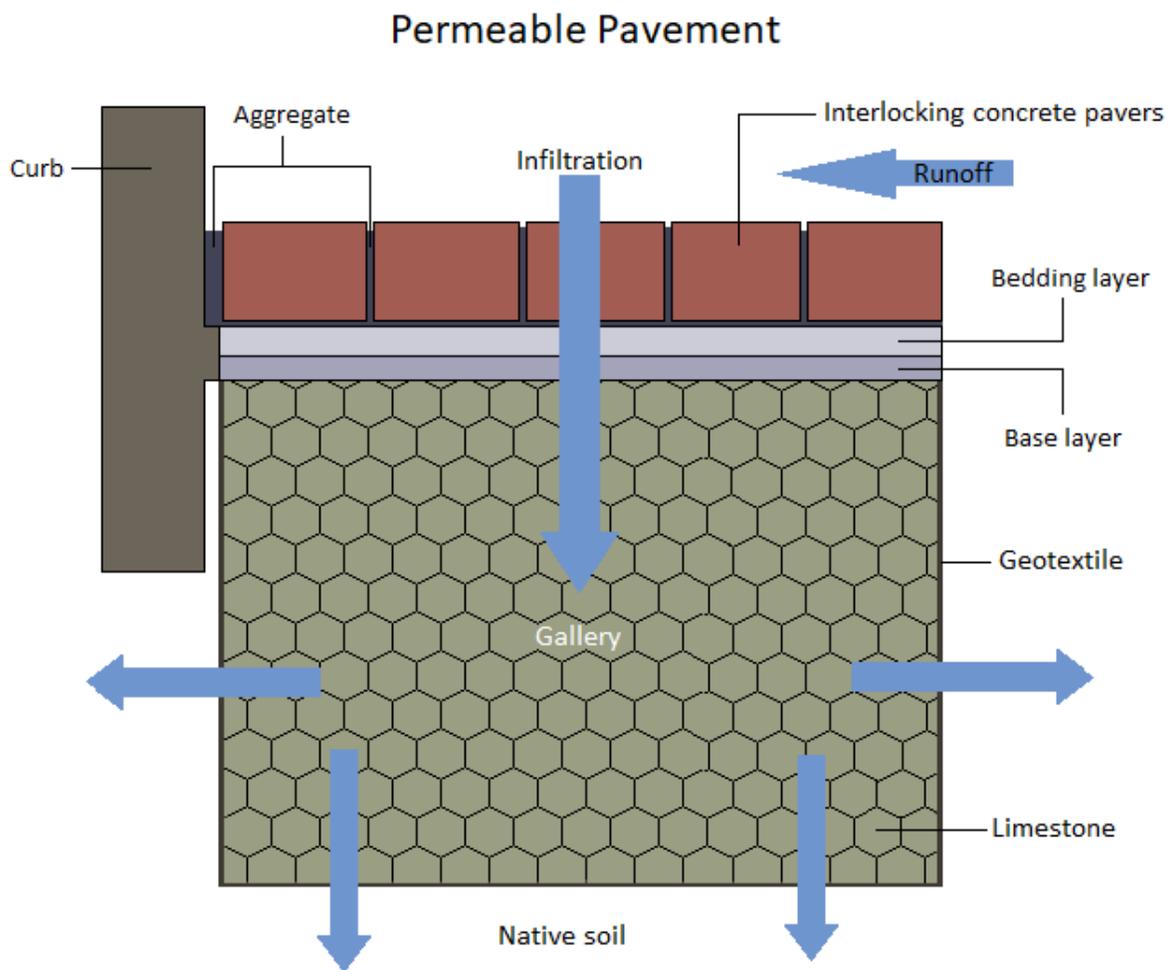


Figure 3. Model diagram of a permeable pavement system using interlocking pavers.

### 1.3.4 Dry Wells

Dry wells allow stormwater to be discharged to deep soil horizons or even directly into the groundwater. (Figure 4). Similar to subsurface devices such as infiltration trenches, but are narrow and take up less surface area space. This allows dry wells to be used to retrofit other infrastructure and/or use in conjunction with other infrastructure such as green roofs. These systems can be susceptible to clogging due to sediment and debris, so a sediment settling tank is sometimes used alongside these wells. Due to the direct infiltration deep into the soil, there is the risk of bypassing geochemical and infiltration mechanisms that can remove contaminants, putting the groundwater at greater risk for contamination.

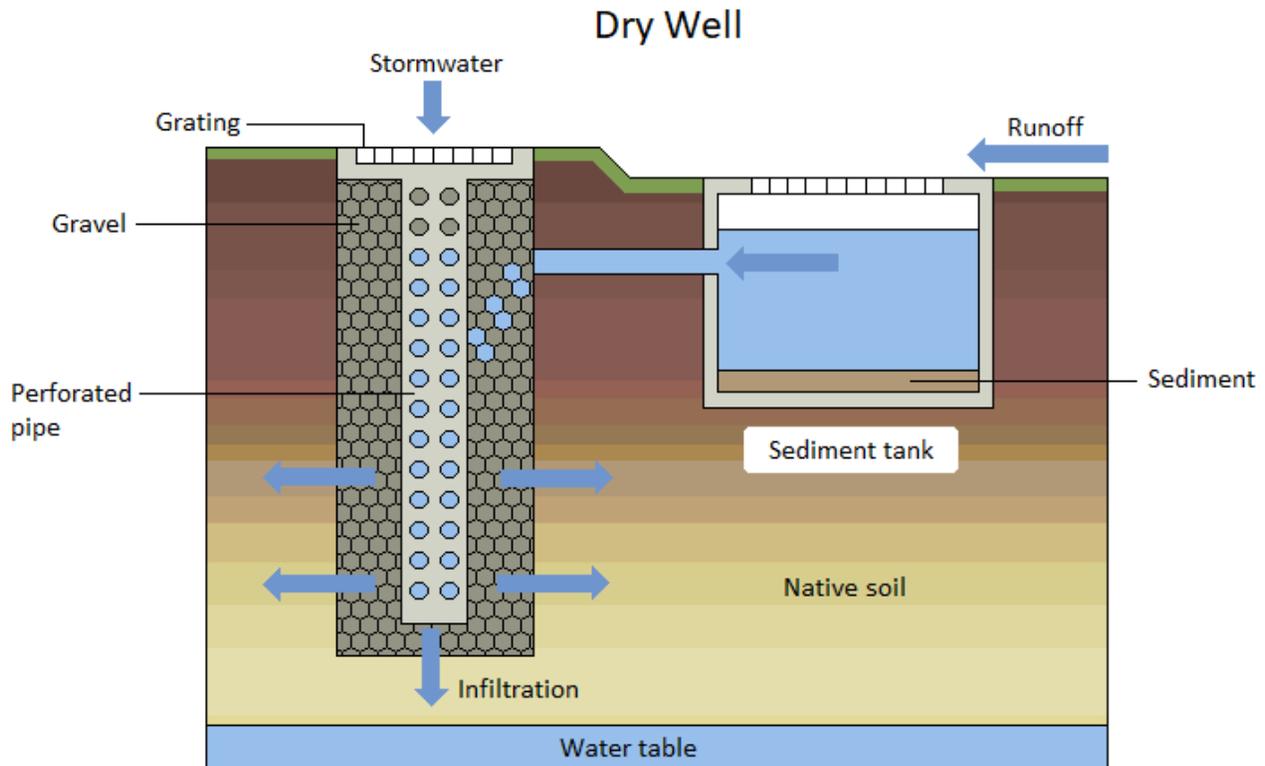


Figure 4. Diagram of a dry well showing the use of a sedimentation tank to collect sediments and debris, and the use of direct runoff collection.

## 1.4 General Information on Groundwater

Groundwater quality is of particular concern because ninety percent of the freshwater supply in the United States is groundwater, and it accounts for 20% of the total water usage in the United States (U.S.) (Michigan DEQ, 2015). Groundwater is used throughout the U.S. for public supply, individual wells, irrigation, livestock, aquaculture, industrial purposes, mining, and thermoelectric production (Maupin et al., 2014). Daily groundwater withdrawals in the U.S. amount to approximately 80 billion gallons of water (Michigan DEQ, 2015). Fifty-three percent of Americans drink groundwater, and 37% of the public water systems are dependent on more than 283,000 wells (Michigan DEQ, 2015). Fifteen million households are on private wells in the United States (U.S. Census Bureau, 2008).

Surface water and groundwater are hydrologically connected due to the geologic and hydraulic conditions that contribute to the flow between them, yet this connectivity is difficult to observe and measure (Winter et al., 1998; Sophocleous, 2002). When developing water management policies, these connections are often ignored, which can be significant because anthropogenic processes can affect these interactions (Winter et al., 1998). Groundwater naturally recharges through the infiltration of precipitation through pervious surfaces such as grasslands or forests (Pitt et al., 1999). In the past, infiltrating water was generally considered uncontaminated; urbanization has led to increased pollution in infiltrating stormwater as the amount of permeable soil surface is reduced. (Pitt et al., 1999) These hardscapes reduce the hydrologic sinks (infiltration, transpiration, etc.) that would occur in the non-urban pervious surfaces, leading to large volumes of runoff and flooding, sewer system malfunction, and impairment of surface and subsurface water resources (Berland et al., 2017). Sophocleous (2002) said water and chemical fluxes between groundwater and surface water are heterogeneous and of various scales, making them a challenge to quantify. The magnitude of the infiltration of surface water into groundwater (specifically channels) depends upon vadose zone hydraulic properties, storage volume in the vadose zone, channel geometry, wetted perimeter, flow duration and depth, antecedent soil moisture, clogging layers and water temperature. The extent of impervious surface, especially in urbanized areas, and permeability of recharge zones also contributes extensively to the magnitude of surface water infiltrating to groundwater.

The chemical interactions between surface water and groundwater are controlled by the type of geologic materials present and the time the water is in contact with these materials. The various chemical reactions that affect the biological and geochemical characteristics of the basin are acid-base reactions, precipitation and dissolution of minerals, sorption, ion exchange, oxidation-reduction reactions, biodegradation, and dissolution and exsolution of gases (Winter et al., 1998). When it comes to managing water resources, the tendency for contaminants to move between the ground and surface water needs to be considered (Winter et al., 1998). For example, evapotranspiration by plants removes water, but not the dissolved salts increasing the soil salinity, and if irrigation follows, the salts can be moved down gradient into the hydrologic systems (Winter et al., 1998). If ground and surface water can be treated as one resource, the fate and transport of contaminants can be addressed more effectively (Winter et al., 1998). It is suggested that more data collection is needed through the coordination between watershed management municipalities.

## **1.5 Contaminants**

### **1.5.1 Contaminants in Stormwater**

Nonpoint source pollution is a major part of the water quality problems in the U.S. waters. Urban stormwater runoff and runoff from construction sites is one source that can contribute to excess nutrients, bacteria, and toxic metals. (US EPA, 2002; Bedan and Clausen, 2009). Different watersheds can produce different pollutant loads generated by runoff (Tedoldi et al., 2016). Groundwater can be contaminated by many constituents: nutrients, metals, dissolved minerals, pesticides, other organics, and pathogens (Pitt et al., 1999). Sources of these contaminants include residues from automobiles, lawn treatments through fertilizers and pesticides, sewer overflows, and road deicing salts (Pitt et al., 1999).

Although only 3% of the U.S. groundwater has been contaminated, contamination has been found in all 50 states (US EPA, 2002; Michigan DEQ, 2015). The potential for groundwater contamination from stormwater is based on several factors: contaminant abundance in stormwater; contaminant mobility in the vadose zone; treatability of the contaminants; and the infiltration process used (Pitt et al., 1999). While groundwater can be polluted by natural sources, many anthropogenic activities such as agriculture

application (fertilizers, pesticides, animal manure), septic systems, leaking underground storage tanks, storm-water drains, improper disposal or storage of waste, and chemical spills at industrial sites can contribute to contamination (US EPA, 2002). Microbial contamination can come from human and animal waste, septic tanks, sanitary landfills, and garbage dumps being the major sources. Septic tanks and large concentrations of animals can also contribute to nitrate pollution. Concentrated animal feeding operations (CAFOs) where thousands of animals are raised in small spaces produce large amounts of animal wastes and manures that can contribute to pathogen and nutrient problems in the ground water. Manures can also contribute to high levels of salts.

### **1.5.2 Contaminants in Groundwater Receiving Infiltrated Stormwater**

Groundwater pollution can naturally come from a variety of sources: microorganisms, radionuclides, nitrates, nitrites, heavy metals, and fluoride (US EPA, 2002). Bacteria, viruses, parasites and other microorganisms can be found in groundwater, with shallow groundwater being at the greatest risk, due to runoff picking up these organisms from wildlife and soils (US EPA, 2002). Radionuclides can be naturally occurring radioactive elements such as radium (Ra) and uranium (U) that occur in the underlying rock (US EPA, 2002). Nitrates ( $\text{NO}_3^-$ ) are commonly from human activities, but are occasionally found naturally. They come from nitrogen compounds breaking down in the soil, and where the flowing groundwater picks them up (USEPA, 2002). Heavy metals can occur naturally in rocks and soils containing arsenic (As), cadmium (Cd), chromium (Cr), lead (Pb), and selenium (Se) (US EPA, 2002). Fluoride, although it is considered useful for dental health, can occur naturally in excessive amounts in some areas.

Jiang et al. (2015) assessed the validity of using harvested stormwater for local applications and risks from microbial contaminants. While this study did not look at groundwater infiltration, it did look at the possible health hazards from using stormwater that could be transferred to groundwater while using the various green stormwater infrastructure infiltration practices discussed in this review. Health hazards include pathogens which introduce risks due to a low dose of microorganisms needed to cause an infection, acute illnesses, risk of secondary transmission, and the potential for large scale outbreaks.

Heavy metals can come from mining and construction activities. Fertilizers and pesticides are used in agriculture, golf courses, and suburban lawns and gardens. These chemicals may end up in the groundwater depending on the types and amount of chemicals used, application process, and local environmental conditions. The fertilizers can be a source of nitrate pollution, breaking down in the soils (US EPA, 2002).

A series of four papers in 2017 assessed contaminants of emerging concern from 25 sources and drinking water treatment plants across the United States (Benson et al., 2017; Conley et al., 2017; Furlong et al., 2017; Glassmeyer et al., 2017). Three of those drinking water treatment plants had groundwater as their source serving between 50,000 and 500,000 people and watersheds of 800 km<sup>2</sup>. Their land usage was a mix of developed, agriculture, and non-developed with developed urban areas taking up most the landscape and agriculture the least. The contaminants assessed included pharmaceuticals, perfluoroalkyl and polyfluoroalkyl substances (PFASs), anthropogenic waste indicators, inorganic constituents, and microorganisms. Inorganics were the only analytes that showed significant reduction from the source for the drinking water treatment plant 24, which served the largest populations. Drinking water treatment plant 5 showed carbamazepine, a medication to treat epilepsy and neuropathic pain, in both the source and treated water (Glassmeyer et al., 2017).

## **1.6 Goal of Literature Review**

The goal of this literature review is to determine what research has been done on GI practices with respect to groundwater quality, and the risks and impacts to the subsurface environment and groundwater quality. All types of GI are assessed, including both surface and subsurface infiltration infrastructures. The literature assessed includes peer-reviewed literature, published reports, and conference proceedings, and provides insight into the current state of knowledge of the influence of GI on the subsurface environment and groundwater. This report points out research gaps to determine future needs for GI, including 1) contaminant risks that need further research 2) new infrastructure that has not been researched in depth and 3) determining local conditions when planning for green infrastructure.

## 2.0 What is Water Quality?

A simple definition of water quality is the chemical, physical, radiological, and biological characteristics of a water body or groundwater. This simplified definition does not consider any potential uses or standards that human activities may place on it, but it does provide a framework for determining standards for potential uses.

Most of the water on Earth is not toxic, but is also not potable. For example, 96% of the total surface water is seawater in the oceans (Eby, 2004). Since seawater is too salty to safely drink, it is not potable. Even if seawater is excluded, the remaining water is generally not toxic, but not potable because of other characteristics.

Surface and ground water quality is a complex topic and is fundamentally coupled to the geology and ecology of the earth and the waters intended use. For this reason, surface and ground water quality is often categorized based on its intended uses, such as water for human consumption; water for industrial and domestic uses; and environmental water quality. These water use categories in turn often have specific water quality standards or regulations associated with them.

### 2.1 Human Consumption

Water used for human consumption needs to be potable—that is, safe to drink or prepare food. Water quality in this case deals with the physical, chemical, biological, and radiologic characteristics that reduce the likelihood of injury or disease in humans. Characteristics that would cause injury or disease in humans are often called contaminants, or generically referred to as pollution. Many countries have established regulations or water quality standards to protect humans from harm due to water consumption. In the U.S., the U.S. Congress established the Safe Drinking Water Act that authorizes the U.S. Environmental Protection Agency to limit the amounts of certain contaminants in public water systems. Under the Safe Drinking Water Act, two basic standards were established: the National Primary Drinking Water Standards and the National Secondary Drinking Water Standards. The primary drinking water standards are regulated contaminants that affect human health. The primary drinking water standards are legally enforceable standards and the U.S. EPA enforces public water systems to comply with these standards. Secondary drinking water standards concern aesthetic characteristics of water that affect the odor, taste, and or appearance of the water. These secondary drinking water standards are not enforceable standards, and the U.S. EPA can only advocate their adoption. There is also the USEPA Unregulated Contaminant Monitoring Rule which collects data on contaminants that are present in drinking water, but set under the Safe Drinking Water Act (USEPA, 2017). Finally, it should be noted that bottled water safety is regulated by the U.S. Food and Drug Administration (21CFR129).

Some important water quality indicators for water that are to be used for human consumption are: alkalinity and hardness; pH; taste and odor; major anions and cations (calcium (Ca), magnesium (Mg), sodium (Na), potassium (K), bicarbonate ( $\text{HCO}_3^-$ ), carbonate ( $\text{CO}_3^{2-}$ ), chloride ( $\text{Cl}^-$ ), and sulfate ( $\text{SO}_4^{2-}$ )); trace metals (As, Pb, Cd, mercury (Hg), etc.) and trace anions (fluoride ( $\text{F}^-$ ), cyanide ( $\text{CN}^-$ ), iodide ( $\text{I}^-$ ), phosphate ( $\text{PO}_4^{3-}$ ),  $\text{NO}_3^-$ , etc.); dissolved organic carbon (DOC); pesticides; volatile organic compounds (VOC); semi-volatile organic compounds (SVOC); radiologic elements (radon (Rn), U, etc.); microorganisms (fecal coliform bacteria, *Cryptosporidium*, *Legionella*, etc.); pharmaceuticals; and personal care products to name a few. Some of these water quality indicators have regulatory limits associated with them, but it is important to note that many are not regulated. Even indicators that do not

have regulatory limits can potentially cause harm or are aesthetically displeasing and need to be considered in any discussion of water quality for human consumption.

## **2.2 Industrial and Domestic Uses**

Water destined for industrial or domestic uses other than direct human consumption does not necessarily need to be potable water. In these cases, water quality deals with the physical, chemical, biological, and radiologic characteristics that reduce the probability of damage to equipment and infrastructure. Important water quality indicators for industrial and domestic use include: total dissolved solids, hardness, pH, etc., and depend on the industrial process the water will be used for.

## **2.3 Environmental Water Quality**

Environmental water quality relates to water that may or may not be used for human activities, but is the water quality that is needed to sustain the ecosystem. Environmental water quality varies geographically and is related to the regional geology and ecology. Environmental water quality is the physical, chemical, biological, and radiologic characteristics that will ensure the protection of ecosystems. Like human consumption, environmental water quality is often regulated for non-drinking water purposes, protection of fisheries, protection of wildlife, and maintaining healthy ecosystems. For instance, in the U.S., one regulation that protects environmental water quality is the Clean Water Act.

Important water quality indicators for environmental water quality uses include: physical indicators (temperature, specific conductance (SPC), total suspended solids (TSS), turbidity, total dissolved solids (TDS), etc.), chemical indicators (pH, biochemical oxygen demand (BOD), chemical oxygen demand (COD), dissolved oxygen (DO), hardness, metals, nutrients, inorganic anions, organic compounds), and biological indicators (bacteria, fish, aquatic plants, insects, etc.). These water quality indicators in some cases can be considered contaminants, but in other cases these indicators will vary based on the ecosystem, local geology, and geography.

## **2.4 Stormwater Composition and Potential Contaminants**

There is a considerable body of literature describing the potential chemical components and potential contaminants in stormwater runoff. These potential chemical components and potential contaminants along with mobility factors are listed in Appendix 1, Table A1. The components or contaminants are both inorganic or organic substances. Inorganic substances include major anions, major cations, trace elements, and trace anions, etc. Organic substances include VOCs, SVOCs, pesticides, NOM, pharmaceuticals, etc. Tables A3 and A4, in Appendix 1, list some chemicals found in stormwater; provide chemical properties that could be used for determining the fate and transport of the chemicals in various environmental media; and, if available, regulatory concentrations for the chemical.

## **2.5 Naturally Occurring Contaminants**

When discussing potential contaminants to groundwater, it is important to recognize there are some contaminants that are part of the natural environment. These naturally occurring contaminants are mainly elements (As, Pb, zinc (Zn), etc.), inorganic compounds ( $\text{NO}_3^-$ , ammonia ( $\text{NH}_3$ ), sulfate ( $\text{SO}_4^{2-}$ ), etc.) or radionuclides (Rn, U, etc.). It is possible through mainly biological processes to produce organic substances (humic acids, fulvic acids, organic acids, etc.) that could be classified as naturally occurring contaminants if their concentrations are high enough to cause problems. Although naturally occurring contaminants may not pose a problem in ambient water, if the water characteristics change, the contaminants could be released and thus induce problems. The processes that could lead to the release of naturally occurring contaminants will be discussed in Section 6.

# 3.0 Hydrology

## 3.1 Hydrology

Hydrology is a very important aspect of GI. Since hydrology is not the focus of this review, a comprehensive review of the relationship of hydrology to GI will not be discussed. However, a basic discussion of hydrology will be presented below. Comprehensive reviews of hydrology can be found in Freeze and Cherry (1979), Domenico and Schwartz (1990), and Fetter (2001).

## 3.2 Hydrologic Cycle

The hydrologic cycle includes evaporation and precipitation at the surface; the ground surface will have overland flow (surface runoff) once reaching infiltration capacity, infiltration; the vadose zone with interflow, recharge, capillary rise, and evapotranspiration; and groundwater with evapotranspiration and base flow (Domenico and Schwartz, 1990; Freeze and Cherry, 1979; Fetter, 2001; Schirmer et al., 2013). Figure 5 is a diagram of the hydrologic cycle for natural environments, urban impact, and GI impact. The overland flow, interflow, and base flow feed surface water, and surface water can also recharge the groundwater. Both surface water and evapotranspiration can return water to the atmosphere, and the atmosphere returns water back to the ground and surface water via precipitation.

In an urban setting, the hydrologic cycle is interrupted. Before urbanization, groundwater was often recharged through rainfall, runoff, and snowmelt infiltrating through grasslands and wooded areas because of their permeable soils (Pitt et al., 1996). As urbanization reduces the pervious surface area available for infiltration, there is less groundwater recharge and more surface runoff (Pitt et al., 1996). This loss of soil water storage capacity reduces the base flows: the sum of the deep subsurface flow and delayed shallow subsurface flow (Walsh et al., 2012). The increased surface runoff can lead to higher peak discharge and volume, causing an increased frequency in nearby stream flooding (Konrad, 2003). Once an urban environment is developed, there can be increases in evapotranspiration, surface runoff, artificial interflow, artificial recharge, and base flow and groundwater withdrawals; there can be decreases in infiltration, transpiration, and natural recharge (Schirmer et al., 2013). Urban stormwater runoff brings in environmental flow problems such as excess water that leads to stream degradation, economic loss, and public health concerns (Walsh et al., 2012; Cizek et al., 2017).

### 3.2.1 Water Movement in the Subsurface

Groundwater can be vulnerable due to the range of hydrogeologic environments. This makes the subsurface highly complex and variable from the various geographic locations. It is outside the scope of this report to discuss these complex groundwater and subsurface interactions, but the reader is referred to the USEPA report (1985) that describes a method to evaluate the pollution potential of groundwater in different hydrogeologic settings across the United States.

Once precipitation strikes the earth's surface it can move downward into the soil column through infiltration, or after the upper soil zone is saturated and met its infiltration capacity, flow across the soil surface as runoff. The objective of green infrastructure is to reduce runoff and enhance infiltration. Infiltration will be the focus of this discussion.

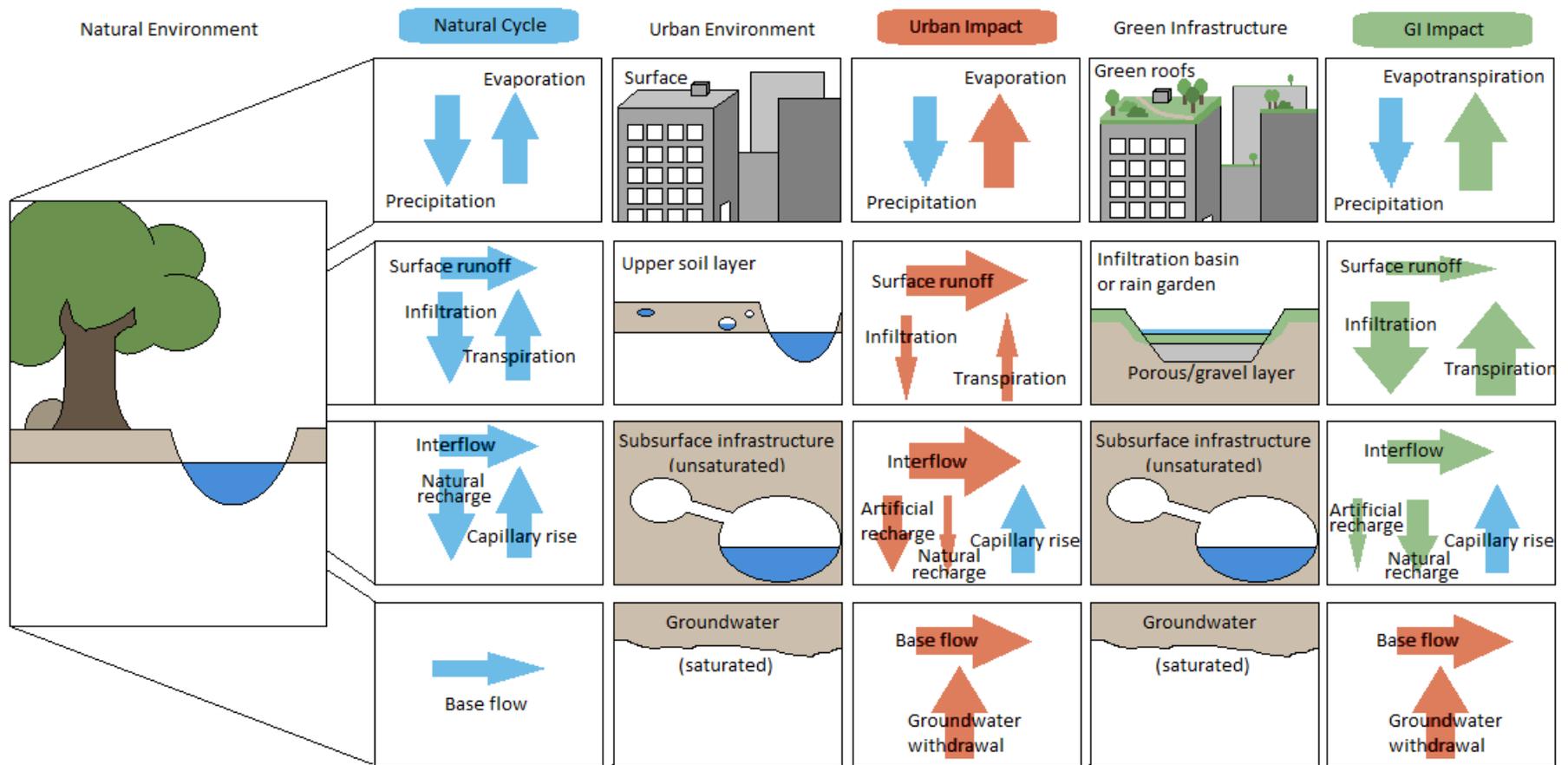
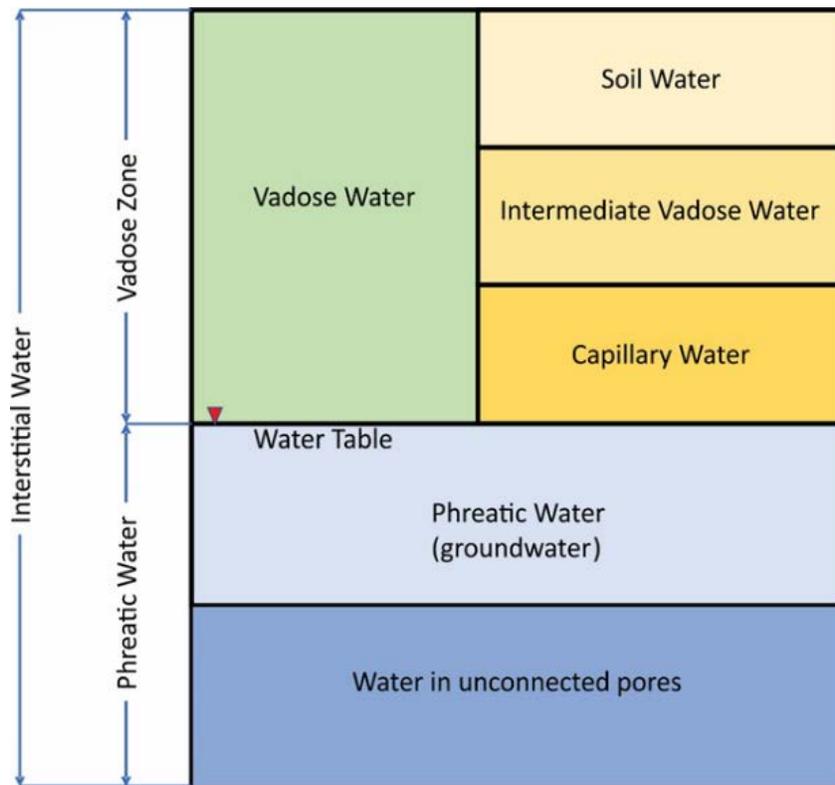


Figure 5. The hydrologic cycle for natural, urban, and green infrastructure systems. Arrows increasing or decreasing in size indicate change from previous hydrologic cycle. Color of arrow indicates whether it is part of the natural hydrologic cycle (blue), urban hydrology (red), or green infrastructure hydrology (green). (Modified from Schirmer et al., 2013).

Water that moves below the earth’s surface is called interstitial water (Freeze and Cherry, 1979; Domenico and Schwartz, 1990; Fetter, 2001). Interstitial water is defined as water in the pore space of soil, sediment, or rock (Freeze and Cherry, 1979; Domenico and Schwartz, 1990; Fetter, 2001). Interstitial water can be further divided as shown in Figure 6. These different zones have different functions and characteristics that are important in investigations of groundwater quality in GI systems.

The region of the soil/sediments (henceforth termed soil) which is partially filled with water, partially aerated, and above the water saturated soil (aquifer) is termed the vadose zone (Figure 6) (Freeze and Cherry, 1979; Domenico and Schwartz, 1990; Fetter, 2001; Brady and Weil, 2002; Eby, 2004). The vadose zone water can be further divided into soil water (water in the soil), intermediate vadose water (water between soil and the capillary fringe), and capillary water (water just above the water table) (Domenico and Schwartz, 1990).



**Figure 6. Subsurface water profile (modified from Domenico and Schwartz, 1990).**

As shown in Figure 6, the zone below the vadose zone, phreatic water, is where all the pores are saturated with water. Phreatic water can have two parts: phreatic water (groundwater) or water that is contained in connected pores. The water can move through the connected pores, but water in unconnected pores is not free to move (Freeze and Cherry, 1979; Domenico and Schwartz, 1990; Fetter, 2001). Phreatic water will be referred to as groundwater in this document and will be discussed in more detail below.

### **3.3 Vadose Zone**

The transport of water through the vadose zone, also known as the unsaturated zone, is a critical component of the hydrologic cycle, agriculture, natural resource protection, subsurface weathering, fate and transport of contaminants, fate and transport of other chemicals and nutrients, and ecological function (Domenico and Schwartz, 1990; Williams et al., 1998; Brady and Weil, 2002; Eby, 2004). The movement of water through the vadose zone is challenging under natural conditions because it is a dynamic process that changes markedly over time and space (Williams et al., 1998). The use of GI or other enhanced infiltration techniques could further confound the natural dynamic nature of water movement through the vadose zone.

The vadose zone is not only involved in the transport of inorganic, organic compounds, and microbes through the soil, but also acts as a sink for them (Brady and Weil, 2002; Eby, 2004). The biological and geochemical processes that occur in the vadose zone can degrade, transform, sequester, and/or retard the movement of chemicals (Brady and Weil, 2002; Eby, 2004). Therefore, it is essential to understand how chemicals behave and are transported in the vadose zone to understand groundwater quality changes that may occur as the result of infiltration.

#### **3.3.1 Infiltration**

Infiltration is the process of water entering the soil, which generally entails a downward movement of water into the vadose zone through all or part of the soil surface (Hillel, 1998; Williams et al., 1998; Brady and Weil, 2002). Knowledge of the infiltration process is prerequisite for managing soil water flux and the transport of contaminants in the vadose zone (Hillel, 1998; Williams et al., 1998). An abbreviated discussion on the infiltration process will be discussed below.

The infiltration rate is the volume flux of water that moves into the soil as a function of soil surface area (Hillel, 1998). However, soils do have a maximum infiltration rate, termed the infiltration capacity, and water supplied will either pond or runoff when the rate exceeds capacity (Hillel, 1998). Hillel (1998) states that when the infiltration rate is less than the infiltration capacity, water can penetrate the soil as fast as it is applied; in this case, the infiltration process is termed supply-controlled (or flux controlled).

The infiltration rate is initially high, gradually decreases, and eventually comes to a steady state (Hillel, 1998). Another way to think about this is the infiltration rate decreases until the soil reaches its infiltration capacity. Once the infiltration capacity is reached, the rate of infiltration tends to approach zero. The initial water content of the soil affects the suction gradient of the soil, so when comparing a dry soil to the same soil when wetter, the wetter soil will have a lower suction gradient (Hillel, 1998). This means that the wetter soil initially is, the faster it reaches a final infiltration rate. Soils with higher hydraulic conductivity will in general have higher infiltration capacity (Hillel, 1998). If the surface of the soil is porous (related to the soil texture) the infiltration will be greater, but is still limited by the infiltration capacity. Physical properties of the soil, such as compaction or dense surface zones can act as a barrier to infiltration (Hillel, 1998). Under dry soil conditions, hydrophobic organic substances coating soil particles can severely delay water infiltration (soil water repellency) and enhance runoff (Müller et al., 2018). If the soil surface or the soil layers are composed of fine-textured materials, this can impede the infiltration of water into the surface of the soil and slow the rate of water movement through the soil (Hillel, 1998). Coarse grained materials can, in some situations, also impede the flow of water through the soil layers.

Once the process of infiltration ends, the vertical movement of water does not cease (Hillel, 1998; Brady and Weil, 2002). Water will continue to penetrate further into the subsurface under the influence of gravity and suction gradients. This process will distribute water into different depths in the soil. The rate and time of this redistribution of water depends on the water storage capacity of the soil. Further discussions on the process of water movement in the vadose zone is beyond the scope of this effort.

### 3.3.2 Soil Texture

The size distribution of mineral particles found in the soil is referred to as soil texture (Brady and Weil, 2002). A common approach for describing soil texture is the United States Department of Agriculture (USDA) classification (Brady and Weil, 2002).

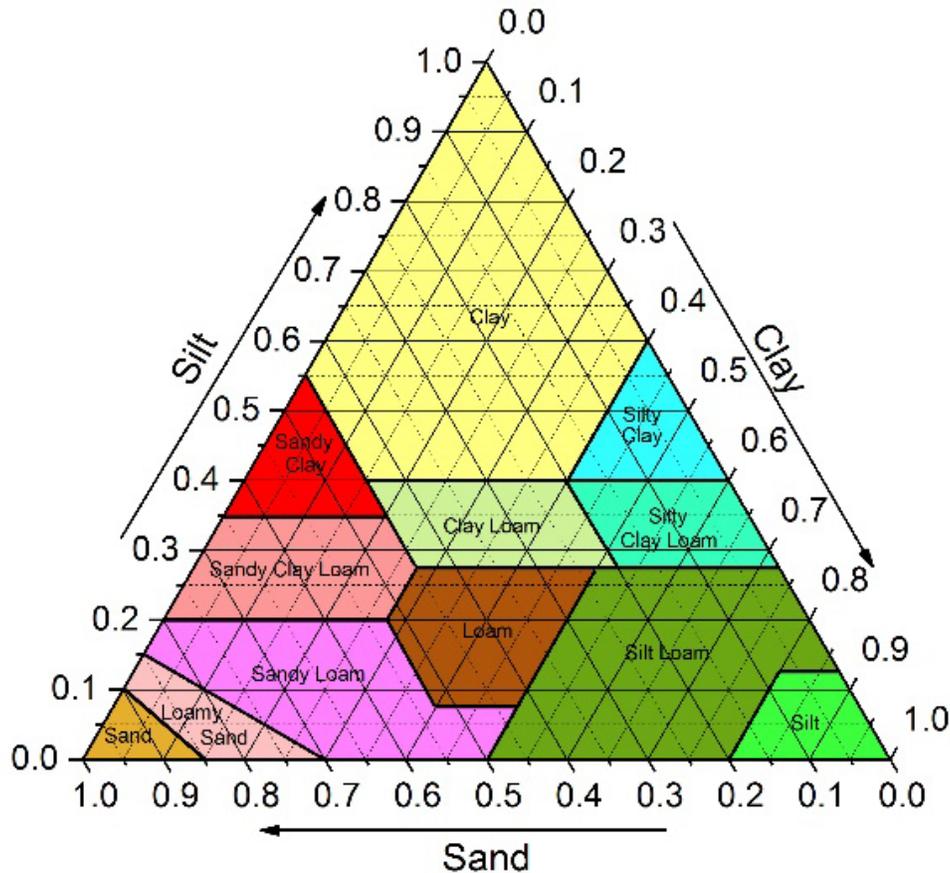


Figure 7. Soil textural classes showing the twelve USDA soil textural classes. (Figure modified from Brady and Weil [2002]).

The pure end members of the particle size classes rarely exist in the natural environment. For this reason, the USDA developed a soil textural class system, which contains 12 different textural classes (Brady and Weil, 2002). Figure 7 shows the major textural classes as defined by the percentages of sand, silt, and clay (Brady and Weil, 2002). Soils dominated by sand tend to have low water holding capacity, a high drainage rate, low decomposition of organic matter rates, low compactability, very low shrink swell capacity, and high leaching potential. Soils that are dominated by silts tend to have medium to high water holding capacity, slow to medium drainage rates, medium decomposition of organic matter rates, low compactability, low shrink swell capacity, and medium leaching potential. Finally, soils dominated by clays have high water holding capacity, slow to very slow drainage rates, slow organic matter decomposition rates, high compactability, moderate to very high shrink swell potential, and low leaching potential (unless cracks are present) (Brady and Weil, 2002).

### **3.3.3 Soil Structure**

Soil structure is the arrangement and organization of the primary particles in the soil (Brady and Weil, 2002; Hillel, 1998). Soil particles differ in size, shape, orientation, and the associations and interlinking between other soil particles. Soil particles typically form irregular patterns which can be impossible to characterize in geometric terms. Soil structure is unstable in nature and is not consistent in time and space (Hillel, 1998). Finally, soil structure is affected by biological activity, management practices, changes in climate, and is vulnerable to mechanical and chemical processes (Brady and Weil, 2002; Hillel, 1998).

The important aspect of soil structure in this discussion is soil pores. Soil pores are basically void spaces in the soil structure between particles that make up the soil structure (Hillel, 1998; Brady and Weil, 2002). The pore spaces can vary in size and shape which largely determine the role the soil pore plays (Hillel, 1998; Brady and Weil, 2002). Soil pores facilitate the transport of water, chemicals, and colloids through the soil profile. The size of soil pores can be classified into three basic categories: micropores, capillary pores (mesopores), and macropores (Hillel, 1998; Brady and Weil, 2002).

Micropores are one class of soil pores that are generally less than 0.001 mm in diameter (Hillel, 1998; Brady and Weil, 2002). Micropores are typically found in clayey soils. Water in these narrow pores is subject to adsorptive forces, and is often different than the water in wider pores (Hillel, 1998). Micropores, according to Brady and Weil (2002), retain water that plants can use, and can accommodate most bacteria.

Capillary pores are typically pores that range in size from a few micrometers to a few millimeters (Hillel, 1998). The water moving through capillary pores obeys the laws of capillarity and Darcy (Hillel, 1998). Brady and Weil (2002) refer to capillary pores as mesopores. These pores retain water after drainage (Hillel, 1998; Brady and Weil, 2002).

Macropores range in size from a few millimeters to centimeters, and are visible to the naked eye (Hillel, 1998; Brady and Weil, 2002). Examples of macropores are cracks and fissures in soil, decayed root channels, and earthworm burrows. Water travels through macropores under the influence of gravity, and the flow through macropores is rapid (Hillel, 1998; Brady and Weil, 2002). Macropores act as barriers to capillary flow when empty and can act as preferential flow pathways (Hillel, 1998).

Soil structure and soil pore size control water movement through the vadose zone (Brady and Weil, 2002; Hillel, 1998). In a sandy soil, the pore size is generally larger, allowing water to move more rapidly, whereas soil with a high clay content has a smaller pore size and slows the movement of water. Also, a well aggregated soil promotes high infiltration rates as there are more pores for water to flow through.

### **3.3.4 Soil Organic Matter**

Soil organic matter (SOM) is a term used to describe all the organic components of the soil (Sposito, 1989; Brady and Weil, 2002). Therefore, SOM includes the living biomass (living organisms), detritus (dead identifiable tissue), and humus (nonliving, nontissue) (Sposito, 1989; Brady and Weil, 2002). Brady and Weil (2002) further subdivide the humus portion of SOM into humic substances (humin, humic acids, and fulvic acids).

Soil organic matter (SOM) influences soil physical properties, soil chemical properties, and affects the biology and ecology of the system (Sposito, 1989; Brady and Weil, 2002). SOM influences many properties and a complete discussion of this topic is beyond the scope of this report. It is found that SOM helps to reduce the plasticity, cohesion, and stickiness of clayey soils. More importantly, SOM increases infiltration rates and water-holding capacity in soils. The increase allows water to more easily move through the vadose zone.

Soil organic matter in the vadose zone is important for many geochemical processes that occur such as contributing to ion exchange, sorption of metals, sorption of organic compounds, and soluble organic compounds are involved in chelation and complexation reactions in the soil. Other important chemical properties that have not been discussed are that SOM also provides a significant contribution to the pH buffering capacity in the vadose zone, and humic substances contribute to the weathering process occurring in the vadose zone (Brady and Weil, 2002). A final contribution of SOM is the formation of organic colloids, which will be discussed in Section 5.6.

### **3.4 Groundwater**

The transport of water through the saturated zone (phreatic zone) is an essential component of the hydrologic cycle, agriculture, natural resource protection, subsurface weathering, fate and transport of contaminants, fate and transport of other chemicals and nutrients, and ecological function (Freeze and Cherry, 1979; Domenico and Schwartz, 1990; Fetter, 2001; Brady and Weil, 2002; Eby, 2004;). The movement of water through the saturated zone is an active process that changes over time and space (Freeze and Cherry, 1979; Fetter, 2001; Domenico and Schwartz, 1990;). The use of GI or other enhanced infiltration techniques could cause changes to the movement of water through the aquifer. The movement of groundwater is important for understanding potential impacts to groundwater quality that could result from GI practices. Therefore, a brief introduction to groundwater hydrology will be provided, and more detailed discussions on ground water hydrology can be found in Freeze and Cherry (1979), Domenico and Schwartz (1990) and Fetter (2001).

#### **3.4.1 Porosity and Permeability**

Porosity is defined as the void spaces in a rock or unconsolidated material (Freeze and Cherry, 1979; Domenico and Schwartz, 1990; Fetter, 2001). Equation 1 defines the relationship between total porosity ( $n$ ), the void volume ( $V_v$ ), and the total volume ( $V_T$ ) as a percentage.

$$n = \frac{V_v}{V_T} \quad \text{Equation 1}$$

It is important to understand that total porosity considers both the connected pores and the unconnected pores. Since the unconnected pores do not contribute to the movement of water through the saturated zone, total porosity is not necessarily the best parameter to use in the discussion of water movement through the saturated zone. The effective porosity, which only considers connected pores, is a more appropriate parameter to use when discussing the flow of water in the saturated zone.

Permeability can be defined as the ease by which water can move through a porous media, and is quantified by a rate of appropriated units (Freeze and Cherry, 1979; Domenico and Schwartz, 1990; Fetter, 2001). The primary importance of permeability is understanding the zones within a porous media in which water can flow freely, and zones that impede the movement of water through them. Low permeable media are sometimes referred to as aquitards (Domenico and Schwartz, 1990). Aquitards are

sometimes referred to as confining layers, as they do not allow free movement between different zones in the aquifer.

### 3.4.2 Saturated Groundwater Flow

In saturated sediments or rock, to get water to move requires energy (Freeze and Cherry, 1979; Price, 1985; Domenico and Schwartz, 1990; Fetter, 2001). In a groundwater system, this energy is typically supplied through hydraulic head or head (Freeze and Cherry, 1979; Price, 1985; Domenico and Schwartz, 1990; Fetter, 2001). Head is the distance water rises in a column in relationship to a fixed point (Freeze and Cherry, 1979; Price, 1985; Domenico and Schwartz, 1990; Fetter, 2001). The difference in heads between two measurement points separated by a fixed distance (also called the hydraulic gradient) provides the energy that drives groundwater flow. The hydraulic gradient ( $i$ ) is shown mathematically in equation 2.

$$i = \frac{dh}{dL} = \frac{h_1 - h_2}{L} \quad \text{Equation 2}$$

Where  $h_1$  is the head at point 1,  $h_2$  is the head at point 2, and  $L$  is the distance between point 1 and point 2.

As was discussed earlier, the flow through a saturated porous system is related to permeability, i.e., how easy or hard it is to move water through pore spaces in the media. The permeability measure in groundwater flow systems is called the hydraulic conductivity ( $K$ ) (Freeze and Cherry, 1979; Price, 1985; Domenico and Schwartz, 1990; Fetter, 2001). The more permeable the aquifer is, the higher the  $K$ .

In a saturated system, the groundwater flow rate ( $Q$ ) through a cross sectional area ( $A$ ) in the aquifer is described by Darcy's Law (Freeze and Cherry, 1979; Price, 1985; Domenico and Schwartz, 1990; Fetter, 2001). Equations 3 and 4 demonstrate two different expressions of Darcy's Law.

$$Q = KiA \quad \text{Equation 3}$$

$$Q = KA \frac{h_1 - h_2}{L} \quad \text{Equation 4}$$

### 3.4.3 Surface Water – Groundwater Connection

Surface water and groundwater are usually connected as a localized process at the sediment/water interface (Freeze and Cherry, 1979; Price, 1985; Domenico and Schwartz, 1990; Fetter, 2001; Hayashi and Rosenberry, 2002; Parsons et al., 2004;). It is important to understand this connectivity as it relates to the hydrology of GI systems and its potential to influence groundwater quality. As was shown in Figure 5, water that enters the subsurface can be returned to surface water by interflow and base flow.

With respect to water in the vadose zone, it is possible that not all the water that enters is transported to the aquifer. If the water reaches a layer that is impermeable, or if there are layers in the vadose zone of contrasting permeabilities, the water may flow laterally and enter a surface water body. This type of flow is known as interflow (Freeze and Cherry, 1979; Price, 1985; Domenico and Schwartz, 1990), whereas groundwater that flows from the saturated zone into a surface water body is termed base flow (Freeze and Cherry, 1979; Price, 1985; Domenico and Schwartz, 1990).

### 3.5 Hydrology and GI

Urban water quantity problems are strongly linked to urban development (Schirmer et al., 2013). Urban areas have historically relied on groundwater from springs and shallow wells for potable water, but accelerated urbanization has increasing demands and groundwater supplies are declining (Schirmer et al., 2013). The over-exploitation of groundwater under urban areas has led to declining water levels, possible land subsidence, and salt water intrusion in coastal cities (Schirmer et al., 2013).

The goal of GI is to return urban hydrology back to a more natural hydrologic cycle by using retention and infiltration techniques (Eckart et al., 2017). There are two types of infiltration used in GI: surface infiltration and subsurface infiltration. Surface infiltration allows the passage of stormwater directly through the soil surface into the vadose zone. Subsurface infiltration directs the stormwater directly into the deeper horizons and sometimes directly into the aquifer. The infiltration based techniques can recharge and restore the groundwater baseflows, although these can be dependent on the site conditions that influence the performance of the infrastructure (Eckart et al., 2017). Retention based technologies, such as wetlands and ponds, can reduce the peak flows by storing the stormwater and reducing volume through evapotranspiration and percolation into the vadose zone, but this does increase the duration of the flows (Eckart et al., 2017).

Concerns with GI include: fluctuations in groundwater levels, limitations with large precipitation events, clogging, and soil limitations. Increased groundwater levels due to mounding have been observed in previous stream restoration projects, bioretention cells, and regenerative stormwater conveyance systems (Hammersmark et al., 2008; Endreny and Collins, 2009; Cizek et al., 2017); though, mounding usually only occurs for short periods of time after precipitation events (Cizek et al., 2017). For example, retention basins can recharge and potentially raise the water table. Additional research is needed to address the biological and geochemical processes that uniquely occur in the vadose zone when mounding is present. This can adversely impact subsurface infrastructure, and thus negate the benefit from using green infrastructure to naturalize the urban hydrology (Endreny and Collins, 2009). Various types of green infrastructure function on different hydrologies. Simulations have shown that physical properties of the soils and soil depths are important for the infiltration and surface runoff processes (Xiao et al., 2007; Eckart et al., 2017;). Rainfall volume, duration, and time to peak ratio can impact the performance of green infrastructure such as grassed swales, green roofs, permeable pavement and bioretention cells such that the larger the precipitation event, the less effective this infrastructure can be (Hunt et al., 2008; Qin et al., 2013; Eckart et al., 2017). Hydrology of infiltration is dependent on the clogging rate of the infrastructure. Line et al. (2012) found there was eventual clogging in bioretention cells that prevented infiltration, and Bergman et al. (2011) saw decreases in infiltration rates due to clogging by fine particles in two infiltration trenches over 15 years.

# 4.0 Biology

## 4.1 Microbiology

### 4.1.1 Types of Microbial Contaminants

Microbial contaminants within stormwater include a range of organisms spanning Eukaryota, Bacteria, and viruses originating from various sources discussed below. Waterborne disease, especially acute gastrointestinal illness (AGI), arising from the consumption of contaminated groundwater as drinking water, is an important potential public health risk that may be influenced by stormwater infiltration practices. Epidemiological studies have shown that untreated groundwater was the source of ~30% of waterborne disease outbreaks in the U.S. and The Freely Associated States (reported in the Waterborne Disease Outbreak Surveillance System (WBDOSS)) between 1971 and 2006 (Craun et al., 2010) and numerous more cases worldwide (Murphy et al., 2017). Within the WBDOSS reported groundwater-associated outbreaks the primary etiological agents were Bacteria (~20%), viruses (~13%), and parasites (~8%) (Wallender et al., 2014) and many reports of pathogen presence in groundwater systems under outbreak and non-outbreak conditions globally (Murphy et al., 2017).

**Table 2. Some organisms of public health concern in groundwater used for drinking and associated incubation periods for infection after exposure. Organisms of interest and incubation periods reproduced from (Moe, 2007).**

Agent	Incubation Period (days)
Bacteria	
<i>Campylobacter</i> spp.	3 - 5
<i>Escherichia coli</i> (pathogenic strains)	<1 - 6
<i>Salmonella</i> spp.	<1 - 28
<i>Shigella</i> spp.	1 - 7
<i>Vibrio cholerae</i>	<1 - 5
<i>Yersinia</i> spp.	2 - 7
Protozoa	
<i>Cryptosporidium parvum</i>	7 - 14
<i>Entamoeba histolytica</i>	14 - 28
<i>Giardia lamblia</i> (intestinalis)	5 - 25
Viruses	
Astrovirus	1 - 4
Enterovirus	3 - 14
Hepatitis A virus	15 - 50
Hepatitis E virus	15 - 65
Norovirus	1 - 3
Rotavirus	1 - 3

Reliable recovery and enumeration of some of the pathogens listed in the Table 2 is difficult using traditional cultivation-based methodologies such as Most Probable Number (MPN) or Membrane Filtration (MF) with reasonable sample sizes. For this reason, public health officials have been assaying much more easily cultivated “indicator organisms” for over a century to quantify the potential of public health-related microbial contamination of water sources. Ideal attributes of indicator organisms have been reviewed in detail by the National Research Council as well as comparisons of enumeration methods targeting indicator organisms or the specific pathogens of interest (National Research Council, 2004). Briefly, indicator agents should be present in higher populations than the relevant pathogen within the original source material and correlated with a health risk. Additionally, the indicator should behave (die-off, filtration, sedimentation, biological survival mechanisms, etc.) similarly to the pathogen under environmental selection pressures. Fecal indicators such as fecal coliforms, *Enterobacteriaceae* spp., and *E. coli* for enteric bacteria and F+RNA coliphage for enteric viruses are routinely quantified from water samples for public-health-risk testing. The presence of these indicator agents is indicative of fecal contamination and correlated with the presence of enteric pathogens and disease risk. With the development and adoption of molecular biology methods, more sensitive assays for the direct enumeration of enteric pathogens and host-specific microflora have allowed for better water quality monitoring and identification of pollution source through microbial source tracking (National Research Council, 2004). Despite these advances, the cultivation-based MPN and MF methods remain the “gold standard” accepted procedure for public health monitoring of microbial water quality.

#### **4.1.2 Sources of Microbial Contaminants**

Previous studies have demonstrated the high microbial loads of stormwater from a variety of settings including agricultural, residential, and industrial areas (Geldreich et al., 1968; Mallard, 1980; Page et al., 2012; Sidhu et al., 2013). Stormwater runoff microbial populations vary between any given storm event or location based on numerous factors such as: storm intensity, land use practices within the stormflow area, seasonal/environmental conditions, etc.; however, unified principles in water quality considerations and human health risks have been obtained through years of investigation (Mallard, 1980; Selvakumar and Borst, 2006; McCarthy et al., 2008; Page et al., 2016).

A previous study of stormwater collected from various land use settings (urban streets, suburban business district, and rural) identified contact with soils and impervious surfaces as the primary source of biological pollutant densities in stormwater runoff (Geldreich et al., 1968). Rainwater was shown to generally harbor very few pollution indicator organisms (<1/100 mL) with the uncommon detection of fecal coliforms/Enterococci (<2/100 mL) found to be associated with particulates from soils, insects, or vegetation (Geldreich et al., 1968).

The impact of livestock on fecal contamination of agricultural runoff is well known, but warm-blooded animal fecal material is a key source of stormwater runoff pollution within urban and suburban settings as well. Microbial pollutants have been quantitatively and taxonomically linked to human fecal material from combined sewage system overflow (Sauer et al., 2011; Sidhu et al. 2012; Sidhu et al., 2013), as well as the presence of rodents, dogs, cats, and birds within city stormwater runoff events (Geldreich et al., 1968; Mallard, 1980). Fecal coliforms have been observed at concentrations on the order of  $10^3$  –  $10^5$  CFUs/mL in urban stormwater with values varying across seasons and land use types (e.g., commercial/industrial, low-density residential, high-density residential) (Geldreich et al., 1968; Maestre and Pitt, 2005; Selvakumar and Borst, 2006; Page et al., 2016). In each study, the highest fecal coliform loads were found in high-density residential areas.

Upon entering stormwater flow, microbial contaminants have been shown to be primarily adsorbed to particulate material within the stormwater rather than existing as planktonic cells and communities (Characklis et al., 2005). There are very few studies on the dynamics of indicator organisms or pathogens within stormwater, but the main factors positively influencing removal have been identified as duration within the surface aquatic environment (time), temperature, and sunlight exposure (Geldreich et al., 1968; Mallard, 1980; Selvakumar et al., 2007). More detailed survival studies by Selvakumar and Borst (2006) illustrated that exposure to sunlight resulted in significantly greater, though organism-specific, reduction rates of indicator organisms and pathogens in stormwater. Based upon these findings, the authors recommended that BMP for urban stormwater handling emphasize engineering solutions to extend the lag time between surface runoff and stormwater introduction to receiving waters while ensuring greater sunlight exposure times to maximize reduction of microbial pollutants within stormwater (Selvakumar et al., 2007).

#### **4.1.3 Fate and Transport of Microbial Contaminants in Porous Media**

Fecal-associated microorganisms are adapted for optimal growth in a habitat characterized by stable moderate temperature (host body temperature, ~37°C) and high concentrations of labile organic matter (e.g. ingested food boluses, intestinal mucosa). Groundwater as a habitat is very different than these conditions with temperatures typically ranging from 5 - 25°C and limited labile organic matter pools. While gut-associated microbial contaminants are not expected to grow and thrive within the groundwater environment, their rates of removal are affected by several, often interdependent, environmental factors.

Overall, research has shown there is a general trend of differential survival for the various contaminant organism types. Viruses tend to have the longest persistence times within any groundwater environment, with some observed residual infectivity times of months to years (Charles et al., 2009; Regnery et al., 2017). Enteric eukaryotes (*Cryptosporidium* spp. and *Giardia* spp.) and enteric bacteria typically have die-off rates of five to ten times, and over one hundred times greater than enteric viruses, respectively (John and Rose, 2005; Regnery et al., 2017). Within this general trend, physiological differences between species and even strains of each type of organism result in variations in observed pathogen removal rates. Pathogen removal or die-off rates are typically reported based upon first order decay models; however, field and laboratory experiments have shown that biphasic models better approximate the removal behavior of fecal eukaryotes and viruses within groundwater systems (Charles et al., 2009; Toze et al., 2010). These studies have shown there is an initial rapid removal phase for the first few days after introduction, followed by a slower phase two to hundreds of times less than the initial phase that can lead to months or years of persistence. This biphasic behavior is likely due to a smaller subpopulation of the contaminant organism that is more resistant to the unfavorable environmental conditions allowing for greater persistence.

#### **4.1.4 Saturated Zones**

Temperature is the most studied, and one of the most important environmental factors influencing pathogen survival in groundwater, since all chemical and biological processes are influenced by temperature. Reviews compiling the data of numerous studies evaluating the relationship between temperature and pathogen survival have shown that across system type, pathogen survival time decreases with increasing groundwater temperature. (John and Rose, 2005; Foppen and Schijven, 2006; Regnery et al., 2017).

Aquifer hydrogeology can affect the physical removal of microbial contaminants from groundwater through colloid adsorption and size-exclusion processes (Foppen and Schijven, 2006). Filtration is a physicochemical process wherein the contaminant organisms adsorb to the surfaces of the aquifer matrix. This process is dependent upon several chemical properties, namely groundwater chemistry (ionic strength (IS) and pH) as well as cell surface and viral capsid properties such as charge, isoelectric point, and hydrophobicity (Harvey and Ryan, 2004). Adsorption to solid media surfaces can extend the persistence of enteric bacteria and viruses within the subsurface 10 to 100 times longer relative to survival times suspended in groundwater (Hurst et al., 1980; Pachepsky and Shelton, 2011). This removal is reversible depending upon groundwater chemistry (pH, IS, and salt types) and flow. Several previous studies have documented increased colloid retention within porous media with increasing IS and/or increases in cation valence number (e.g. Ca vs. Na) (Fontes et al., 1991; Grolimund et al., 1998; Redman et al., 1999; Knappett et al., 2008; Zhuang and Jin, 2003). Similarly, the rapid infiltration of precipitation has been shown to promote the release of adsorbed organisms back into the groundwater due to increased hydrodynamic shear and decreased IS (Johnson et al., Assemi, 2007; Shen et al., 2007). In contrast, physical straining is an irreversible process by which contaminant organisms become entrapped in pore spaces too narrow to allow transport, removing them from the downstream flow (McDowell-Boyer et al., 1986). Unlike filtration, straining is dependent only on pore space geomorphology and thus cells are not released by changes in groundwater chemistry or flow. Both filtration and straining affect the retention and removal of microbial contaminants within saturated porous media systems. Groundwater chemistry and the ratio of colloid (microbial contaminant) size to grain size are important factors for determining the contributions of each process to the overall retention and removal.

The indigenous microorganisms within groundwater habitats play a significant role in the removal of introduced enteric pathogens (Banning et al., 2002; Gordon and Toze, 2003). This antagonism by native microflora is from a combination of competition for nutrients and predation. Introduced viruses are removed by grazing protozoa (Pinheiro et al., 2007) and proteolytic exoenzymes that degrade the capsids of enteroviruses secreted by common resident heterotrophic bacteria (e.g. *Pseudomonas* spp.) (Cliver and Herrmann, 1972). Inhibition of these heterotrophic bacteria by sterilization or by chemical inhibition of the aerobic respiratory chain dramatically reduce virus decay rates (Elliott et al., 2011a; Gordon and Toze, 2003). As mentioned earlier, increasing temperature significantly increases the rate of pathogen removal from groundwater systems. Microbial metabolic activity increases with increasing temperature up to maximum viable temperature of a specific organism. Thus, temperature plays a direct and an indirect role in the removal of pathogenic organisms and viruses within these systems. Gordon and Toze (2003) documented maximum virus inactivation rates under conditions of available dissolved oxygen at 28°C and low exogenous nutrient input (Gordon and Toze, 2003), conditions that would stimulate maximum exoenzyme activity for metabolically active resident heterotrophic microorganisms.

#### **4.1.5 Unsaturated Zones**

The removal processes described for saturated conditions (mechanical filtration, straining, wedging, and adsorption) apply to unsaturated zones as well. The presence of an air phase within unsaturated zones creates two additional interfaces (air-water and air-sediment) not encountered under saturated conditions that affect microbial contaminant attachment and removal. Specifically, microorganisms can become adsorbed to grain surfaces at the air-water interface due to capillary action, and entrapment at the air-sediment phase when the wetted grain surface layer becomes thinner than the organism (free-living or particle-associated). As the moisture content decreases within the unsaturated medium, microorganisms are subject to die-off and/or inactivation through desiccation. As flow is established through transient

flushing events, these attached microorganisms and those entrained in small disconnected water bodies can be released back into the bulk flow through the system (Rippy, 2015; Peng et al., 2016).

Unsaturated porous media can be subdivided into the rooting zone and the vadose zone below the rooting zone. The rooting zone of a soil profile contains the highest native microbial populations, greatest community diversity (mesofauna, protozoa, fungi, bacteria, archaea, viruses, etc.) and overall metabolic activity (Fierer et al., 2003). As stormwater infiltrates through vegetated soil layers, the microbial pollutants are subjected to competition for space and nutrients by the resident organisms that are specifically adapted to living within this habitat. In addition, they face selection pressure from plant-derived root exudates that select for beneficial resident microflora.

Below the rooting zone, resident microbial populations and activity decrease rapidly with increasing depth. In this area of the vadose zone, biological antagonism is greatly reduced as a selection pressure, and the physicochemical processes discussed above for desiccation and biocolloid transport exert the greatest influence on microbial contaminant removal.

#### **4.1.6 Microbial Methods**

During the past few decades there has been rapid advancement of molecular microbiology methods for identification and enumeration of microorganisms within environmental samples independent of cultivation. These methods have been used to study the gut microflora of humans and other warm-blooded animals to identify host-specific taxa for use in contamination source identification in Microbial Source Tracking (Santo-Domingo et al., 2010; Harwood et al., 2014; McLellan and Eren, 2014; Tan et al., 2015). Recently, studies employing these methods have evaluated the detection and cooccurrence of traditional fecal indicator bacteria and their associated pathogens within contaminated environmental samples. Some studies have found low correlation values for single organism indicator – pathogen relationships indicating that currently accepted public health monitoring strategies may not be sufficient (Harwood et al., 2005; Payment and Locas, 2011; McLellan and Eren, 2014).

## **4.2 Macrobiology**

Macrobiological organisms can be used to enhance green infrastructure, but can also cause complications. Vegetation is commonly used in green infrastructure for various reasons: nutrient and metal retention, enhance ecosystem services, increase infiltration, and mimic natural hydrology.

Trees are frequently studied for use in green infrastructure systems, providing stormwater control by canopy interception loss, evapotranspiration and infiltration (Berland et al., 2017). Canopy interception is the water that is stored in the tree canopies, and the volume of water stored varies depending on the tree's size, health, species, age, leaf area, canopy shape, and leaf/branch angles. Eventually this water evaporates from the canopy (Berland et al., 2017). Canopy interception in various forests can reduce 18 to 45% of the precipitation that reaches the ground, consequently reducing the volume of stormwater runoff, soil erosion and contaminant washout (Berland et al., 2017). Evapotranspiration is the combined loss of water from the soil through evaporation, and water taken up by plants and transpired to the atmosphere through plant leaf surfaces (Berland et al., 2017). Macrophytes assimilate contaminants into their tissues and sequester or transform the contaminants (Maine et al., 2007). By increasing the rhizosphere diversity in the subsurface environment, various chemical and biochemical reactions can occur (Jenssen et al., 1993; Maine et al., 2007). *Eichhornia crassipes* (Mart.) Solms. is a fast-growing species known to take up nutrients and contaminants (Tchobanoglous et al., 1989; Vesk and Allaway, 1997; Maine et al., 2007). *Typha domingensis* (Pers.) is also fast-growing and can obtain large biomass,

thus allowing the efficient accumulation of nutrients and contaminants (Gersberg et al., 1986; Ellis et al., 1994; Manios et al., 2003; Maine et al., 2007). Tanner (1996) discussed the nutrient uptake of different macrophytes and concluded that as the total macrophyte biomass increased, the total nitrogen removal increased as well. Maine et al. (2007) found that *E. Crassipes* sequestered metals in the biomass. During senescence, the metals were retained in the sediments. For *T. domingensis*, metals were mostly retained in the sediment, but chromium (Cr) and nickel (Ni) were retained in the biomass. Nitrate and  $\text{SO}_4^{2-}$  were removed by all vegetation. The authors recommended the use of *Typha* for wastewater treatment due to their tolerance of high pH and conductivity as well as their efficient nutrient and metals removal capabilities.

The types of plants commonly used in GI systems such as constructed wetlands, rain gardens, and bioswales include common reeds (*Phragmites australis* (Cav.) Trin.), cattails (*Typha* spp.), bulrushes (*Scirpus* spp.) and reed canarygrass (*Phalaris arundinacea* L.). These types of plants have also been used in domestic and industrial wastewater treatment wetlands (Shepherd et al., 2001; Mbuligwe, 2005; Vymazal, 2005; Vymazal and Kröpfelová, 2005; Maine et al., 2009;). Plant selection is important because they need to survive potentially toxic contaminants and the variability of the systems (Maine et al., 2009).

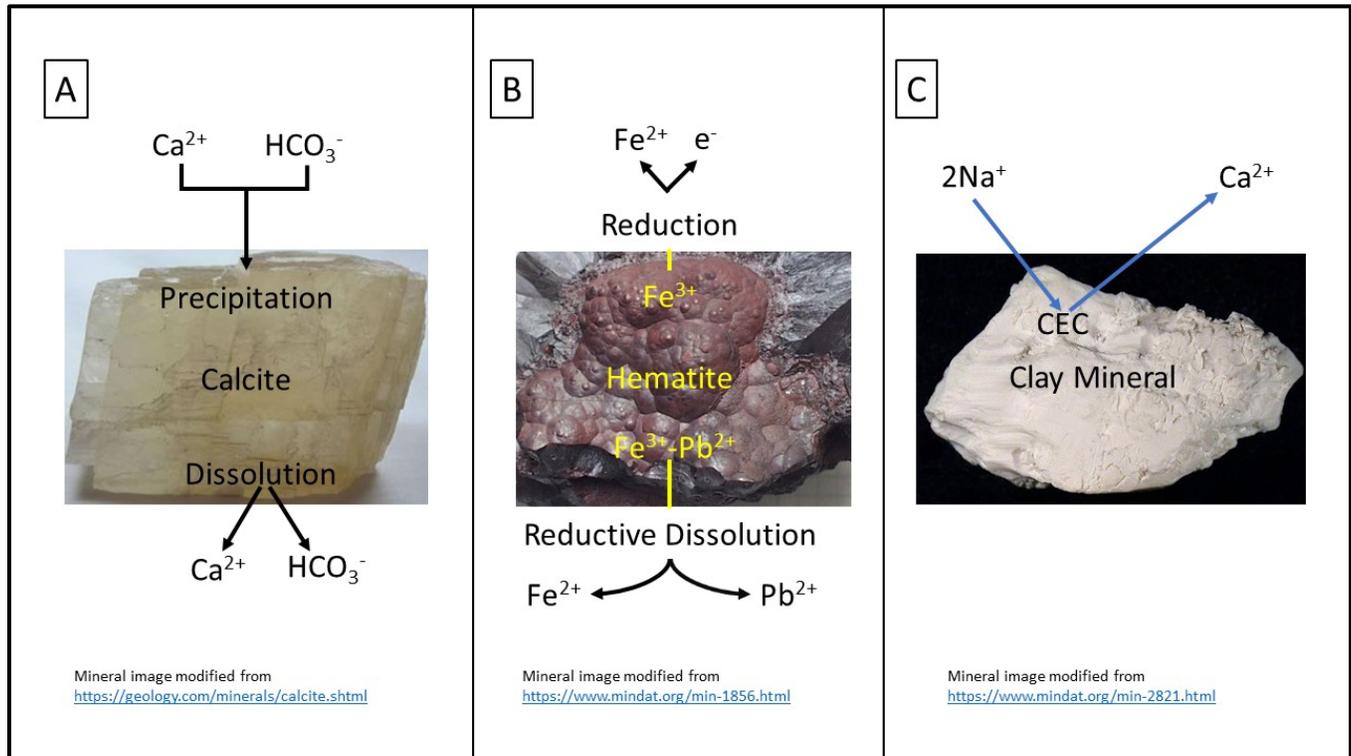
The sediments in many infiltration basins used to manage stormwater runoff can be high in contaminants such as heavy metals and hydrocarbons. Very few aquatic species can live in such environments, one being the oligochaetes *Limnodrilus hoffmeisteri* (McCall and Tevesz, 1982; Mermillod-Blondin et al., 2001; Shang et al., 2014). This species is known to be a bioturbator in freshwater sediments and a sediment feeder that creates biogenic structures in the sediments and egests fecal pellets at the water-sediment boundary, thus possibly contributing to the nutrient contamination (Matisoff et al., 1999; Shang et al., 2014).

There are few studies into how various macroorganisms may influence green infrastructure. Pigeneret et al. (2016) studied the impact of tubificid worms on the biological and geochemical processes in infiltration basins. The authors showed that as contamination increased, the burrowing activity of the worms also increased, thus increasing the macropores in the sediment. It was also observed that in the presence of tubificid worms, the mean concentrations of ammonium ( $\text{NH}_4^+$ ), nitrite ( $\text{NO}_2^-$ ) and  $\text{NO}_3^-$  increased, most likely because of the mineralization of organic matter caused by the worms (Mermillod-Blondin et al., 2008). While their study primarily focused on the contaminants effects on the worms in the sediments, they demonstrated that increased burrowing activities could possibly affect infiltration and nutrient concentrations.

# 5.0 Geochemical Processes

## 5.1 Dissolution and Precipitation Reactions

Dissolution and precipitation reactions are two examples of a geochemical process that can modify water quality in aqueous environments (Sposito, 1989; Hounslow, 1995; Stumm and Morgan, 1996; Eby, 2004). An example of a dissolution or precipitation reaction is the dissolution of calcite ( $\text{CaCO}_3(s)$ ) (Figure 8A).



**Figure 8. Geochemical Process that are important in determining water quality. A. Dissolution/ precipitation, B. Redox processes/ reductive dissolution, and C. Ion exchange processes. Calcite image in A is modified from <https://geology.com/minerals/calcite.shtml>, hematite image is modified from <https://www.mindat.org/min-1856.html>, and montmorillonite (clay) image is modified from <https://www.mindat.org/min-2821.html>.**

In the soil and unsaturated sediments (vadose zone), there is a continual cycle of dissolution and precipitation based on wetting and drying cycles (Sposito, 1989). One can apply the wetting and drying cycles to the calcite example. As water enters the dry sediments, the water hydrates the calcite surfaces and begins to dissolve the calcite, forming  $\text{Ca}^{2+}$  and  $\text{HCO}_3^-$  ions in solution (Sposito, 1989; Hounslow, 1995; Stumm and Morgan, 1996; Eby, 2004). The water containing the dissolved constituents is transported through the vadose zone under the influence of gravity. During transport, the water becomes attached to surfaces through hydration of dried surfaces or reacting with the sediments, and the total volume of water decreases. The reduction in the volume of water increases the concentration of dissolved constituents, in this case  $\text{Ca}^{2+}$  and  $\text{HCO}_3^-$ , and favors the precipitation of calcite (as was shown in Figure 8A). If the volume of water is sufficient to move the dissolved constituents to the saturated zone without precipitation of calcite, or if given enough time for wetting and drying cycles to move the dissolved constituents to the groundwater, changes to groundwater quality may occur.

## 5.2 Redox Processes

Another geochemical process that can lead to changes in water quality is redox reactions (Sposito, 1989; Hounslow, 1995; Stumm and Morgan, 1996; Eby, 2004). Redox reactions involve the transfer of electrons from one chemical species to another, as is shown in Figure 8B. Figure 8B shows that the  $\text{Fe}^{3+}$  in the hematite is being reduced to  $\text{Fe}^{2+}$ , and  $\text{Fe}^{2+}$  is being released into solution.

Hematite and other minerals have been shown to have sorbed contaminants such as metals and organic compounds (Sposito, 1989; Hounslow, 1995; Stumm and Morgan, 1996; Eby, 2004). Figure 8B shows that as the hematite is dissolved under reducing conditions, it also liberates the sorbed  $\text{Pb}^{2+}$ . The liberated  $\text{Pb}^{2+}$  changes the water quality, since there was no soluble  $\text{Pb}^{2+}$  when the solution was oxidized. The oxidation state of the Pb did not change. Conversely, if you have water containing dissolved  $\text{Fe}^{2+}$  and  $\text{Pb}^{2+}$ , it will cause  $\text{Fe}^{2+}$  to precipitate out of the solution and form a solid Fe phase as the system becomes oxidized. This solid phase is then available to sorb the  $\text{Pb}^{2+}$ , and thus alter the water quality.

## 5.3 Ion Exchange Processes

The ion exchange is a process by which an ion on the solid surface (mineral or natural organic matter (NOM)) is exchanged with another ion in solution Figure 8C (Sposito, 1989; Hounslow, 1995; Stumm and Morgan, 1996; Eby, 2004). If the surface has a net positive charge then it is capable of exchanging anions and is called anion exchange capacity (AEC), whereas if the surface has a negative charge it exchanges cations and is called cation exchange capacity (CEC) (Sposito, 1989; Hounslow, 1995; Stumm and Morgan, 1996; Eby, 2004). Generally, in natural environments, most surfaces are negatively charged except at low pH (Sposito, 1989; Stumm and Morgan, 1996), so CEC will be used to demonstrate ion exchange processes in this discussion.

Ion exchange processes can affect water quality. For example, if a permeable pavement parking lot has accumulated metals associated with brake pads in subsurface sediments and road salt is applied, then those metals could be mobilized because of the influx of sodium replacing these metals on the exchange sites.

## 5.4 Adsorption/Desorption Processes

Adsorption (sorption) processes are collective processes where there is an accumulation of matter on a solid surface and the solid-water interface (Sposito, 1989; Hounslow, 1995; Stumm and Morgan, 1996; Eby, 2004). Sorption processes influence the distribution of substances between solution and surface, which affects their transport, the electrostatic properties of the solids, and the reactivity of surfaces (Sposito, 1989; Stumm and Morgan, 1996). Sorption reactions are generally thought of in terms of the intermolecular interactions between solute and solid phases (Stumm and Morgan, 1996). These intermolecular interactions include: surface complexation, hydrophobic interactions, adsorption of surfactants, and adsorption of polymers (Stumm and Morgan, 1996).

Organic molecules can interact with the surface via hydrophobic interactions. Organic molecules that are hydrophobic, such as hydrocarbons, chlorinated compounds, PAHs, etc., are generally incompatible with polar environments and will seek a nonpolar or less polar environment. These organic molecules will seek to partition to a less polar or nonpolar parts of the mineral surface or organic surfaces. The interaction with the surface will affect the mobility of the organic substance. Partitioning coefficients

such as  $K_{ow}$  and  $K_{oc}$  can be used to predict the mobility of the organic compound (Stumm and Morgan, 1996; Eby, 2004).

Many surfactants are characterized as having both hydrophobic regions and hydrophilic regions in the same molecule (Stumm and Morgan, 1996). Surfactants lower the interfacial tension and will be sorbed at the surface (Stumm and Morgan, 1996). Because of their properties, surfactants can flocculate or disperse naturally occurring colloids, increasing turbidity and changing water quality.

Desorption is the opposite of sorption, where a molecule goes from a surface back into solution. In general, desorption occurs when there is a chemical change on the surface (dissolution of mineral, changes in surface charge, etc.) or changes in solution chemistry (pH, increased DOC, etc.). Desorption can affect water quality by mobilizing contaminants.

Sorption and desorption processes are important reactions with respect to water quality in GI systems. These processes could have both positive and negative consequences with respect to GI systems. Positive effects would be the removal of potentially harmful constituents in the infiltrating water through sorption onto surfaces and the potential immobilization of these constituents. On the other hand, negative consequences could be the desorption of harmful, naturally occurring constituents found in the soils and sediments of the vadose zone or aquifer due to the changes caused by the infiltrating stormwater.

## 5.5 Mixing Relationships

Mixing relationships can be useful when studying water infiltration into groundwater, estimating concentrations of mixtures of source waters, or estimating the percent of source water mixing with ambient groundwater (Eby, 2004; Hounslow, 1995). The other important use for mixing models is to ascertain if the mixing of two waters is possible given the concentration of a mixed water. The more complex the mixing, the more difficult the calculations become and graphical methods or software are needed (Hounslow, 1995).

A simple binary mixing model is shown in Equation 5, where  $C_m$  is the concentration of the mixture,  $C_1$  is the concentration of the more concentration solution,  $C_2$  is the concentration of the more dilute solution and %Mix is the percentage of the mixture between solution 1 and solution 2.

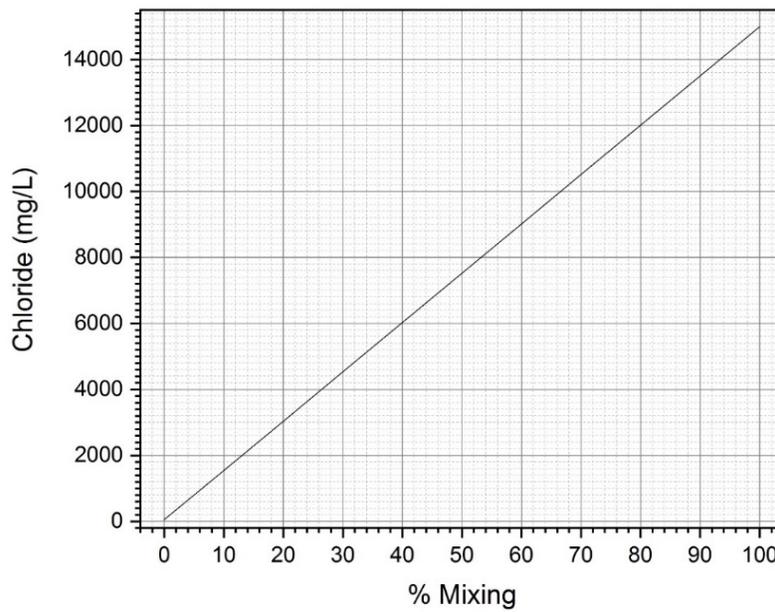
$$\%Mix = \left( \frac{C_m - C_2}{C_1 - C_2} \right) \times 100 \quad \text{Equation 5}$$

An example of mixing could be the mixing of infiltration water from a parking lot treated with road salt (NaCl). The infiltrating water has a  $Cl^-$  concentration of 12,000 mg/L, the ambient groundwater has a  $Cl^-$  concentration of 50 mg/L, and a well down gradient of the infiltration gallery has a concentration of  $C_1$  of 1500 mg/L. This would mean that the percentage of infiltrating water in the down gradient well would be 12%.

If the end member concentrations are known, it is possible to plot the %Mix as a linear function of end member concentration as is shown in Equation 6 and Figure 9.

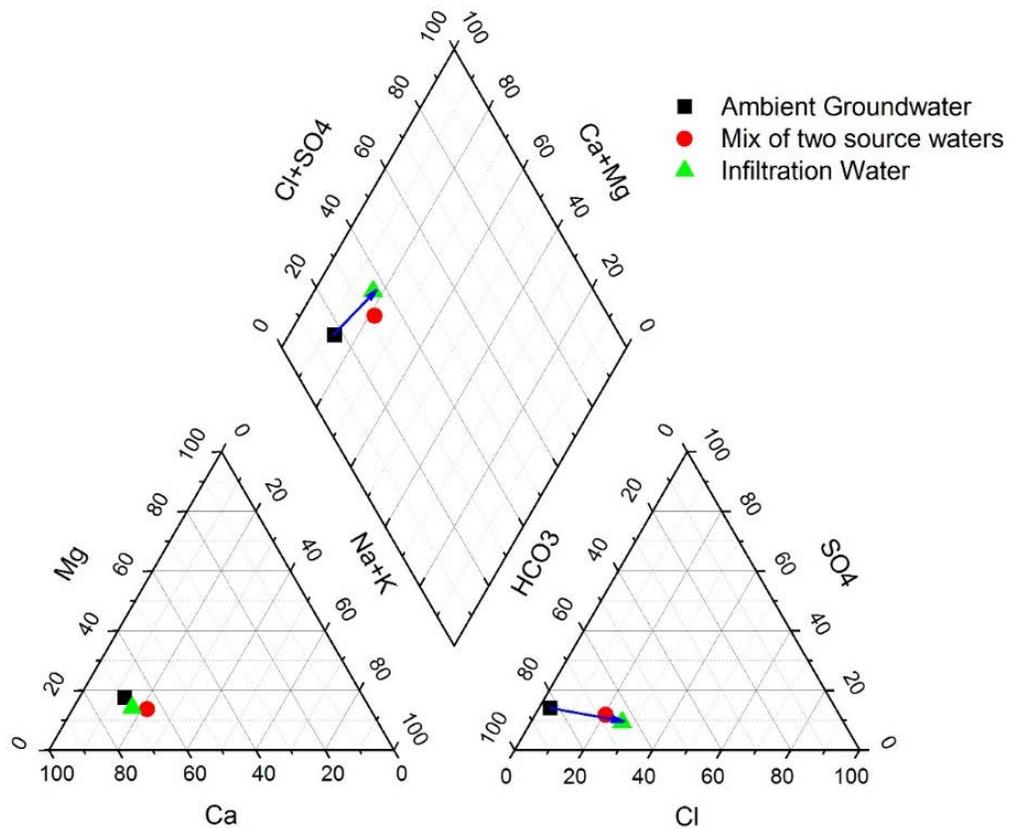
$$C_m = \left[ \frac{\%Mix}{100} (C_1 - C_2) \right] + C_2 \quad \text{Equation 6}$$

Using Equation 6, it is possible to calculate the approximate  $\text{Cl}^-$  concentration of a mixture at 10% of the solutions previously described, which would be 1545 mg/L or 50 % which would be 7525 mg/L. A graphical representation of this mixing is shown in Figure 9. If one wants to know the mixing at any percent mixing, one could find that percentage on the x-axis and use the mixing curve to find the corresponding  $\text{Cl}^-$  concentration of the y-axis. Conversely, if one knows the  $\text{Cl}^-$  concentration, one can find the corresponding percent mixing of the two waters using this plot.



**Figure 9. A chloride mixing curve for the mixing of two waters, one containing 15,000 mg/L chloride and one containing 50 mg/L chloride.**

The previous example demonstrated the mixing of one component with ambient groundwater and infiltrated water. The use of a graphical method, such as a piper diagram, can assess multiple components (Figure 10). In this example, ambient groundwater is mixed with infiltrated water after road salt application. The mixing components are the major cations (Ca, Mg, Na, and K) and the major anions ( $\text{Cl}^-$ ,  $\text{SO}_4^{2-}$ ,  $\text{HCO}_3^-$ ). The bottom left field is a trilinear plot of the major cation data. This plot demonstrates major cations were minimally impacted by the infiltrated water. The lower right field is a trilinear plot of the major anions. This plot demonstrates mixing of the ambient ground water with the road salt impacted infiltration water, causing the percentage of  $\text{Cl}^-$  in the sample to increase, and the percentage of  $\text{HCO}_3^-$  to decrease. The blue arrow shows the direction of change with respect to ambient groundwater. The advantage of the Piper diagram is that any changes in the lower fields are plotted in the “mixing” diamond in the upper plot. The blue line once again shows the direction of shift from the ambient groundwater condition to the infiltration water endpoint. This indicates there is minimal change in the cations, whereas the anions show an increase in  $\text{Cl}^-$  contribution and a decreased contribution of  $\text{HCO}_3^-$  in the mixed water.



**Figure 10. A Piper Diagram demonstrating the mixing of ambient groundwater with infiltrated water that was impacted by road salt application.**

## 5.6 Colloids

Colloids are small particles ranging in size between true solutions and suspensions (Eby, 2004; Bradford and Torkzaban, 2008). Colloids can be inorganic, organic, or biological, and have variable size ranges in literature (Sposito, 1989; McGechan and Lewis, 2002; de Jonge et al., 2004; Eby, 2004; Shein and Devin, 2007; Bradford and Torkzaban, 2008). A soluble species in water is defined as any substance that passes through a  $0.45\mu\text{m}$  filter (Eby, 2004). However, Eby (2004) also points out there are many molecular sized particles that can pass through a  $0.45\mu\text{m}$  filter that are still considered to be colloidal in nature. Eby (2004) defines colloids as particles that range from 10 nm to  $10\mu\text{m}$  in size, which is supported by others (Sposito 1989, Bradford and Torkzaban, 2008, Drake et al., 2014). de Jonge et al. (2004) provide a range of 1nm to  $10\mu\text{m}$  for the size of colloids. The range proposed by Shein and Devin (2007) is 1nm to 100 nm. McGechan and Lewis (2002) state the size range of colloids for nonliving colloids is  $> 2\mu\text{m}$  in non-karst situations and  $> 4\mu\text{m}$  in karst situations. However, these authors discuss biological colloids in different size ranges: viruses range from 20 - 200nm, bacteria range from 500nm –  $3\mu\text{m}$ , and protozoa from  $4\mu\text{m}$  –  $14\mu\text{m}$  (McGechan and Lewis, 2002). Because of their unique nature, colloids are important to chemical and biological transport in the vadose zone and in groundwater (McGechan and Lewis, 2002).

Colloidal transport is an important mechanism by which contaminants can move through the soil (Sposito, 1989; McGechan and Lewis, 2002; de Jonge et al., 2004; Eby, 2004; Shein and Devin, 2007; Bradford and Torkzaban, 2008; Zhang et al., 2010). It is important to note the colloid can itself be the

contaminant, or the contaminant is sorbed to the otherwise benign colloid (McGechan and Lewis, 2002). In most cases the colloid itself is harmless, and the contaminants sorbed or associated with the colloid is the concern (Sposito, 1989; McGechan and Lewis, 2002; de Jonge et al., 2004; Eby, 2004; Shein and Devin, 2007; Zhang et al., 2010).

Unlike soluble components, which can freely move in solution, colloids have restricted movement in some cases (McGechan and Lewis, 2002). Colloids can be restricted by capture, sorption and by electrostatic interactions (Sposito, 1989; McGechan and Lewis, 2002; de Jonge et al., 2004; Shein and Devin, 2007). Capture can be by straining where the colloid is larger than the pore and movement is stopped by the pore (McGechan and Lewis, 2002). Straining is also implicated as an important capture mechanism for biological colloids (Shein and Devin, 2007). Another capture mechanism is filtration where the colloids can travel into the pore and are deposited inside the pore (clogging of the pore) (McGechan and Lewis, 2002). Colloids are subject to the same geochemical process as solution species due to their chemical composition, i.e., they can be sorbed to surfaces and can interact with surfaces electrostatically, which will impede their transport (Sposito, 1989; McGechan and Lewis, 2002; Bradford and Torkzaban, 2008). Colloids in macropores can move relatively unrestricted and quickly (McGechan and Lewis, 2002; Shein and Devin, 2007). Other researchers discuss the chemical and physical properties that are important for the transport of colloids (de Jonge et al., 2004; Shein and Devin, 2007; Bradford and Torkzaban, 2008). Sposito (1989) and de Jonge et al. (2004) discuss in broad terms the role of dispersion and aggregation; ionic strength; and pH as important factors that control colloid transport. Shein and Devin (2007) state that as pH increases, the concentration of microbes in groundwater increases, and water with low IS favors transport of microbe. Natural organic matter (NOM) has also been implicated in influencing colloid transport (Kretzschmar et al., 1995). In the presence of NOM, there is a decrease in attachment efficiency to the soil and sediments which increases mobility. Also, when NOM is sorbed onto the colloid, there is more stability which leads to a decrease in filter efficiency of the media, and thus facilitates transport (Kretzschmar et al., 1995). Zhang et al. (2010) discuss colloids in unsaturated conditions. These authors found because colloids are retained more under high IS, the greatest transport will be right after rainfall (lower IS), and colloids are less well retained in the lower portions of the vadose zone due to concentration effects. Additionally, chemotaxis needs to be considered in the transport of biological colloids (Shein and Devin, 2007). Some microbes are capable of movement of their own making which allows them to move towards nutrients and away from harmful environments.

As discussed previously, colloid-facilitated transport is an important mechanism for the movement of contaminants into groundwater (de Jonge et al., 2004). This phenomenon has been demonstrated both in laboratory experiments and in field based research. Colloid-facilitated transport has been shown for inorganic and organic substances. Examples of organic substances includes atrazine, prochloraz, DDT, and glyphosate. Inorganic examples include chromium (Cr), cesium (Cs), copper (Cu), Ni, phosphorous (P), Pb, and zinc (Zn).

Colloids play an important role in the potential transport of contaminants in GI systems. In the environment, dust is commonly deposited on the ground surface and can come from many sources such as industrial sites, urban landscape, automobile exhaust, brake pads, etc. Dust particles from sources such as these have contaminants associated with them. Precipitation can wash the deposited dust into the GI system and could potentially allow the transport of the contaminant through the vadose zone into the groundwater on the colloid. Many GI systems are designed to capture the colloids and minimize the likelihood of transport into the subsurface. However, this could allow the buildup of contamination in the upper portions of the vadose zone.

## 6.0 Models

There have been many reviews of models and tools used for modeling stormwater management, and for modeling green infrastructure implementation (Zoppou, 2001; Vaze and Chiew, 2003; Elliott and Trowsdale, 2007; Obropta and Kardos, 2007; Jayasooriya and Ng, 2014). Zoppou (2001) reviewed the mathematical processes in various stormwater models that focused on quality and quantity, but did not discuss any incorporation of GI into those models. Vaze and Chiew (2003) compared models to estimate event diffuse contaminant loads from urban stormwater, but did not address GI. Elliott and Trowsdale (2007) studied modeling tools with GI in urban stormwater drainage modeling, but their discussion was focused on water quantity. Ahiablame et al. (2012) compared three different models to assess benefit for simulating GI effects on quality and quantity. Obropta and Kardos (2007) compared various urban stormwater quality models, comparing the deterministic, stochastic, and hybrid approaches to modeling, but did not evaluate their potential for modeling GI for stormwater management. Jayasooriya and Ng (2014) present a comprehensive review of models for green infrastructure practices, but also evaluated the model for the economics of GI.

Generally, a model is a description of an often complex system (Haefner, 2005) into a simplified form (Imboden and Pfenninger, 2012). Since the 1960s, software tools have been used for water resource management such as the design and optimization of sewer systems and wastewater treatment technologies (Zoppou, 2001; Jayasooriya and Ng, 2014; Eckart et al., 2017). Modeling tools to simulate the stormwater runoff quantity and quality emerged in the 1970s (Zoppou, 2001; Jayasooriya and Ng, 2014; Eckart et al., 2017). Simulation modeling is a valuable method to obtain spatial and temporal information on a wide range of scales. There is a wide range of monitoring practices over various periods and conditions, but the high cost can make modeling a more efficient method (Elliott and Trowsdale, 2007; Eckart et al., 2017). The growing impact of urban stormwater on surface water quality has driven the need for more accurate models for stormwater pollution and management (Beck, 2005; Obropta and Kardos, 2007). Many urban stormwater management models often focus solely on the surface flows and not subsurface flows (Barnett et al., 1995; Blanc et al., 1995; Palla and Gnecco, 2015; Her et al., 2017; Herrera et al., 2017; Kong et al., 2017; Paule-Mercado et al., 2017). It is challenging to develop stormwater and groundwater models alone, but there are additional difficulties to linking them to portray groundwater quality effects.

Stormwater models can support many needs for management such as development of total maximum daily loads (TMDLs), CSO management, BMP selection and placement, land use change impact assessments, and an analysis of water quality changes and opportunities with stormwater management systems (Obropta and Kardos, 2007). It is useful to include GI practices in stormwater management tools so researchers and managers can better understand the behavior of the various GI practices (Jayasooriya and Ng, 2014). Models on green infrastructure management have been used to compare various scenarios for policy decisions on improving large scale water distribution and stormwater treatment (Sharma et al., 2008).

Green stormwater infrastructure is part of integrated urban water management (IUWM) that achieves greater sustainability for drinking, waste, storm, and receiving water management, which drives the need for more accurate stormwater quality models (Beck, 2005; Obropta and Kardos, 2007). It is needed for stormwater management to evaluate the efficiency of different control strategies and optimal solutions (Beck, 2005; Rousseau et al., 2005; Obropta and Kardos, 2007). IUWM is dependent on integrated models to provide decision support of alternative options, allow for long term planning, inform for

design upgrades, and provide real time system control actions (Rauch et al., 2002; Beck, 2005; Rousseau et al., 2005; Olsson, 2006; Obropta and Kardos, 2007).

Elliot and Trowsdale (2007) did a review of models for LID and found most current models do not incorporate enough contaminants that are a concern to stormwater quality. It was also found that it can be difficult to link the hydrologic models to other processes such as toxicity, habitat models, land use, and cover maps (Thomas and Tellam, 2006; Elliott and Trowsdale, 2007). Hydrologic models are more common than water quality models as the water quality data needed to calibrate these models is less available and quality is more difficult to model (Elliott and Trowsdale, 2007; Eckart et al., 2017;). There are many commonly used models for stormwater management: Stormwater Management Model (SWMM), Storage Treatment Overflow Runoff Model (STORM), Hydrologic Simulation Program-Fortran (HSPF), Distributed Routing Rainfall Runoff Model – Quality (DR3M-QUAL), Hydroworks, and Model of Urban Sewers (MOUSE) (Obropta and Kardos, 2007). Commonly used groundwater and soil moisture flow models include Modular Flow Model (MODFLOW), Hydrus, Saturated-Unsaturated Transport (SUTRA), Soil Water Atmosphere and Plant (SWAP), Finite Element Subsurface FLOW System (FEFLOW), and Transport of Unsaturated Groundwater and Heat (TOUGH). SWMM, Model for Urban Stormwater Improvement Conceptualization (MUSIC), MOUSE, and Hydrologic Modeling System (HEC-HMS) are popular and widely used software for urban hydrology and stormwater drainage management, but there needs to be more research in modelling rainfall-runoff (Fletcher et al., 2013; Eckart et al., 2017). Models such as SWAP (Kroes and Van Dam, 2003) and Hydrus (Šimuněk et al., 1999) are rarely used for stormwater infiltration due to their complexity, and very few models take clogging into account (Browne et al., 2008). While there are a variety of modelling techniques to evaluate the effectiveness of urban GI practices, the primary issues needed to determine the model's accuracy and certainty of the result are calibration, sensitivity analysis, and uncertainty analysis (Eckart et al., 2017). SWMM is an EPA runoff model, and seems to be one of the more comprehensive modeling programs that already incorporates permeable pavements, rain gardens, green roofs, street planters, rain barrels, infiltration trenches, and vegetated swales (Jayasooriya and Ng, 2014). Many vadose zone models were developed initially to predict organic or inorganic plume migration beneath landfills (Clark and Pitt, 2007).

Often a large model is an integration of multiple submodels, and reliability of these integrated models is dependent on the quality of the smaller models. Combinations can create integrated models such as the water supply, wastewater treatment, stormwater, and receiving water models. The stormwater models are considered the least developed, making them a hindrance to integrated model performance; for this to improve, the stormwater model predictions need to improve (Rauch et al., 2002; Obropta and Kardos, 2007).

Models can be classified by two different approaches: deterministic and stochastic. Deterministic models attempt to perfectly simulate the physical world through cause and effect relationships, while the stochastic models use statistical patterns to simulate the phenomenon. Differences between deterministic and stochastic models are not absolute. Deterministic models can have some sort of randomness and uncertainty, but stochastic models do not ignore the causal relationships. The best way to differentiate between the two classifications of models is that deterministic models will always have the same response from the same model input, but stochastic models will have varying responses from the same input but with consistent statistical properties (Nix, 1994; Zoppou, 2001; Obropta and Kardos, 2007).

There are various limitations to each classification of models. The deterministic models do not consider variable uncertainty. The complex deterministic models can contain multiple optimum parameter sets where no single parameter set can be identified as a unique solution set (Beven and Freer, 2001; Obropta

and Kardos, 2007). For example, parameter non-uniqueness can be seen in rainfall/runoff and stormwater quality models (Doherty and Johnston, 2003; Kanso et al., 2003; Obropta and Kardos, 2007). The stochastic model's limitations are that they are often only developed from available data without detailed knowledge of the underlying processes, and when this data is sparse or noisy, it is often inadequate to make reliable predictions (Vojinovic et al., 2003; Obropta and Kardos, 2007;). Stochastic models cannot be extended to analyze alternate scenarios such as increased urbanization effects on stormwater contaminant loads. These models are only applicable to specific conditions and data for what the model is based on, but the advantage of these models is that uncertainty for the variables is built into the model (Scholz, 1997; Zoppou, 2001; Obropta and Kardos, 2007).

It has been shown that stochastic modeling techniques are not just a substitute for deterministic models, they can be combined with deterministic models to create hybrid deterministic-stochastic models that enhance stormwater quality modeling (Warwick and Wilson, 1990; Gong et al., 1996; Scholz, 1997; Obropta and Kardos, 2007). Hybrid models integrate stochastic and deterministic models which can take the best of both types of model method, thus reducing stormwater quality model prediction error and uncertainty (Obropta and Kardos, 2007).

Each model requires different sets of parameters and inputs depending on the purpose of the model. Inputs can include catchment size, scale, human activities, climate, and natural characteristics such as soil type and porosity. Outputs depend on the question the model is supposed to answer, but can include runoff volume, runoff rates, and contaminant loading (Jayasooriya and Ng, 2014). An important part of planning and implementing GI stormwater management practices is the spatial multi criteria analysis (Eckart et al., 2017). Criteria that need to be considered when planning GI installation are site characteristics, site suitability, performance of runoff controls, economic feasibility, slope, water table depth, soil types, rainfall patterns, and catchment size (Jia et al., 2013; Eckart et al., 2017; Johnson and Sample, 2017; Joyce et al., 2017). There is extensive literature that discusses different models and tools for evaluating the best GI or treatment train for stormwater management (Jia et al., 2013; Charlesworth et al., 2016; Eckart et al., 2017; Johnson and Sample, 2017; Joyce et al., 2017).

It is often the assumption that a simulation model of a system must include all the variables and pathways, but more model complexity does not always improve the accuracy of the simulation (Doherty and Johnston, 2003; Obropta and Kardos, 2007). The more complex the model, the greater the number of non-unique parameter solution sets and model uncertainty, thus the fine detail of a system can only be replicated by the amount of necessary complexity needed. Even the most complex model that represents a system's true behavior will have uncertainty limits of its predictions and all models will need to have uncertainty analyses done to understand the limits to the stormwater quality predictions (Obropta and Kardos, 2007). Complex models come with the costs of more parameters to calibrate, more data to collect, and more time and resources to implement the model. As the number of parameters in the model increases, the harder it is to manually calibrate the model. The innate complexity of stormwater quantity and quality processes drives these models to be more complex, requiring more automatic calibration methods. These automatic methods need to locate global optima, have numerical stability, and computational burden. The optimization criteria are important to perform automatic calibrations (Skahill and Doherty, 2006; Obropta and Kardos, 2007). There are many examples of methods used to calibrate water quantity and quality modes such as linear, nonlinear, heuristic, and probabilistic methods (Gauss-Marquardt-Levenberg method, genetic algorithms, and Monte Carlo Markov chains), but few examples of automatic calibrations with stormwater quality models. This could be due to automatic calibrations needing more data, and this data is often lacking (Obropta and Kardos, 2007).

Due to the expense of monitoring and sampling, the data needed to calibrate stormwater models is lacking and oftentimes collected inconsistently, which is a great hindrance to the development of suitable algorithms (Zoppou, 2001; Rauch et al., 2002; Obropta and Kardos, 2007). There are various examples of simple models simulating comparably to complex models from Liner-Lundsford and Ellis (1987), Vaze and Chiew (2003), and Ahyerre et al. (2005). Ultimately the intended use of the model is what guides the selection of model complexity, but the least complex model that reliably answers the questions is ideal (Rauch et al., 2002; Obropta and Kardos, 2007;). As there is an increased need for accurate predictions, there will be a need for more complex models.

Water quality is not modeled as often as hydrology, more often being studied through experimentations (Eckart et al., 2017). One of the difficulties of this is more field research is needed to characterize the runoff water quality from different land use types (Ahiablame et al., 2012; Eckart et al., 2017). A Water Quality Capture Optimization and Statistics Model (WQ-COSM) was developed to determine the water quality capture volume for GI design (Guo et al., 2014; Eckart et al., 2017;). The SUSTAIN model has been used to study water quality performance for a watershed in Taiwan (Chen et al., 2014). Chen et al.'s (2014) model compared pervious pavements, bioretention cells, and grass swales for total phosphorus (TP), total nitrogen (TN), suspended solids (SS), and biological oxygen demand (BOD) removal; finding that pervious pavements had the largest reduction. Seo et al. (2017) modelled GI impacts on water quantity and quality using the Soil and Water Assessment Tool (SWAT) and showed that GI practices reduced contaminant loading for various land use types.

Modeling stormwater quality is generally dependent on the model of stormwater quantity, because if the flow is not modeled correctly, the quality predictions will not be correct. Stormwater deposits, erodes, and transports many different contaminants at uncertain rates and concentrations, and the contaminants can vary between being soluble or particulate. The contaminants can also interact with and/or alter the microbial degradation processes in the system. Most current stormwater quality models are less accurate than the quantity models due to these many uncertain parameters. Most stormwater models are based on modeling sediment loads and concentrations where the contaminant levels are then estimated as a fraction of the sediment load. This makes the sediment grade and particle size important because smaller particles will absorb a greater proportion of contaminants. The surface sediment processes are often done separately from the subsurface processes due to the flow being different, and the contaminants deposit differently than on the surface (Obropta and Kardos, 2007). Buildup and wash-off models are the most common methods used to simulate stormwater runoff quality, which applies to most overland processes, but can be very uncertain and unreliable (Robien et al., 1997; Zoppou, 2001; Kanso et al., 2003; Obropta and Kardos, 2007;). Dechesne et al., (2004a) modeled a stormwater infiltration basin to simulate hydraulic performance and pollution retention. The authors satisfactorily modeled the hydraulics and clogging, but pollution flow was not as reliable. This study was not intended to model contamination of the groundwater, but to determine the clogging and life-time of these basins.

Models have been developed to simulate contamination of groundwater, but not in green infrastructure conditions. The modeling of groundwater recharge has been done on various scales, but is often limited to the quantity and flows of the groundwater recharge and not the quality of the groundwater after recharge (Abbott and Stanley, 1999; Bogena et al., 2005). Lee et al. (2006) modeled nitrogen pollution in groundwater from cattle feed lots addressing the fate and transport of nitrogen species, dissolved organic carbon, dissolved oxygen (DO), and biomass in the groundwater. They developed the code for this using MODFLOW. While not a green infrastructure model, the processes can be transferred to simulating green infrastructure practices. Thomas and Tellam (2006) developed a model in GIS to estimate spatially distributed recharges and recharged water quality in unconfined aquifers. Although they did not incorporate green infrastructure into this model, the model has the attributes that could be

used for green infrastructure assessment. The processes included in their model included runoff and interception using the curve number methods; Penman-Grindley evapotranspiration; the empirical index approach to interflow; volatilization using Henry's law; distribution coefficient for sorption; and first order decay for degradation. Data needed for this model includes meteorological, land use/cover, hydraulic and geochemical attributes, and topographic and water table elevation. The model predicted the distributions of surface runoff, infiltration, potential recharge, ground level slope, interflow, actual recharge, surface runoff contaminant fluxes, travel time for contaminants through the vadose zone, and the contaminant fluxes at the water table. This model showed promise for exploring the influences on urban groundwater quantity and quality from land use changes. The results of the model included spatial detail that could be incorporated with green infrastructure. Lee et al. (2003) developed a stochastic model using GIS to evaluate groundwater quality in urban areas. Their study focused on concentrations of nitrate based on land use, rainfall, and well depth. Although their model did not address green infrastructure, their model found that land uses correlated well with nitrate concentrations.

Browne et al. (2008) developed a one-dimensional stormwater infiltration model to simulate the hydraulic processes of inflow from a stormwater model for stormwater infiltration systems. Bioretention and rain garden models were developed using the Green-Ampt equation (Dussailant et al., 2003) and Richard's equations (Dussailant et al., 2004). The Green-Ampt model is the simpler of the two methods, and shown to be effective when compared to Richard's equation, and thus is the most commonly used equation for rain gardens/bioretention, although it would be challenging to apply the Green-Ampt model to two-dimensional systems (Atchison et al., 2006). Their one-dimensional model contained four blocks: the storage, clogging layer, saturated soil zone, and unsaturated soil zone. They focused on the problem of simulating flow-through and ponding within the storage zone. One of the difficulties the authors addressed was the transition between the boundary zones of storage and saturated soil. They concluded their numerical model could obtain reliable results for both the unsaturated and saturated flows with ponding, and thus be useful for stormwater infiltration systems' various conditions. This model did not address the contaminants of the stormwater entering the soil and groundwater, but addressing the difficulties of combining stormwater and groundwater models for infiltration needs to first be done hydrologically before contaminant models can be reliable. The authors summarized approaches to modelling flow through infiltration systems: 1) Estimate the time it takes for the system to empty based on design events using empirical equations. 2) Assume a constant saturated hydraulic conductivity and infiltration rate. 3) Assume clogged conditions control infiltration. 4) Adopt a general unsaturated soil flow model. (Browne et al., 2008).

Clark and Pitt (2007) presented a methodology to predict the potential for contamination from infiltration into the groundwater. The steps include determining the contaminants of concern, identifying soil characteristics, and then using this information to predict the potential for groundwater contamination. The model suggested for predicting groundwater contamination is SESOIL (Seasonal Soil Compartment model). SESOIL can use actual or estimated field data to predict the depth of migration, and model both organic and inorganic contaminants migrating through the vadose zone. It is a theoretical fate and transport model that results in mass balances and assumes equilibrium partitioning between phases. It incorporates three submodels: hydrologic cycle, sediment wash-load cycle, and contaminant fate cycle.

# 7.0 The Potential for GI to Impact Water Quality

The potential for impacts to groundwater quality is the primary focus of this review, only a limited literature could be found that discusses the potential for groundwater impacts or impacts to groundwater. The following section will present this literature.

## 7.1 Sampling Methodology

The literature demonstrated many different sampling design methods used for monitoring contamination and potential changes to groundwater quality from stormwater infiltration. These methods all consider several common questions in their soil and groundwater sampling design: 1) How many sampling points are needed to adequately assess if groundwater changes occurred? 2) Where should the sampling points be located? 3) At what depths should the groundwater samples be collected? and 4) How do you get a background or control groundwater sample that will reflect the initial condition (Tedoldi et al., 2016)? Tedoldi and colleagues concluded that there was no systematic method for determining the number of sampling points; however, based on statistical consideration, a minimal number of sampling points could be approximated to estimate a true mean contaminant concentration (Gilbert, 1987; Pennock et al., 2007). Others, such as Mason et al. (1999), have suggested that professional judgement can play a role in sampling design based on the researcher's knowledge of the study location. One common sampling point is usually near where the water enters the infiltration structure (Mikkelsen et al., 1996; Dechesne et al., 2004b; Dechesne et al., 2005; Winiarski et al., 2006; DiBlasi et al., 2009; Jones and Davis, 2013; Paus et al., 2013; Tedoldi et al., 2016).

A 550 m<sup>2</sup> infiltration basin was studied by Dechesne et al. (2004b) with what the authors termed a high spatial resolution sampling design. They collected two sediment samples per 100 m<sup>2</sup> at the following depths: surface, 30 to 40 cm, 60 to 70 cm, and 100 to 110 cm. They concluded only three points were needed to estimate the total mass of trapped contaminants within approximately a 25% error. The three points were a point near the inflow, the lowest point in the basin, and a point that is representative of most of the basin. This researcher then applied this method to other basins and found similar results (Dechesne et al., 2005). Le Coustumer and Barraud (2007) in studying metals in a 7,400 m<sup>2</sup> infiltration basin found spatial heterogeneity of surface contamination. They found using a regular sampling grid of approximately 15 points was not significantly different than using a grid of 100 points, and the standard deviation was roughly constant for 30 samples or 100 samples. However, fewer samples spaced out over a grid could lead to missing potential hot spots and erroneous conclusions. Other types of infiltration structures, such as filter strips that receive runoff sampling points, should run in sampling transects perpendicular to the filter strip (Boivin et al., 2008; Kluge and Wessolek, 2012; Kluge et al., 2014).

The vertical distribution of contaminants in the soil or below the infrastructure shows there is no consensus on the sampling depth (Barth et al., 1989; Barraud et al., 2005; Ingvertsen et al., 2012; Jones and Davis, 2013; Tedoldi et al., 2016). Two phase sampling was found to be the most effective strategy for vadose zone sampling (Barth et al., 1989). The first phase sampling should give an understanding to how sampling should be carried out in the second phase, which is the recommended sampling strategy by the U.S. EPA (Barth et al., 1989) and by ISO 10381-5 (ISO, 2005), though it is rarely adopted (Tedoldi et al., 2016). Instead, the depths sampled range from a few centimeters to several meters (Mikkelsen et al., 1996; Mikkelsen et al., 1997; Norrström and Jacks, 1998; Mason et al., 1999; Dechesne et al., 2004b; Barraud et al., 2005; Dechesne et al., 2005; Winiarski et al., 2006; Aryal et al.,

2007; Strömvall et al., 2007; Camponelli et al., 2010; Ingvertsen et al., 2012; Kluge and Wessolek, 2012; Jones and Davis, 2013; Paus et al., 2013; Tedoldi et al., 2016;). Several researchers used depth information from previous studies and tried to apply this information to set a maximum depth to be sampled at in their current studies (Barraud et al., 2005; Ingvertsen et al., 2012; Jones and Davis, 2013). This method led those researchers to very different depth values; 110 cm, 25 cm, and 90 cm, respectively. Sampling only 20 - 30 cm assumes the contaminants are sequestered in the upper layers of the vadose zone and will not account for anomalies, or see if contamination extends below the upper layers (Tedoldi et al., 2016). Based on the literature discussed, the concentration of a contaminant is generally the greatest near the surface and decreases with depth.

To assess if there is contamination or a change in water quality, a control concentration or background concentration needs to be established (Tedoldi et al., 2016). Tedoldi et al. (2016) also point out “uncontaminated” or “background” can lead to some confusion since in urban and industrial areas, the concentrations could have already been elevated prior to the installation of the infiltration structure. Desaules (2012) provided four possible methods for determining a reference value for metals in soil systems or in the vadose zone. One method is a statistically derived value, such as percentiles from measurements at wider scales. Another method is what Desaules (2012) termed “usual” content, which is similar to the uncontaminated context. A third approach was deriving reference values by looking at the concentrations in the deeper layers. This assumes deeper layers are not contaminated to any extent. The final approach was to use the residual content in a sequential extraction method. Desaules (2012) does point out difficulties can arise when using the first two methods for a site with low levels of contamination and may not be appropriate. Except for Dechesne et al. (2004b; 2005), none of the review literature were verified to see if their control sample was comparable to the background or uncontaminated values. This can be problematic and cause significant bias on the site specific initial conditions. Bowen et al. (2015) caution that use of national data sets could be problematic for determining background concentrations due to potential issues with sample size limitations and problems with the spatiotemporal distributions.

## **7.2 Review of Potential Contamination from Green Infrastructure Infiltration Studies**

The potential for groundwater contamination resulting from the use of several different types of GI practices has been studied to a limited extent, and results and information on study methods will be presented below.

### **7.2.1 Infiltration Basins/Recharge Basins**

Infiltration basins or recharge basins are a commonly used form of enhanced infiltration or GI. Essentially, stormwater runs off mainly impervious surfaces and is collected in a basin where the water is stored until infiltration occurs or it evaporates. The impacts of these structures on vadose zone and groundwater will be discussed in this section.

Nightingale and Bianchi (1977) studied the effects of a recharge basin on groundwater. The facility rests on a compound alluvial fan comprised of heterogenous layers of sands, silts, and clays. Groundwater sampling was accomplished using a ten well monitoring network, and each well was screened below 20 m. An additional 25 to 30 wells were sampled semi-monthly. The sampled water was analyzed for SPC,  $\text{NO}_3^-$ , and  $\text{Cl}^-$ . After five years of use, an estimated 65,815,000  $\text{m}^3$  of water infiltrated into the subsurface. Researchers found that in the beginning of the annual recharge period the SPC would increase, but would then decrease by the end of the recharge year. They also found  $\text{NO}_3^-$  leached from the soil during the first recharge event. Chloride concentrations followed the same pattern as  $\text{NO}_3^-$ . The

researchers concluded that use of this recharge basin decreased the groundwater salinity downgradient for 1.6 km.

At a different site, Nightingale (1987a, 1987b), examined the impact of urban stormwater runoff on alluvium soils comprising the vadose zone and groundwater at five retention/recharge basins using ceramic/Teflon vacuum water extractors up to 26m deep for two years. The silt plus clay percentages ranged from 3-74% for Basin F, 14-68% for Basin G, 5-46% for Basin M, 24-92% for Basin EE, and 2-8% for Basin MM with extensive stratification. Basin F was 2.4 ha in size with a depth of 2.7 m, had a 50,085 m<sup>3</sup> storage capacity, and drained 132 ha. Basin G was 3.4 ha with a depth of 2.4 m, a storage capacity of 43,793 m<sup>3</sup>, and drained 163 ha. Basin M drained 251 ha, was 3.6 ha in size, had a depth of 3.0 m, and a storage capacity of 122,067 m<sup>3</sup>. Next was Basin EE, which had a storage capacity of 246,847 m<sup>3</sup>, was 8.8 m deep, and was 4.0 ha in size. The final basin, Basin MM was 3.4 ha in size, had a depth of 6.4 m, stored 145,344 m<sup>3</sup> of water, and drained 310 ha. To assess groundwater contamination or the potential for future ground water contamination, both the soil in the vadose zone and soil porewater were sampled (Nightingale, 1987a, 1987b). The soil samples were taken at a point near the outfall and midway between topographic highs and lows of the basin floor. The depth intervals the soil samples were taken from were 0 – 2, 2 – 5, 5 – 8, 8 – 14, 14 – 20, and 24 – 30 cm, and they were analyzed for As, Ni, Cu, and Pb. Soil porewater samples were obtained using a vacuum ceramic-teflon water extractor. The porewater samplers were placed on top of a soil layer with low hydraulic conductivity. The soil porewater samples were analyzed for SPC, major cations and anions, trace metals, and organic compounds. Soil sampling showed metals accumulated in the top few centimeters of the soil, and there was no significant leaching of the metals into lower portions of the vadose zone. The results of the soil porewater sampling showed there was no trace metal contamination or contamination from most organic compounds; however, diazinon was detected in three samples. The salinity, major cations, and anions were reflective of the levels in the stormwater runoff.

Groundwater contamination by organic compounds from a rapid infiltration site was investigated in a study authored by Tomson et al. (1981). The authors used soil column effluents, made from PVC and filled with pea gravel and sandy loam soil, to collect samples. The soils types and geology under this basin were not reported in this study. Although this examined the infiltration of secondary treated wastewater, the results from this study would be a reasonable proxy to stormwater infiltration. In this study, there were four parallel infiltration basins; each basin was 40,000 m<sup>2</sup> with a 300,000 m<sup>2</sup> holding pond. The results showed the infiltration basin could not remove 100% of the organic compounds, that there was little daily variation in organic compound concentration in the groundwater, and that different classes of compounds had different removal efficiencies.

The results of a study exploring the impacts of three infiltration basins on groundwater in Perth, Australia were presented by Appleyard (1993). They were located over predominantly sand sediments with clay and limestone with these superficial formations forming an extensive aquifer under Perth. Infiltration basin 1 received stormwater runoff from a light industrial area and a residential area. This infiltration basin was about 1,200 m<sup>2</sup> and drained at least 67,000 m<sup>2</sup> catchment. The second infiltration basin was in a residential area, was 380 m<sup>2</sup>, and drained 74,000 m<sup>2</sup>. The final infiltration basin received water mainly from a major highway and a residential area. This infiltration basin's catchment was approximately 58,000 m<sup>2</sup> and was about 600 m<sup>2</sup> in size. Three to four boreholes were sunk next to each site, located to measure groundwater quality upgradient and downgradient of each basin and samples were collected twice to observe seasonal variations. The results showed a major change in groundwater quality down gradient of the infiltration basins because of a significant reduction in TDS due to stormwater infiltration. These changes were, however, short-lived. Dissolved oxygen (DO) appeared to control the concentration of Fe in the shallow groundwater. Infiltration increased DO concentration

downgradient which in turn decreased the concentration of Fe in the shallow groundwater. Water withdrawals near the infiltration basin could be affected by clogging and encrustation because of the potential precipitation of Fe. The concentrations of pesticides, phenolic compounds, metals, and nutrients near the infiltration basins were low. The concentrations of Pb at the third infiltration basin were more than 3,500 mg/kg, which the author attributed to vehicular traffic.

Barbosa and Hvitved-Jacobsen (1999) conducted a study that considered the sequestration of heavy metals contained in highway runoff by an infiltration pond. Three Portuguese soils were tested: Vila Real and Costa da Caparica soil which were sandy with carbonates, and Estremoz soils which had high silt, clay, organic matter and CEC. The first 25 cm of soil samples were collected from areas not too close to the roads and used for sorption and desorption experiments. They monitored highway runoff using a rain gauge, flowmeter, and an automated sampler. The total area of the catchment was 5,970 m<sup>2</sup> with 2,500 m<sup>2</sup> being road pavement. The heavy metals analyzed were Cd, Cr, Cu, Pb, and Zn. There was no detectable Cd or Cr in the samples so they were excluded from further study. Samples were collected prior to the runoff entering the pond, and then at different time intervals during ten rainfall events. Results showed higher Zn concentrations than expected which authors attributed to galvanized guardrails found on mountain roads. In addition, evidence of a first flush effect was observed. Because of the effect, the first half of the runoff volume transported between 61 – 69% of the contaminants. Authors also noted soil pH was important for controlling metal uptake by soils. Based on the results, Barbosa and Hvitved-Jacobsen (1999) concluded the best soil type for removing contaminants from infiltration ponds is one with a high sorption capacity and a high resistance to desorption at low pH.

A stormwater infiltration pond at the University Claude Bernard was the subject of a study by Datry et al. (2003). The infiltration pond at that time had been in use for more than 30 yrs with a 2 m layer of cobbles laid over fluvial sand and gravel sediments. It drained 2.5 ha, and received water from buildings, parking lots, roads, and lawns. The infiltration pond had a surface area of 750 m<sup>2</sup>, and a storage capacity of 4,000 m<sup>3</sup>. A series of pipes drained the catchment with one pipe responsible for 75% of the drainage. The water table under the infiltration bed was 1.5 m below ground surface during dry periods, and the regional water table was between 2.5 – 3.5 m below ground surface. Groundwater was monitored through a network of 30 piezometers at the main inlet set at a 1.3m depth. The water and sediment were analyzed for particulate organic carbon (POC), total N (TN), total P (TP), Cd, Cr, Ni, Pb, total hydrocarbons (THC), As, boron (B), cobalt (Co), manganese (Mn), molybdenum (Mo), selenium (Se), monocyclic aromatic hydrocarbons, polycyclic aromatic hydrocarbons (PAHs), halogenate aliphatic compounds, halogenated aromatic compounds, polychlorinated biphenyls (PCBs), phenolic compounds, and pesticides. Investigators found the sediment stored in the infiltration pond had high concentrations of POC, particulate N, TP, and TN, and were contaminated with hydrocarbons and heavy metals. Zinc accounted for 60%, Pb for 24%, and Cu for 11% of the total metal concentrations in the sediments. There were 33 organic compounds found in the sediments, 10 PCBs, 15 PAHs, and diuron. No concentration gradient was found moving further from the inlet in the monitoring piezometer network. The water in the infiltration sediment bed was enhanced in chemical composition compared to the stormwater composition, but hydrocarbons and heavy metals were not detected in the water during dry periods. During and shortly after rainfall events, the infiltrated water lowered the concentrations of most contaminants. Dissolved oxygen and redox potential (ORP) rose during and shortly after the rainfall event. The oxic conditions lasted only 1.2 days. Nitrate concentrations were much higher following the rain event, and metals and hydrocarbons were detected at low concentrations in some of the piezometers.

Datry et al. (2004) followed up Datry et al. (2003) in the same infiltration basin. A new nest of monitoring wells was installed 1.6 m from the main stormwater inlet. The wells were evenly spaced in a

1 m circle, and were installed at depths of 2, 3, 4, 5, 6, and 7 m below the surface of the infiltration bed. Each well, except the 7 m well, had a 0.5 m screen while the 7 m well had a 2.5 m screen. Well 1 was used to sample the stormwater circulating in the infiltration bed. Wells 3, 4, 5, and 6 collected groundwater at depths of 1, 2, 3, and 4 m below the surface of the infiltration bed. Well 2 collected water at the intersection of the infiltration bed and the fluvial sediments, and was dry throughout the study period. An additional set of wells was installed at a reference site. Wells 7, 8, and 9 collected groundwater at 1.5, 3, and 4 m below the groundwater table. The results for the sediment samples were similar to Datry et al. 2003. During dry periods, the water in the infiltration bed was enriched when compared to the water composition at the pipe inlet, which was similar to what was reported in the previous study (Datry et al., 2003). At the reference site, there were no differences in solute concentrations among depths. The SPC and the concentrations of  $\text{HCO}_3^-$  and Ca in the groundwater below the infiltration basin did not differ from the reference site during dry weather, but were significantly lower during rainfall events. The reference site concentrations of  $\text{SO}_4^{2-}$ , Cl<sup>-</sup>, silica, Mn, and Na were always lower in the groundwater at the infiltration site with greater difference during rainfall events. Hydrocarbons were detectable at the reference site during dry weather, but never in the groundwater below the infiltration basin. Groundwater below the infiltration site was enriched with  $\text{PO}_4^{3-}$  and DOC compared to the reference site. Vertical concentration gradients developed in the groundwater below the infiltration basin during rainfall events. For  $\text{HCO}_3^-$ ,  $\text{NO}_3^-$ , and DOC the vertical gradients were largest 12 hrs. after an event, dissipated gradually with time, but were still detectable after 108 hrs. Phosphate had the highest differences between depths at the end of the recharge event.

Sixteen stormwater detention basins were examined to determine if there were impacts to groundwater quality in an area of New Jersey with a shallow water table and sandy, unconsolidated soils (Fischer et al., 2003). The basins examined shared common depths, surface areas, and number and sizes of inlets. They were all similar in ground coverings, which consisted of grass and mixed weeds, and the basins were in urban watersheds that drained commercial, public, and residential areas. In each basin, a well was installed near the center by hand auguring to a depth of 1 m, and the wells were 1.9 cm in diameter and stainless-steel construction. The water was sampled for metolachlor, prometon, atrazine, desethyl-atrazine, simazine, dieldrin, carbaryl, chloroform, MTBE, toluene, benzene, PCE, 1,1,1-TCA, carbon disulfide, DO,  $\text{NH}_3$  + organic N, and  $\text{NO}_2^-$  +  $\text{NO}_3^-$ . The results of this study revealed DO concentrations in the study wells were lower than background groundwater, and the study wells had a higher detection frequency for benzene and toluene. Metolachlor, prometon, and carbaryl detection frequency was larger in the basin wells than in the background groundwater in the winter and summer. Atrazine was detected more frequently in the study wells in the summer than in the background wells, whereas desethyl-atrazine, simazine, and dieldrin were detected more frequently in the background wells than in the basin wells.  $\text{NH}_3$  + organic N was detected more frequently in the basin wells than it was in the background wells. Conversely  $\text{NO}_2^-$  +  $\text{NO}_3^-$  was detected more frequently in the background wells than in the basin wells.

Dechesne et al. (2004a) examined an infiltration basin that drained water from a 7 ha truck parking lot with the capacity to hold 2,616 m<sup>3</sup> of stormwater. The basin had permeable calcareous sand, pebbles, and rocks. Research focused on soil sampling at 10 points and at several depths in an xy-grid to determine the concentrations of various contaminants including THC, six PAHs, Cd, Cr, Cu, Ni, Pb, Zn, and As. The results showed after 14 years of operation, THC and PAH concentrations were highest in the upper portion of the vadose zone and quickly decreased with depth. They were not detectable at a 1 m depth. Zinc however, was not well retained the upper 1 m of the vadose zone and was determined to be very mobile. The other metals studied were retained in the upper portion of the vadose zone and rapidly decreased in concentration with depth.

Dechesne et al. (2005) built upon the Dechesne et al. (2004a) study using the same basin with a reported depth of groundwater at 4.0 m and three additional basins with fluvioglacial soils that were highly calcareous with low organic matter. The first new infiltration basin was in an urban use area. This basin drained 33.7 ha with a water holding capacity of 3,800 m<sup>3</sup>. The depth to the water table was 2.8 m. The second new infiltration basin was also in a mixed urban use area, drained 74 ha, and had a water holding capacity of 20,700 m<sup>3</sup>. The depth to the water table was 10 m. The final infiltration basin was in a mixed urban use area. This basin drained 50 ha, and had a water holding capacity of 7,960 m<sup>3</sup>. The depth to the water table was 10 m. Ten soiling sampling points were made in each basin and collected at four different depths. Researchers only looked at THC for organic analysis. There were significant concentrations of THC in the upper portions of the vadose zone, and THC was still detectable to a depth of 0.03 – 0.40 m. As was the case in the earlier study, zinc was very mobile, but even after 21 yrs. zinc did not go more than 0.50 m into the vadose zone. The other metals were concentrated in the upper layers of the vadose zone, and rapidly decreased in concentration with depth and did not extend below 0.30 – 0.40 m in depth.

Birch et al. (2005) examined the ability of a stormwater infiltration basin to remove contaminants from urban stormwater. The basin was layered with filtration media of zeolite and coarse quartzitic sand, but there was no information given on the deep geological layers. The site was in an urban suburb in eastern Sydney, Australia and was monitored over nine rainfall events from the outflow drain. The total area of the basin was 450 m<sup>2</sup>, the catchment area was 2.668 ha, and it drained an urban area with terrace houses, streetscapes, and parklands. Authors measured the concentrations of TSS, TP, total kjeldahl nitrogen (TKN), Nitric oxide (NO<sub>x</sub>), TN, Cd, Cr, Cu, Fe, Mn, Ni, Pb, Zn, organochlorine pesticides, and fecal coliforms. Results showed decreases in the concentrations of TSS, Cu, Pb, Zn, fecal coliforms, TP, and TKN. The basin was ineffective at reducing TN, but authors did note a conversion of TKN to NO<sub>x</sub> was possible. Concentrations of Cr, Fe, and Ni did increase, and authors believe this may be due to leaching of clay minerals from the filter bed.

Winiarski et al. (2006) studied an infiltration basin in France with a fluvio-glacial complex in the underlying aquifer. This infiltration basin was approximately 5.5 m deep with a surface area of 7,406 m<sup>2</sup>, and held up to 30,856 m<sup>3</sup> of stormwater. The depth to the water table was approximately 13 m. The infiltration basin primarily drained an industrial area, and was in operation for about 20 years at the time of publication. Soil samples were taken near the inlet pipe, the infiltration basin, in the middle of the infiltration basin, and point a distance away from the stormwater inlet pipe. The soil samples were taken every 0.05 m to 0.10 m, every 0.1 m to 0.5 m, and every 0.2 m to a depth of 4 m. The metals of interest were Pb, Cd, and Cu as well as bacterial counts. The samples nearest the inflow were found to have the highest concentrations of Pb and Cd near the surface which then decreased to background concentrations at 1.5 m below the surface. The pH of the soil was lowest at the surface and then increased in a pattern similar to the decrease in Pb and Cd concentrations to 1.5 m below surface. Total organic carbon (TOC) and silt content followed a pattern similar to Pb and Cd concentrations. At 2.4 – 3 m the sample in the middle of the infiltration basin showed an increase in metal concentrations with decreasing pH while TOC and silt content increased. The point a distance away from the inlet showed little influence by the infiltration water and the profile was similar to the background sample profile. The bacterial populations were 10<sup>4</sup> – 10<sup>7</sup> CFU/g at the three sampling locations. The bacterial populations were found to decrease with depth with approximately a two-log concentration difference from the surface to the deepest depth. The authors concluded metals were concentrated in the upper portion of the vadose zone after 20 years of infiltration due to the presence of carbonates that can either precipitate metal carbonates or exchange metals and retain the metals. Since the metal concentrations were correlated with silt particles and organic matter, they theorized the metals were possibly transported in the colloidal fractions and that filtration could be a mechanism for the retention of metals in the upper portion of the vadose zone.

Another stormwater detention pond was studied by Mayer et al. (2008). The capture zone of this pond was 130 ha and 80% of the water it received was road runoff. The pond was approximately 300 m long and 25 – 40 m wide. It was made up of two parts: a sediment forebay and settling pond, but the underlying geology was not discussed. The authors investigated the sediment in the pond and the porewater at the deepest point in the settling pond, approximately 4 m depth. To examine porewater, dialysis membrane samplers were used with cells placed 1-cm apart. Soils were collected that were 15-20 cm long using gravity corers. Results indicated  $\text{Cl}^-$  from the road salt increased, metal chloride complexes increased, and that certain metal chloride complexes would be stabilized in solution. Metal chloride complexes were especially of concern for Cd.

Mermillod-Blondin et al. (2008) looked at potential effects of bioturbation of tubificid worms and stormwater sediment (SWS) thickness on the release of nutrients and contaminants in stormwater retention systems. The authors collected SWS and tubificid worms and transferred them into 18 columns. Six had a SWS layer of 2-cm, six had a 5-cm thick SWS layer, and six had an 8 cm thick SWS layer. Experiments were conducted for 24 days (10 days before worms, and 14 days after worms were introduced). Results showed nutrient release increased with SWS thickness and bioturbation, and the release of two PAHs (acenaphthene and naphthalene) was influenced solely by SWS thickness. There was no evidence to suggest that bioturbation of the tubificid worms influenced contaminant mobility. Based off these findings, authors concluded stormwater pond management should include a control for SWS accumulation and biological activity to limit the release and mobility of contaminants and nutrients to prevent potential disruption of nearby ecosystems.

A research study conducted by Camponelli et al. (2010) investigated Cu and Zn in sediments below a stormwater retention pond. This pond drained a six-lane highway and had an approximate surface area of 446 m<sup>2</sup>, but the authors did not report on the local geology under the retention pond. Runoff water samples were collected from twelve storm events at the culvert that enters the pond. Soil samples were collected along three transects, with seven samples per a transect. Retention pond sediment cores were collected over six transects with two samples per a transect. Results showed stormwater draining into the retention pond sometimes exceeded US EPA water quality criteria. In the vadose zone the top 10 cm had the highest concentrations of Cu and Zn, and Cu and Zn concentrations dropped with increasing depth. The variability in the decrease of Cu and Zn concentrations in the vadose zone was due to heterogeneity of the sediments.

Denitrification rates were studied in a managed groundwater recharge system (Schmidt et al., 2011). It sat over a perched aquifer within an eolian, fluvial, and alluvial sediments. The system consisted of a 3 ha infiltration pond that was created in a natural depression and had a maximum depth of 6 m. Infiltration caused a wetting front and an inverted water table downward into the vadose zone forming a 1 – 2 m saturated zone. Infiltrating water then passed 20 – 30 m of vadose zone before reaching an aquifer. The water from the infiltration pond was collected from under the infiltration pond by twelve piezometer nests that were screened between 30, 50, 60, 90, and 100 cm. The study found the dominant form of nitrogen was  $\text{NO}_3^-$  in the pond water. The  $\text{NO}_3^-$  concentrations in the saturated zone below the pond vadose zone interface was less than the pond concentrations for all depths, indicating that denitrification is a significant process for N inputs from a managed aquifer recharge system.

Mermillod-Blondin et al. (2015) examined the ability of three managed aquifer recharge (MAR) systems to process dissolved organic matter (DOM) moving from infiltration basins to urban aquifers. These underlying aquifers consisted of fluvio-glacial carbonated sediments, consisting of quartz, carbonates and trace clays chlorite and anorthite. Fluorescent spectroscopic properties, biodegradable DOC and refractory fraction of DOC, and DOC consumption by microorganisms were monitored during a

stormwater event. Samples were collected for 24 hrs. after the stormwater event at 50 cm below the basin surface from recharge and reference wells. Results revealed a large decrease in DOC concentrations at all three MAR systems. Retention was higher for biodegradable DOC versus refractory DOC which led authors to conclude that microbial processes were dominant in DOC removal during infiltration. However, even with the high retention rates of the DOC, results showed DOC levels in the aquifer increased after stormwater runoff infiltration.

Al-Rubaei et al. (2017) investigated the performance efficiency of a 19-year-old combined pond-wetland system for surface water quality and quantity treatment and no information was given on the geology of the site. They monitored the removal of TSS, nitrogen compounds, phosphorus, and heavy metals over the course of four seasons through water and sediment sampling. Water was sampled for stormwater inflow, after pre-treatment in the pond, and the discharge from the wetland. Sediment samples were taken at nine locations using a Van Veen grab or plastic scoop. They identified the main factors affecting system performance: season, air temperature, antecedent dry period, rainfall depth/intensity, and duration of storm events. The wetland system studied was built to reduce pollution flowing to a lake. The catchment area of the wetland had 130 ha of residential area, 190 ha of industrial/commercial area, and major roads. A total of 53 storms were monitored for flow, and 13 were sampled and analyzed for water quality. Results showed the mean volume reductions were 40% for the pond, 28% for the wetland, and the pond-wetland system decreased peak flows by 41 – 95%. The system also removed an average of 91% of the TSS, 80% of the phosphorus, 94% of the particulate Cd, 91% of the particulate Cu, 83% of the particulate Pb, and 92% of the particulate Zn. Removal of Ni and nitrogen compounds was highly variable. Results also showed removal of dissolved heavy metals were less efficient and more variable.

Stormwater infiltration basins were the most commonly assessed GI for groundwater contamination found in the literature, but much of the literature focuses on surface water analysis or basin soil assessment. There were a wide variety of monitoring methods and variables assessed for effects on groundwater. Metals were found to be retained in the soils or vadose zone of the infiltration pond, but often metal transport did not go deeper into the groundwater (Nightingale, 1987a, 1987b; Datry et al., 2003; Dechesne et al., 2005; Winiarski et al., 2006; Camponelli, et al., 2010). Other studies solely assessed the reduction of contaminants in the surface water (Barbosa and Hvitved-Jacobsen, 1999; Al-Rubaei et al., 2017). If reduction in contaminants are found, it can be deduced that the contaminants remain in the infiltration basin and could act as a potential groundwater contaminant over time. This issue has yet to be addressed in the literature. It was also found that denitrification could be a significant process in managing nitrogen in stormwater infiltration basins (Schmidt et al., 2011), but there is a risk of nitrogen leaching out of the sediments during recharge events (Nightingale and Bianchi, 1977). Other studies observed the risk of groundwater contamination from organic compounds and found that 100% of organic contaminants cannot be removed in infiltration basins leading to increased groundwater detection in some cases, but their detectability decreased with depth in the vadose zone (Tomson et al., 1981; Datry et al., 2003; Fischer et al., 2003; Dechesne et al., 2004a and 2005). It seems that stormwater infiltration basins can be successful at removing several contaminants from stormwater, but 100% of these contaminants will not be removed and will concentrate in the vadose zone. Long term implications for this have not been explored, causing concern for future changes to groundwater.

### **7.2.2 Roadside Stormwater Runoff Infiltration Systems**

Another type of infiltration system used to decrease stormwater runoff are roadside stormwater runoff infiltration systems. In this system, runoff from roadways is collected and infiltrated into the subsurface.

The following discussion will examine the potential for groundwater impacts from the use of these systems.

Two old infiltration systems were investigated for potential contamination from Cr, Co, Ni, Cu, Zn, Cd, Pb, absorbed organically bound halogens (AOX), and PAHs (Mikkelsen et al., 1996). The first site was in an area used for light industrial activities, mixed residential, and agricultural land use. The infiltration system was a series of depressions in the road shoulder which had been in operation since 1959. The depth to groundwater was 15 – 20 m, and the vadose zone was comprised of gravel. The second site was in a heavily trafficked area with a waste incineration plant 1.2 km from the site. At this site, road runoff was drained into infiltration shafts which were built in 1949 and 1982. The groundwater depth at this location was 20 m, and the vadose zone was also comprised of gravel. At the first location, researchers reported high concentrations of total and dissolved metals in the upper 30 cm which decreased to background concentrations further down in the vadose zone. PAH and AOX concentrations were highest in the upper layers but were near background concentration by 1.5 m. In both shafts at the second location, the runoff sludge had infiltrated to a depth of 70 cm in shaft one and 30 cm in shaft two. The researcher found the sludge in shaft one was anaerobic and had a sulfide and petroleum smell. Shaft two, was half full of water because it was clogged and anaerobic. In both shafts, the total concentrations of metals, PAHs, and AOX correlated with the extent of the sludge layer. The researchers concluded that runoff from a large area was concentrated into a small area which caused a significant buildup of metals, PAHs, and AOX in upper soil layers and sludge layers. Researchers also point out the results of this study may not be applicable to other sites with different soil characteristics and geology, so there is a need for more field research investigating stormwater infiltration.

Mikkelsen et al. (1997) is a continuation of the research at the study sites used in 1996. At the first site, results indicated the top layer in the vadose zone was formed by runoff born particles. The contamination was retained in this layer by filtration of particles in the runoff, and by sorption of contaminants to the particles in this layer. Any contaminants that passed through this layer also passed through the sub-base gravel and were retained in upper sediments of the vadose zone. The second site patterns were more difficult to ascertain according to the researchers; they found pH increased with depth, which slowed the downward migration of metals through sorption and precipitation. However, precipitation of metal sulfides and reductive dissolution of Fe and Mn oxides could possibly account for the variable depth profiles observed. This research concluded, based on the depth to groundwater, groundwater contamination is unlikely in a reasonable time frame. However, the authors noted that soluble inorganic and organic contaminants were not studied and could potentially reach groundwater.

Norrström and Jacks (1998) examined the effects of stormwater runoff and the use of de-icing salts on groundwater under an infiltration pond near a roadside built in 1964. The study site was in the recharge area for the reserve water supply of a nearby town. Prior to the road being built in 1964, the groundwater Cl<sup>-</sup> concentrations were 15 mg/L. In 1998, the Cl<sup>-</sup> concentrations were 120 mg/L, and were shown to be the result of road salt application. In thirty-four years, the Cl<sup>-</sup> concentrations increased eight-fold.

Stephenson et al. (1999) designed a stormwater runoff system to convey stormwater from a highway and drain it into a sinkhole, thereby directly into a karst aquifer. In this system, they drained three different portions of the highway into one sinkhole. Most of the drainage volume went through a 107-cm concrete culvert. The second drainage route was a 1.8 m concrete ditch, and the final drainage route was through a 46-cm corrugated metal culvert. The groundwater quality was monitored at a nearby spring for Zn, Pb, TPH, PAH, TDS, and total volatile solids (TVS). They found detectable Zn, TPH, TDS and TVS in the only stormwater runoff sampling event monitored. The authors concluded the contaminant load was closely related to the runoff volume rather than the concentration of the contaminants. The authors also

stated that contamination of groundwater is enhanced in karst areas because there is a more direct path to the groundwater.

A roadside infiltration trench used to infiltrate stormwater into the subsurface was studied by Norrström (2005). Three piezometers were installed at 2.5 and 4.5 m below land surface near the infiltration trench. Both whole samples and dissolved samples (filtered to  $< 0.45 \mu\text{m}$ ) were taken. The researcher looked at the effects of road salt on the infiltration of metals to groundwater. Results indicated that Cd and Pb content was moderately high in the groundwater and was primarily associated with the particulate fraction. Zinc on the other hand was, to a lesser extent, bound to the particulate fraction. The results indicate the greatest risk to groundwater in areas subjected to road salt was the colloidal transport of Pb. Cadmium and Zn were likely transported as chloride complexes and free ions. The factors that primarily modify the transport of these metals are soil properties, accumulation of metals in the vadose zone, and the hydrogeology.

The leaching behavior of organic contaminants from ditches that received stormwater runoff from roadways was investigated by Strömvall et al. (2007). This study looked at PAH content and SVOCs in soil cores taken from the ditches. Research indicated near surface concentrations were the highest for both PAHs and SVOCs, and concentrations decreased with increased depth. Semi-volatile hydrocarbons were detected to a depth of 1 m, and PAHs were detected to a depth of 1.5 m. The researchers hypothesized that these organic compounds were likely transported into deeper zones through colloidal transport with the compounds sorbed to colloids.

Kluge et al. (2014) also investigated metal concentrations (Cd, Cr, Cu, Ni, Pb, and Zn) in embankments along a highway. In this study, wick lysimeters were used to sample roadway runoff, and soil cores were used to examine metal concentrations with depth. The runoff samples were subsequently split into two samples: one not filtered and one filtered with a  $0.2 \mu\text{m}$  filter for dissolved metals. The  $>0.2 \mu\text{m}$  fraction was considered the particulate bound metals. For metals Cr, Cu, Ni, Pb, and Zn, the particulate bound fraction was the dominant fraction. However, for Cd the dissolved fraction was the dominant fraction. In the vadose zone there was a horizontal gradient except for Cd. Metal concentrations were the largest near the highway and decreased in concentration as distance from the highway increased. In the case of Cd, samples at 2 m had significantly lower Cd concentrations than at 5 m. From a 5 m distance, there was a decrease in Cd concentrations with distance from the highway. The vertical gradient showed concentrations for all metals were highest in the upper portion of the vadose zone and decreased with depth.

Aryal et al. (2006) studied the Experimental Sewer System constructed in 1986 in a highly urbanized residential area. The drainage area was 8.25 ha with 244 infiltration trenches and inlets. The researcher sampled all the infiltration inlets, but only data from 17 of the infiltration inlets were used in this investigation. They also collected roadside dust from all locations. The heavy metal content of the dust and sediments were similar in concentration, but were higher than soil metal concentrations in the area. They concluded that sediments at a 3-cm depth were different than sediments at the other depths, that road dust was the source of metals in the infiltration inlets, and the decreasing metal content with depth could be due to metal leaching to underlying sediments. The positive correlation of Cu and Zn indicates a common source, but the other metals were location specific.

The research presented in this study (Aryal et al., 2007) was an extension of the research conducted earlier (Aryal et al., 2006). In this study, they present more detailed data on two of the 17 cores. The cores were chosen because of the thick sediment above the mouth of the trench. The continuation of this research was in part because of the unreliability of using total metals as a surrogate for metal mobility

and contamination. Researchers used a sequential extraction method to parse out the metal concentrations in various fractions on the sediment layers. The authors concluded metal concentrations decreased with increasing depth, and speciation of the metals in the sediments showed different orders of mobility. The first conclusion suggested metals were possibly released to lower layers of the vadose zone or to groundwater. The second conclusion applied to the long-term fate and transport of the metals in the vadose zone for several reasons. If the pH of the sediments became more acidic then Zn, Mn, and Co would be more mobile. If the sediments become more reduced, then Pb, Mn, and Co become more mobile, but if the sediments become more oxidized, Cu, Pb, and Cr become more mobile.

Azah et al. (2017) examined potential risks of roadway and stormwater residuals for 16 PAHs in samples collected from Florida. Samples consisted of street sweepings, catch basin sediments, stormwater pond sediments, and ditch cleaning sediments from 14 municipalities. The results showed that of the 16 PAHs, benzo [a] pyrene was the only one detected above appropriate risk-based regulatory values for soil and groundwater. The least detected PAH was acenaphthylene. Based on the results, authors concluded there may be beneficial use for ditch cleanings, stormwater pond sediments, and catch basin sediments in industrial and residential settings, but street sweeping debris may pose a threat to residential areas. They also found the residuals used in the study were unlikely to leach PAHs into the groundwater. As was the case with infiltration/retention ponds, roadside stormwater runoff infiltration systems did not show impacts to groundwater quality in most cases, and the contamination was generally confined to the upper portion of the vadose zone. Norrström and Jacks (1998), Stephenson et al. (1999), and Norrström (2005) did show roadside stormwater runoff infiltration systems could cause changes to groundwater quality from significant increases in  $Cl^-$  concentrations, and metal and organic contamination. It is important to note that in most cases contamination did not occur. Although the contaminants appeared to be concentrated in the upper portion of the vadose zone and decreased with depth, this could indicate that leaching of the contaminants was occurring and there could be a future impact to groundwater quality.

For roadside runoff infiltration systems, results could vary based on differing soil conditions and geology. This suggests a need for more studies across various conditions. Most of these studies showed that contaminants tend to concentrate into small areas in the upper layers of the soils and most did not assess contamination of the groundwater below these systems. With insoluble contaminants, it is unlikely that groundwater will be contaminated, but there is a concern with colloidal transport from road salt could occur if the contaminants sorb to the colloids. Areas with karst aquifers are at risk because the direct path to the groundwater could mean biogeochemical processes are bypassed and not removing contaminants.

### **7.2.3 Permeable Pavement Systems**

Permeable pavements are made from construction materials that have large enough pores to allow water to flow through, or use paver systems that allow water to pass in the space between pavers. The permeable pavement systems allow stormwater to flow from the surface, and in some cases, enter a reservoir (infiltration gallery) that stores the stormwater for a short period of time while infiltration occurs. Groundwater quality could potentially be impacted by permeable pavement systems, which will be discussed in this section.

Permeable pavement systems act as in situ bio-reactors that contain diverse communities of bacteria, fungi, protozoa, and other invertebrates. The native microbial communities have been shown to remove more than 98% of hydrocarbons (e.g. from motor oil and fuels) from infiltrated stormwater runoff (Coupe et al., 2003; Scholz and Grabowiecki, 2007). Engineers have promoted this beneficial

bioremediation process through the incorporation of nutrient-releasing geotextiles within permeable pavement systems (Spicer et al., 2006). Despite these studies, very little information is available on the fate and transport of microbial contaminants through porous pavement systems. As unsaturated systems, stormwater microorganisms would likely be subjected to the removal mechanisms discussed above for saturated and unsaturated systems.

The characterization of particles that could potentially clog porous pavements or be transported to groundwater was conducted by Colandini et al. (1995). Particle size distribution, heavy metal content, and interactions between metals and particles were investigated in this study. The authors took samples from pervious asphalt pavements used for highway and motorway constructions and porous pavements with a reservoir structure. Samples were collected by spraying high-pressure water on the pavement and then collecting the sludge through depression-suction. Results showed most of the clogging materials were sand with a variable silt content and low clay content. The researchers also found the materials were often contaminated with heavy metals (Pb, Cu, Cd, Zn). Porous pavements with reservoir structures also showed lower heavy metal content in clogging particles, however, all particle sizes are contaminated with heavy metals, with fine particles showing the highest metal concentrations.

Brattebo and Booth (2003) studied the long-term effectiveness of four commercial permeable pavement systems for structural durability, ability to infiltrate precipitation, and water quality of the infiltrated water compared to traditional asphalt. The four different systems used were Grasspave<sup>®</sup>, Gravelpave<sup>®</sup>, Turfstone<sup>®</sup>, and UNI Eco-Stone<sup>®</sup>, and construction occurred in 1996. After six years, the authors found no major signs of wear and tear, and virtually all rainwater infiltrated into the permeable pavements with very little surface runoff. Results showed infiltrated water had lower levels of Cu and Zn than direct runoff from the asphalt, however, concentrations of Zn increased during the five years for infiltrated water and direct runoff. Authors also found 89% of the samples from the asphalt had motor oil, but none of the samples from the infiltrated water had any. Neither lead or diesel fuel were detected in any sample, and conductivity and hardness stayed constant.

Gilbert and Clausen (2006) compared replicated asphalt, permeable paver, and crushed-stone driveways for water quality and quantity from stormwater runoff. Runoff samples were taken weekly, and rainfall was measured for 22 months. Contaminants analyzed were TSS, TKN, NO<sub>3</sub>-N, NH<sub>4</sub>-N, TP, Zn, Pb, and Cu. The sites chosen were driveways in residential areas. Results showed runoff from asphalt driveways was much higher, while crushed stone driveways had the least amount of runoff. Permeable paver driveways had the lowest concentration of contaminants, and crushed-stone driveways and asphalt driveways had similar levels of contaminants. The authors also noticed infiltration rates for the permeable paver and crushed-stone driveways decreased over time, but stated they believed the decrease was due to particle clogging. They concluded permeable pavers and crushed-stone driveways are preferable to asphalt for pollution control even with the decreased infiltration rates.

Boving et al. (2007) conducted a study to investigate the performance of permeable pavement, and to determine the fate and transport of metals, nutrients, PAHs (naphthalene, acenaphthylene, acenaphthene, fluorene, phenanthrene, anthracene, fluoroanthene, pyrene, chrysene, and benzo (a) pyrene), and bacteria in a functioning parking lot. The site was a porous asphalt parking lot over a regional aquifer. It was built in 2002, had 800 parking spaces, and an area of 24,200 m<sup>2</sup>. Results showed the pavement clogged with sand in heavy traffic areas, and areas that remained porous were effective at removing organic and metal contaminants, but not nutrients and anions. They also found contaminant concentrations varied with the seasons, i.e., higher metal and chloride concentrations in the winter and early spring due to road salting. PAH flux was lower than typical reported concentrations in Rhode Island suggesting the structure was effective in removing PAHs.

A study was conducted during 2007 to look at water quality differences between four different types of permeable pavement and standard asphalt (Collins et al., 2010a). The permeable sections studied were pervious concrete, permeable interlocking concrete pavement (one with an open surface area of 12.9%, and the other 8.5%), and concrete grid pavers filled with sand. Composite samples of atmospheric deposition and asphalt runoff were compared to subsurface samples from the permeable pavement sections. The authors focused on different nitrogen species (TN, TKN, organic N (ON), NO<sub>2</sub>-N, NO<sub>3</sub>-N and NH<sub>4</sub>-N) and pH. Samples were collected from 20 different storm events at seven different locations (atmospheric deposition, two asphalt runoff areas, and the four permeable pavement areas). The results showed permeable pavement drainage had higher pH than the asphalt runoff, and the pervious concrete had the highest pH of all the permeable pavement sections. Permeable pavements had more buffering capacity, however all pavements were able to buffer acidic rainfall. Permeable pavement drainage also had lower NH<sub>4</sub>-N and TKN concentrations and higher NO<sub>2</sub>-N and NO<sub>3</sub>-N concentrations.

Borst and Brown (2014), examined three permeable pavement systems in relation to potential road salt effects on groundwater quality. This study was conducted at a 0.4 ha parking lot. The three permeable pavement systems used were interlocking concrete pavers, pervious concrete, and porous asphalts installed into three separate rows. The driving lanes were paved with traditional impervious hot mix asphalt. Each pavement system received direct rainfall and runoff from the driving lanes immediately upgradient. Under each row was an infiltration gallery constructed of recycled crushed concrete aggregate. The water collected in the infiltration gallery was routed to collection tanks with a capacity of 3.4 – 4.1 m<sup>3</sup>. Calcium chloride (CaCl<sub>2</sub>) was applied to the parking lot surface for de-icing. Although no vadose zone samples or groundwater samples were taken, the collected runoff samples were useful for demonstrating the potential environmental effects to the vadose zone and groundwater. The results indicated that Cl<sup>-</sup> concentration from the winter salt application was attenuated as it passed through the permeable pavement system, but the Cl<sup>-</sup> concentration was still significant. After April, the Cl<sup>-</sup> concentrations were detectable, but did not exceed 230 mg/L.

Drake et al. (2014) conducted a study to compare three different partial infiltration permeable pavement systems and a standard asphalt system on their ability to trap pollutants and improve water quality. The three permeable pavement systems used were AquaPave<sup>®</sup>, Eco-Optiloc<sup>®</sup>, and Hydromedia<sup>®</sup>. The contaminants they focused on were TSS, TPH, Cu, Fe, Mn, Zn, TN, and TP. The pavement cells were 230 - 233 m<sup>2</sup> in size, and had a parking capacity for 8-10 vehicles. The cells were also separated by concrete to prevent cross-flow of stormwater. The authors found the different permeable pavement systems performed similarly and reduced TSS by >80%. Results also showed the effluent from the permeable pavement systems contained less heavy metals (65-93% removal), and reduced the concentration for most nitrogen species (decreased concentration of NH<sub>3</sub>, NO<sub>2</sub><sup>-</sup>, and ON, but increased levels of NO<sub>3</sub><sup>-</sup>) and phosphorus in comparison to the standard asphalt.

Brown and Borst (2015) studied three permeable pavement types: interlocking concrete pavement, pervious concrete, and porous asphalt, located at a 0.4 ha parking lot in Edison, NJ. They investigated how these three different pavement types would influence nutrient concentrations in stormwater runoff. Thirteen rainfall events were sampled over a year with infiltrate samples collected directly from the tanks below the pavement systems. It was found that permeable pavement systems were unlikely to reduce the TN in stormwater due to lacking anaerobic conditions. Different pavement types did affect the amount of bioavailable phosphorous with the most reduction seen in porous asphalt. The authors suggest a need to research the different permeable pavement construction materials to assess the risk of leaching from the materials. There is also a concern of nitrogen contamination from organic material when runoff is from vegetated areas. This organic bound nitrogen could clog the pavement surfaces and be a contaminant risk to groundwater if it infiltrates through to non-anaerobic conditions.

Most cases of the permeable pavement systems research looked at infiltrated water rather than the water traveling from the system through the vadose zone. Researchers believed for the most part that contaminants would not impact groundwater because contaminants were particle bound or retained in the infrastructure. However, in two studies there was a potential risk to groundwater quality from  $\text{Cl}^-$  (Borst and Brown, 2014) and  $\text{NO}_3^-$  and phosphorous (Drake et al., 2014).

#### **7.2.4 Roof Runoff Systems**

Another type of GI is the roof runoff system which captures stormwater running off roofs and infiltrates the runoff into the subsurface. The potential for groundwater contamination is discussed next.

Roof runoff infiltration was investigated by Mason et al. (1999) at an industrial site using large building roofs in which the roof runoff was channeled into a single PVC pipe that emptied into an infiltration pit with an area of  $106 \text{ m}^2$ , and was 3 – 4 m in depth. The approximate volume of the infiltration pit was  $150 \text{ m}^3$ . Researchers sampled the infiltration pit and lysimeters at depths of 1 m and 1.6 m and analyzed for nutrients ( $\text{NH}_3$ , ortho-P,  $\text{NO}_3^-$ , TN, and DOC), heavy metals (Cd, Cr, Cu, Pb, and Zn), major anions ( $\text{Cl}^-$  and  $\text{SO}_4^{2-}$ ), and major cations (Ca, Mg, Na, and K). The results showed that for  $\text{NO}_3^-$  and ortho-P in the lysimeters, samples followed the concentration patterns in the roof runoff, and the higher initial concentrations were later diluted by the soil water or the large flush of runoff later in the event. This indicated there was not an effective retention mechanism for ortho-P or nitrate in the vadose zone. For DOC there was slight retention in the vadose zone.  $\text{NH}_3$  showed significant decreases in concentration further down in the vadose zone which was attributed to possible denitrification processes and sorption to negatively charged surfaces. The dissolved metal fraction ( $<0.45 \mu\text{m}$ ) was the focus for the transport of metals in the vadose zone because the researchers assumed that particulate bound metals would be trapped in the upper layers of the vadose zone. Cadmium, Cr, Cu, and Zn were primarily associated with the dissolved phase in study, whereas, Pb was found to be associated with the particulate phase. Researchers considered the metal accumulation in soil water in the upper portion of the vadose zone, metals in the soil water in the unsaturated zone, and metal accumulation in the vadose zone. In the soil water in the upper portion of the vadose zone, the concentrations were much lower compared to metal concentrations in the soil water at 1 m and 1.6 m. This indicated a significant portion of the metals were transported downward through the vadose zone. They concluded this was likely caused by dilution of the infiltrating water, and uneven water distribution in the infiltration pit causing inadequate contact of inflow water in the upper layers of the vadose zone. In the unsaturated zone, researchers found there were two different behaviors for the metals. Initially concentrations for Cu, Cd, and Cr were high, and then their concentrations decreased which followed the trend found in inflow water. They were only found to be partly retained in the vadose zone, and a significant fraction was transported into lower portions of the vadose zone. Zn and Pb did not follow the inflow water trend. Instead, Zn and Pb concentrations decreased quickly even when their concentrations were high in the inflow water. This behavior showed retention of Zn and Pb in the vadose zone took place, at least in the short term. Finally, researchers looked at the accumulation of these metals in the vadose zone during a two-year period at depths of 0 -0.05, 1, and 1.6 m. They found that, except for Cr, all metal concentrations increased in the upper depth during the first four months, and Zn and Cd increased more than Cu and Pb. This showed there was some retention of the metals except Cr initially. In 1994, the researchers found metal concentrations were unevenly distributed in the various depths in the vadose zone, with the largest concentrations in the deepest portion (1.6 m) of the vadose zone. However, by 1996 the metals were evenly distributed between the 1 m and 1.6 m zones. There appeared to be a temporary accumulation of some metals in the upper portion of the vadose zone to a depth of approximately 1 m. As time passed there was a downward migration of metals deeper into the vadose zone. Chloride and  $\text{SO}_4^{2-}$  behaved as conservative tracers and did not react with the components of the vadose zone. In fact, the authors used

these species as internal tracers for the study. The major cation species on the other hand were found to interact considerably with the matrix in the vadose zone, and Ca concentrations in the lysimeters were found to be in equilibrium with calcite. The authors concluded major cations approached the typical composition of water in contact with carbonate rocks.

Farreny et al. (2011) investigated four different types of roofs (clay tiles, metal sheet, polycarbonate plastic, and a flat gravel roof) for their ability to harvest stormwater runoff. They compared slope and roughness of the roofs, and analyzed the physicochemical contamination of the roof runoff. Results showed smooth sloping roofs could collect up to 50% more rainwater than flat rough roofs. They also showed flat roofs had higher concentrations of contaminants except for NH<sub>3</sub> due to weathering, particle deposition, and plants.

There is a potential for groundwater quality changes with the use of roof runoff systems. The research presented by Mason et al. (1999) indicated that metals are transported to the vadose zone, and with time, metals move downward in the vadose zone. However, Cl<sup>-</sup> and SO<sub>4</sub><sup>2-</sup> behaved as conservative species and were not retained in the vadose zone.

### **7.2.5 Bioretention Systems**

Biofilters are engineered infiltration systems typically consisting of a water retention or ponding zone, vegetated surfaces, subsurface filtration media, and a drainage layer. Rain gardens, bioswales, and bioretention ponds are all examples of biofilters. These systems are typically subdivided into two groups based upon whether the infiltration design incorporates a submerged (also known as a saturated) zone within the vertical profile (Rippy, 2015; Peng et al., 2016). Submerged zones maintain moisture within the infiltration profile between precipitation events. Designs that incorporate a submerged zone typically include labile organic carbon additions to the saturated section to promote microbial denitrification for the remediation of influent nitrate. This anaerobiosis in the submerged zone contrasts the aerobic unsaturated conditions at the surface of the biofiltration system.

Vegetated infiltration systems enhance removal of introduced microbial contaminants by maximizing the antagonistic pressure from the native communities. Rain gardens and bioswales can be managed to enhance populations of mesofauna and protozoan grazers within the infiltration systems to promote greater removal of introduced stormwater microbial contaminants (Mehring et al., 2015; Chandrasena et al., 2017; Pavao-Zuckerman and Sookhdeo, 2017). The antagonistic biological processes together with the physicochemical discussed earlier result in greater stormwater microbial contaminant removal in vegetated infiltration systems (Mehring et al., 2015; Rippy, 2015; Petterson et al., 2016; Richkus et al., 2016; Chandrasena et al., 2017).

Hares and Ward (1999) conducted a study to compare wet biofiltration and dry pond treatment facilities and their removal efficiencies for selected stormwater contaminants. They monitored the concentrations of vanadium (V), Cr, Mn, Co, Ni, Cu, Zn, Mo, Cd, antimony (Sb), and Pb. The sites chosen were along a motorway which had a high daily traffic density. Samples were taken during dry weather conditions and during initial stages of storm events. Results showed the two ponds had similar discharge volume, but the biofiltration pond had higher heavy metal removal efficiencies than the dry pond treatment facility. The performance and efficiency of bioretention systems for removing heavy metals and nutrients were studied by Davis et al. (2001) using synthetic stormwater runoff. The synthetic stormwater was prepared so that it mimicked actual urban stormwater runoff concentrations, and results indicated that adsorption of heavy metals to soil increased with increasing pH. They found very little adsorption at pH≈4, and the release of Cu and Zn at very low pHs. The authors also found Pb had the strongest adsorption to the soil whereas Zn was the weakest. Results showed high reductions in Pb, Cu,

and Zn (>92%), and moderate reduction for phosphorus (80%), TKN (65-75%), and NH<sub>3</sub> (60-80%). However, results for NO<sub>3</sub><sup>-</sup> were varied with little removal.

Davis (2007) conducted a study of two bioretention systems with different designs (one system was a standard design while the other had an anoxic sump) to compare water quality from parking lot stormwater runoff. The contaminants analyzed were TSS, TP, NO<sub>3</sub>-N, total Cu, Pb, and Zn. Twelve events were analyzed for TSS, PO<sub>4</sub><sup>3-</sup>, and Zn, nine events were analyzed for Cu and Pb, and three events were analyzed for NO<sub>3</sub>-N. Authors found both bioretention systems showed large improvements in parking lot runoff without any significant differences in efficiency between the two. Median percent removals were 47% for TSS, 76% for PO<sub>4</sub><sup>3-</sup>, 57% for Cu, 83% for Pb, 62% for Zn, and 83% for NO<sub>3</sub>-N.

Hunt et al. (2008) conducted a study to look at the effectiveness of a bioretention cell in reducing contaminant concentrations in an urban setting. The bioretention cell treated runoff from a nearby municipal building parking lot, and had a drainage area of 0.37 ha. Composite and grab samples were collected and monitored for TKN, TN, NO<sub>2</sub>-N, NO<sub>3</sub>-N, NH<sub>3</sub>-N, BOD, fecal coliform, E. Coli, TSS, Cu, Zn, Fe, and Pb. Results showed a decrease in the concentrations of TN, TKN, NH<sub>3</sub>-N, BOD, fecal coliform, E. Coli, TSS, Zn, Cu, and Pb, however the concentration of NO<sub>2</sub>-N and NO<sub>3</sub>-N remained almost unchanged, and the concentration of Fe increased. Peak flows were also shown to be reduced by at least 96% which authors believe may support the use for bioretention systems to help protect from local flooding.

Li and Davis (2008) studied bioretention to reduce metal concentrations in infiltrating stormwater runoff. Each cell was in a parking lot and the cells dimensions were 2.9 × 5.4 × 6.3 m with a median depth of approximately 1.1 m. The cells were comprised of mulch (20 % by volume), sand (50 % by volume) and top soil (30 % by volume). To evaluate the effectiveness of the media in retaining metals, soil cores were taken from the surface to 85 – 90 cm. The metals of interest were Zn, Pb, and Cu. Results indicated that most of the metals captured, based on sequential extractions, were of anthropogenic origin. The metals were mainly in the upper layers of the media and decreased in concentration further down the media profile.

A monitored bioretention cell was studied to understand the behavior of PAHs in stormwater runoff (DiBlasi et al., 2009). This bioretention cell received stormwater runoff from a catchment that was 90% impervious surfaces (parking lots, roads, and sidewalks), trapezoidal in shape with dimensions 50.3 m × 2.4 – 4.8 m, and area of 181 m<sup>2</sup>. The incoming stormwater entered the bioretention cell through a flume and incoming water was sampled. There was an underdrain that directed the infiltrating water to a discharge manhole where it was sampled again. The results showed PAH content was reduced from 31–99% in the bioretention cell, and indicated the dominant PAHs were fluoranthene and pyrene.

This study was conducted at three different locations to look at the effect of bioretention systems on water quality (Elliott et al., 2011b). Data was collected over a period of six years, and examined the water quality of inflow, effluent, the vadose zone, and groundwater. The contaminants analyzed were Cl<sup>-</sup>, TSS, NH<sub>3</sub>, ON, NO<sub>3</sub><sup>-</sup>, TKN, NO<sub>2</sub><sup>-</sup>, TP and PO<sub>4</sub><sup>3-</sup>. The three locations were chosen to see if surrounding land uses had any effect on resulting water quality after treatment. Samples were collected from lysimeters and observation wells installed both in and outside the rain gardens. Results showed the rain gardens were effective in reducing concentrations of some nutrients like nitrogen and total phosphorus, however authors found Cl<sup>-</sup> was continually leached into the vadose zone, posing a potential problem. The authors also concluded the rain gardens could remove nutrients before recharge water reached groundwater, and the rain gardens did not have a negative impact on groundwater.

David et al. (2015) studied the efficiency of a bioretention system in a semiarid region. The bioretention system was 427 m<sup>2</sup> and consisted of four rain gardens and one bioswale with a drainage area of about 16,200 m<sup>2</sup>. The contaminants monitored were total Hg, PCBs, PAHs, dioxins, TSS, Cd, Cu, Ni, Pb, and Zn. Samples were collected from various storm events (different seasons and different intensity storms) before and after installation of the bioretention system. The results showed the system was very effective at removing metals and organic contaminants. Authors also found contaminants originating from cars and pavements had higher concentrations in the runoff and were effectively treated, whereas contaminants originating from atmospheric sources had lower levels in runoff and were reduced less after treatment.

Valtanen et al. (2017) conducted a study using large scale lysimeters to look at how biofiltration systems retain nutrients and metals, and infiltrate water in different seasons. They used eight open-air lysimeters that were either vegetated or non-vegetated. Six of the lysimeters contained three plant species tolerant to drought and excessive soil moisture conditions. Lysimeters were irrigated six times during the study using artificial stormwater designed to mimic typical rainfall and snowmelt events, and irrigation volume was calculated to represent the runoff depth of an impervious surface of 27 m<sup>2</sup>, so the surface area of a lysimeter was 3% of its catchment area. The contaminants studied were Zn, Cu, aluminum (Al), PO<sub>4</sub><sup>3-</sup>, NO<sub>3</sub><sup>-</sup>, and NaCl. These contaminants and their concentrations in the artificial stormwater were chosen based off measurements taken from the surrounding area. Inflow and outflow rate were measured using pressure sensors, and temperature was continuously monitored at different depths: surface, 35 cm below surface, and 85 cm below surface. One-liter samples were taken during irrigation events at a frequency of one sample per 30 L of drainage water that passed through the soil. Their results suggested that roots in the vegetated soils increased infiltration rates since the vegetated lysimeters had higher outflow rates than the non-vegetated ones. Both the vegetated and non-vegetated systems retained almost 100% of the PO<sub>4</sub><sup>3-</sup> regardless of the season which led authors to conclude PO<sub>4</sub><sup>3-</sup> was retained by the soil and not the vegetation. The root system also had a significant effect on nitrate retention which suggested that inactive and active vegetation increase the bioretention of NO<sub>3</sub><sup>-</sup>. Cu and Zn were well retained by both vegetated and non-vegetated systems which suggested temperature and vegetation do not impact the retention of these metals. Al behaved differently than the other two metals in summer and spring because the amounts of Al in the outflow were much larger than Al levels in the inflow, and salting increased the leaching of Al.

The reviewed literature for bioretention systems found that most contaminants were retained in the media of the bioretention system. Nonetheless, there were exceptions that could pose a risk to groundwater quality over time (Davis, 2007; Hunt et al., 2008; Elliott et al., 2011b; Valtanen et al., 2017). In these exceptions, there was little attenuation of NO<sub>3</sub><sup>-</sup>, Cl<sup>-</sup> and was found to be unretained; the salt inputs would also increase the leaching of Al in some cases.

### **7.2.6 Swale Systems**

Bäckström (2003) reviewed previous studies of grassed swales and their ability to retain TSS, particles, and heavy metals after rainfall and snowmelt events. Results showed a removal of 79 – 98% of TSS during simulated events, and swales with thin vegetation had the least amount of TSS removal. There was no significant removal of TSS when the influent had concentrations of TSS below 40 mg/L. Swales were more efficient in trapping larger particles than smaller particles. The author concluded that grassed swales should be used as primary treatment devices even with their inability to produce consistent removal rates. Bäckström (2003) also states that contaminants are not permanently bound to the grassed swale and during runoff events with low contaminant concentrations, the grassed swales release contaminants.

A study investigating swale-trench system designed for the capture of  $\text{PO}_4^{3-}$ , Cd, Cr, Cu, Pb, and Zn contaminated runoff from roadways was conducted by Ingvertsen et al. (2012). Seven sites were investigated, but details of any of the swale-trenches were not provided. In general, this study showed  $\text{PO}_4^{3-}$  soil concentrations were the highest in the 0 – 5 cm depth, decreased in concentration in the 5 – 15 cm depth, and were lowest in the 15 – 25 cm depth. For most sites, there were no systematic trends in concentration in any of the depths looked at for Cd, Cr, Cu, and Pb. Zinc, however, showed a trend of decreasing concentration with increasing depth.

The water quality purification potential of non-vegetated bioswales using simulated stormwater runoff was investigated by Li et al. (2016). A group of 10 bioswales were studied. Five of the bioswales were permeable, and five had anti-seepage properties. The authors focused on the effects of various media on water purification, time intervals, effects of contaminant concentration on purification, and the effects of the ditch widths on purification. Results showed different inflow concentrations led to poor removal rates for nitrogen and phosphorus while other bioswales with a constant inflow concentration had removal rates of 35 – 50% for nitrogen and 95% for phosphorus. Long running time intervals showed decreases in removal rates for both COD and Zn. The authors concluded that adsorption ability for the bioswale with respect to COD and Zn could not be restored in a short period. Larger widths were associated with higher removal rates for COD and  $\text{NH}_3\text{-N}$  (up to 90%).

The swale systems reviewed indicated that removal of contaminants was similar to what was reported for other GI systems. The one exception was the work of Bäckström (2003) who suggested that grassed swales may release contaminants into the surrounding media. However, this research did not indicate if the contaminants released would travel through the vadose zone and reach groundwater.

### **7.2.7 Aquifer Storage and Recovery Systems**

Aquifer storage and recovery (ASR) stores water in the aquifer by injecting the water into the aquifer and then using the water when it is needed. Surface water is injected from various sources: from treated potable water, treated stormwater, reclaimed water, or river water. The water recovered can be used for municipal, industrial or agricultural uses, but there is concern that mixing of this water into the existing aquifer water can change the water quality for future use.

A study of an aquifer storage and recovery (ASR) project in South Australia was conducted by Herczeg et al. (2004). Stormwater runoff was collected and held in three holding lakes used for aesthetics and as a first order cleanup step. Water from the lakes was injected into a confined aquifer composed of Tertiary limestone and was 50 to 70 m thick. The aquifer had solution cavities and highly variable porosity because the cementation was heterogenous in nature. Samples were taken from the injection well and three monitoring wells down gradient at 25, 65, and 325 m. The researchers found injection of high organic matter and oxygenated stormwater in the suboxic limestone aquifer caused several geochemical reactions to occur: carbonate mineral dissolution, sulfide oxidation, and aerobic  $\text{CO}_2$  production. They found that once dissolved oxygen was consumed, further degradation of the organic matter occurred via anaerobic oxidation using  $\text{Fe}^{3+}$  and sulfide reduction. They also found the initial injection of the stormwater into the aquifer would change the groundwater chemistry by mixing the two different water types and the biogeochemical processes. This caused the release of Ca, Mg,  $\text{HCO}_3^-$ , and  $\text{SO}_4^{2-}$ . They also found there were no new geochemical changes when water was withdrawn from the aquifer.

An ASR system in Florida was studied by Jones and Pichler (2007). The storage zone of the aquifer was the Suwannee Limestone. The water enters in the eastern part of the aquifer and eventually discharges

into the Gulf of Mexico. The results showed that except for Eh, DO, Fe, and As most of the parameters did not vary more than 7%. This showed conditions in the storage zone did not cause large seasonal variations in the water chemistry. The researchers hypothesized mechanisms by which As became more mobile in this aquifer: oxidation of pyrite, release from hydrous ferric oxides, and microbial activity. Pyrite in the aquifer near the injection zone could become more oxidized from the injection water. This caused the dissolution of pyrite and the release of As associated with the pyrite into solution. The hydrous ferric oxides could also be oxidized and cause the ferric oxides to dissolve which would release any As sorbed by the hydrous ferric oxides and/to be released into solution. The researcher indicated the role of microbes would not be great in the immediate area at the injection site. The water down gradient and the presence of nutrients could cause the microbes to oxidize the pyrite and release the As into the groundwater.

Another ASR study was conducted by Page et al. (2010). This study investigated urban stormwater from a mixed residential and industrial catchment that was passed through two settling ponds and a constructed wetland prior to injection into the aquifer. The aquifer was a confined sandy limestone that was 60 m thick and was 160 to 220 m below ground surface. There were six wells that consisted of two recovery wells and four injection wells. The wells were screened over 17 m at depths ranging from 165 – 182 m. This research indicated there were no pathogens in the groundwater prior to injection, there were pathogens in the wetland treated stormwater prior to injection, and there were low levels of pathogens in the recovered water. The researchers also found lower Fe concentrations in the wetland treated water than in the recovered water. The ambient As concentrations in the aquifer and aquifer materials suggested the aquifer could be a source of As in the recovered water. However, the authors point out that although the iron oxides in the aquifer sequester the As, changes in groundwater chemistry could alter this equilibrium and in some cases release As into the groundwater. The authors monitored over 300 organic chemicals and most were not detected. Simazine (a broadleaf herbicide) was the most frequently detected organic compound in the wetland but not in the recovered water.

Vanderzalm et al. (2010) further studied the ASR system that was described in Page et al. (2010). The results of the study indicated the principal changes to water quality in the recovered water was the dissolution of calcite. This caused increases in the concentrations of  $\text{HCO}_3^-$ , Ca, Mg, and Na. The excess concentrations of Ca and  $\text{HCO}_3^-$  can be explained by the dissolution of calcite, but Ca cannot balance the increase in concentration of  $\text{HCO}_3^-$ . Therefore, authors suggested that cation exchange was possibly occurring as well to balance excess  $\text{HCO}_3^-$ . The aquifer was providing a passive treatment for nutrients and DOC during the storage phase. The injection of oxygenated water induces the oxidation of DOC and pyrite in the aquifer. An important aspect of this study was that ion exchange and DOC sorption will reduce the sorptive capacity of the aquifer over time.

Another ASR site was studied to understand the As mobilization in the storage aquifer (Vanderzalm et al., 2011). The aquifer storage zone was the lower portion of a carbonate aquifer approximately 60 m in thickness. The storage zone was roughly 100 – 160 m below ground surface. The results of this study indicated the injection of reclaimed water into this aquifer mobilized As. The mobilized concentrations of As did not decline during the study period. The researchers stated the initial injection of oxygenated water into the aquifer caused reduced minerals in the aquifer to dissolve and release As in the zone near the injection. Further away from the injection site As was released because of desorption of As from the Fe oxyhydroxides in the aquifer. During the storage phase of the ASR cycle microbial action caused by the injected DOC resulted in an increase of As because of reductive dissolution of Fe oxyhydroxides and the loss of sorption sites. Microbial action also played a role in the conversion of  $\text{As}^{5+}$  to the more mobile  $\text{As}^{3+}$  species.

A groundwater replenishment system was studied by Fakhreddine et al. (2015). This system was designed for potable water reuse and in 2015 had a production capacity of 70 million gallons per day and it was anticipated that the production capacity will be expanded to 100 million gallons per day. In this system, secondary treated waste water was collected and further treated using microfiltration, reverse osmosis, ultraviolet light, and a hydrogen peroxide addition to produce water for recharge into the aquifer. Prior to infiltration, the ambient groundwater and recharge water did not have detectable As. The As was found to be associated with the sediments at low concentrations under the basin. The higher clay content layers had the highest As concentrations and redox sensitive elements Fe, sulfur (S), and Mn. The researchers argued that As release was a function of ionic composition and not surface changes caused by low ionic strength. They stated that when the concentrations of Ca and Mg decreased, the release of As into the surrounding water was comparable with NaCl inputs. Therefore, the lack of divalent cations in the low ionic strength infiltration water caused the desorption of As by stripping Ca and Mg from the surface of the clays. The release of As associated with ASR could be mitigated by treating the infiltration water with lime and dolomitic lime prior to infiltration.

Aquifer and storage systems are different than other GI infrastructure systems in that water is directly injected into the aquifer. Because the stormwater is directly injected into the aquifer, changes in groundwater quality occur. Changes in water quality can be as simple as mixing two waters from different sources, which can perturb equilibrium conditions and change the groundwater chemistry. In other cases, unintended consequences can happen, such as the mobilization of As.

### **7.2.8 Dry Wells/Diffusion Wells**

Emmanuel et al. (2009) studied the effects of hospital wastewater effluent discharge into a karstic formation where the groundwater was used for human consumption. The contaminants monitored were COD, chloroform, dichloromethane, dibromochloromethane, dichlorobromomethane, bromoform, Cr, Ni, Pb, and fecal coliforms. The site was at an emergency hospital that used septic tanks to collect effluent which was then treated and discharged into a diffusion well located in a saturated and non-saturated area. Samples were collected from one septic tank, the discharge line of the drinking water supply well, and the effluent of the septic tank. Results showed concentrations for most metals as below threshold values, however concentrations for Cr, Ni, and Pb were high and posed a potential for major health risks. COD concentrations were also high. These authors concluded that hospital wastewater effluent was creating a health risk for consumers who drink the groundwater.

Edwards et al. (2016) evaluated the potential for stormwater to contaminate groundwater using dry wells. They concluded that although dry wells are an effective means to infiltrate stormwater and recharge aquifers, the success of the use of dry wells depends on the local subsurface conditions, the maintenance of the dry well, and the quantity and quality of the influent water. They found in a few of the studies reviewed that the infiltrated stormwater was the possible source of groundwater contamination. They also reported groundwater monitoring showed in many cases there was a decrease in the concentration of contaminants between the land surface and groundwater. This suggested the contaminants were attenuated in the vadose zone. The authors of the review did note that this apparent reduction in groundwater concentration could be due to the study not persisting long enough for the true effects of the contamination to be observed.

The use of dry wells or diffusion wells can in some cases reduce contamination of groundwater from the infiltrating stormwater. However, dry wells or diffusion wells can also be the source of groundwater contamination. If this type of GI is used to infiltrate wastewater there will likely be groundwater quality changes and possible contamination.

### 7.2.9 Rain Gardens/Vegetative Strips

The effects of rain gardens on the water quality was investigated by Tornes (2005). In this study, five rain gardens were selected representing a range of land uses and hydrologic conditions. Land use ranged from impervious surfaces to grassy areas. This study found that some of the effects of rain gardens were enhanced infiltration and the reduction of dissolved ions.

A study, authored by Deletic and Fletcher (2006), focused on the performance of grass filters and grass strips. The sites chosen were a grass filter strip and a grass swale. For the grass filter strip, the focus was on the filtering efficiency with respect to differing particle sizes; in the grass swale, the focus was on removal efficiency for TN and TP. Total suspended solids were recorded at both locations, and both had uneven contaminant inflow. Results showed that grass strips and grass swales were effective in removing sediment from stormwater runoff. Authors did note that TSS removal is mainly a physical process that relies on flow rate, grass density, and particle size and density. Results also suggested sediment deposition may cause soil clogging leading to less infiltration and more over flow. TN and TP removal were also present in the grass swale. With respect to TP, high surface area to volume ratio allowed rapid soil sorption. For TN, rapid removal was also seen, which suggested potential chemical or biochemical processes were responsible for removing nitrogen.

A study was conducted examining the efficiency of a vegetative filter strip in conjunction with a subsurface drainage system installed 1.2 m below the soil surface (Bhattarai et al., 2009). Concentrations for  $\text{NO}_3\text{-N}$ ,  $\text{PO}_4^{3-}$ , and TP were measured from surface water samples entering and leaving the filter strip, and from subsurface outflow. Soil samples were also taken and analyzed for concentrations of plant-available  $\text{PO}_4^{3-}$  and  $\text{NO}_3\text{-N}$ . The vegetative filter strip was built in 1994, reconstructed in 2001, and can treat runoff from a cattle feed lot. Results showed decreases in  $\text{PO}_4^{3-}$ ,  $\text{NO}_3\text{-N}$ , and TP, with as much as 75% for  $\text{PO}_4^{3-}$  and 70% for TP for surface flow. Subsurface outflow results also showed a decrease in  $\text{PO}_4^{3-}$  and TP, but an increase in  $\text{NO}_3\text{-N}$  concentrations. These results led the authors to conclude that while a vegetative filter strip can be effective at removing nutrients from surface flow and reducing runoff, the subsurface drain may increase  $\text{NO}_3\text{-N}$  transport and therefore be detrimental to the environment.

Duchemin and Hogue (2009) conducted a study to evaluate three types of filter strip systems and their effect on water quality within the first year of their implementation. The first system was a vegetative buffer strip comprised of grass, the second strip was comprised of a combination of grass and poplar trees (two years old), and the third was a control strip with no vegetation. The water quality parameters monitored were TSS, P, N, and E. coli. The site was an area that covered 2,328  $\text{m}^2$ , and was divided into four sections each containing three plots (one for each type of strip system). Runoff and drainage water volume were measured for each plot during rainfall events. The results showed that while the vegetated strips were effective in reducing contaminant loads in stormwater runoff (reduced water volumes by 15%, TSS by 85%, TP by 75%, dissolved P by 30%,  $\text{NH}_3$  by 50%,  $\text{NO}_3^-$  by 60%, and E. coli by 25%), there was not a significant difference in performance between the two vegetated strips.

Yang et al. (2010) conducted a study of 42 monophasic and biphasic rain gardens using simulated runoff to compare hydraulic performance and removal efficiencies of dissolved nutrients and atrazine. The design for the biphasic system was two PVC columns with saturated water conditions in the first column, and unsaturated conditions in the second column. The monophasic design used the same setup as the biphasic design except both columns were unsaturated and constructed with a traditional underdrainage approach. Experiments consisted of five simulated runoff events every five days at a constant temperature of 20°C. Temperature, pH,  $\text{NO}_3^-$ ,  $\text{PO}_4^{3-}$ , and glucose (C source) were measured

from the effluent of each column. Their results showed higher removal of  $\text{NO}_3^-$  from the biphasic system (60% removal for low concentration runoff, and 34% removal for high concentration). Most  $\text{NO}_3^-$  removal was observed in column one with some additional removal in column two. Overall removal efficiency of  $\text{NO}_3^-$  in both systems decreased with the high concentration contaminant runoff. The authors concluded that with the high  $\text{NO}_3^-$  loadings, there was not enough available C substrates for complete denitrification. At low concentration loadings, both systems had 100% retention of atrazine and  $\text{PO}_4^{3-}$ . With the higher concentration loadings, both systems were effective at removing atrazine (84 – 89%) and  $\text{PO}_4^{3-}$  (89 – 94%). Higher removal of the  $\text{PO}_4^{3-}$  was observed in the biphasic system which suggested increased saturated retention time may increase  $\text{PO}_4^{3-}$  removal. Results also indicated the soil's capacity to sorb or degrade atrazine became less effective over time. The authors concluded that C substrate increased overall removal efficiency, and the biphasic system was better at reducing peak flow, runoff volume, and contaminant loads.

LeFevre et al. (2012) looked at biodegradation of total petroleum hydrocarbons (TPH) in 58 rain gardens representing different ages, catchment land uses, and four upland (do not receive runoff from impervious surfaces) sites. Specifically, the authors wanted to look at the potential for biodegradation of petroleum hydrocarbons and the factors involved with petroleum hydrocarbon biodegradation, given that rain gardens frequently experience variable contaminant loading and different soil moisture conditions. The authors analyzed concentrations of TPH, 16S rRNA genes for bacteria, and two functional genes that encode for enzymes used in petroleum hydrocarbon degradation. 75 soil samples were collected from just below the ground surface. Results showed the rain gardens all had similar levels of TPH concentrations, and 39% of samples from the rain gardens had concentrations below detection limits. All the upland site samples had concentrations below detection limit. The authors also noted the functional genes and 16S rRNA genes were higher in rain gardens that had deeply rooted plants versus those that had shallow rooting depth (grass) or zero vegetation. They also found the soils were all able to mineralize naphthalene. Based off these results, the authors suggested that concentrations of petroleum hydrocarbons were decreasing instead of accumulating, and that rain gardens may be more efficient than retention ponds for treating petroleum hydrocarbons.

Komlos and Traver (2012) conducted a nine-year study to look at the ability of a biofiltration rain garden to trap  $\text{PO}_4^{3-}$  from stormwater runoff. The site was in a parking lot that had a drainage area of 0.52 ha with about 35% being impervious surface. Data were collected from rainfall events. Results showed the median concentration for  $\text{PO}_4^{3-}$  decreased from 0.21 – 0.25 mg/L to 0.03 mg/L, and there was no decrease in efficiency of removal. Authors examined how much of the  $\text{PO}_4^{3-}$  sorbed onto the soil to determine the potential longevity of the rain garden. They found the first 10 cm were saturated with  $\text{PO}_4^{3-}$ , but calculations based off their data and assumptions indicated that saturation at greater depths would not occur for another 20+ yrs. They concluded that infrequent maintenance would be required for long term operation of rain gardens for removal of  $\text{PO}_4^{3-}$ .

Yang et al. (2013) conducted a two-year study to look at hydraulic performance and removal efficiencies of biphasic rain gardens. They used natural and simulated runoff events, and monitored the influent and effluent. Three agricultural events had high concentrations of  $\text{NO}_3^-$ ,  $\text{PO}_4^{3-}$ , and atrazine, and five urban events had spiked concentrations of  $\text{NO}_3^-$ ,  $\text{PO}_4^{3-}$ , glyphosate, dicamba, and 2,4-D. The biphasic rain gardens were effective in removing  $\text{NO}_3^-$  (91%),  $\text{PO}_4^{3-}$  (99%), atrazine (90%), dicamba (92%), glyphosate (99%), and 2,4-D (90%) with high levels of contaminant loading in simulated events. The authors also found increased retention time of runoff and water saturated conditions significantly affected contaminant removal.

Most of the contaminants were found sequestered in the upper portions of the soil profile for rain gardens and swale systems. It should be noted that none of the studies looked at compounds of interest in the deeper layers of the vadose zone or groundwater. In one case, it was suggested that there was a potential for  $\text{NO}_3^-$  to be transported into deeper layers of the vadose zone (Bhattarai et al., 2009).

### **7.2.10 Riparian Zones**

Urban riparian zones have the potential to function as green infrastructure. They have been studied extensively in the past 20 years, but few of these studies are relevant to an understanding of how riparian zones and riparian buffers affect groundwater quality while providing GI functions. Although the studies that address hydrology and water quality in urban riparian systems have some relevance to effects of riparian zone GI (RZGI) on groundwater, there are no specific studies where RZGI has been applied as a technique for stormwater infiltration. Thus, the understanding of RZGI on stormwater and groundwater will draw on relevant studies of riparian zones in urban settings.

Groffman et al. (2003) in a seminal study on urban riparian ecology suggest that riparian “hydrologic drought”, caused by lowered water tables, is a general effect of urbanization. These urban riparian zones should provide an ideal place to infiltrate stormwater because of this hydrologic drought. As a conceptual model, there are at least three potential ways to achieve this infiltration. One is to direct stormwater across riparian zones to promote direct infiltration. The second is to allow overbank flows into floodplains where flow retention and infiltration can occur. The third is to enhance bank infiltration and hyporheic extraction (Nutzmann et al., 2011). As Correll (2005) proposes, smaller streams can be more actively managed for chemical and sediment retention, while for larger streams it is more a matter of floodplain protection. For urban stormwater, more active management of the floodplain is probably required, especially because current practices typically shunt stormwater to larger streams where possible.

Restoration of stream health should be the goal of stormwater controls driven by the Clean Water Act. A flow regime similar to pre-development flows is necessary to restore stream health. (Poff et al., 1997). Biotic integrity can be maintained in streams with intact riparian zones or significant amounts of groundwater flow (or both), despite a high degree of urban land use (greater than 30%) (Miltner et al., 2004). Source control management techniques can affect baseflow and localized groundwater, but there is poor understanding of how source control stormwater management can make riparian buffer functions possible and allow further contributions to stream health. Studies from agricultural watersheds demonstrate that healthy riparian buffers need watershed management to function (Lowrance et al., 1985). Restoring baseflow is key to restoring riparian buffers and having functioning riparian buffers in alluvial systems. Although, simply infiltrating water in the upper watershed will not assure restoration of baseflow, and water harvesting systems that retain water in the watershed may not help to restore baseflow (Walsh et al., 2012, Hamel et al., 2013). Without a good understanding of local hydrogeology and the interactions with enhanced infiltration systems, the effects on streamflow are highly speculative unless enhanced recharge takes place in nearby stream areas.

Riparian zones in urban settings are often affected by lowered localized water tables. These lowered water tables are generally due to less water infiltrating locally, primarily due to impervious and less pervious surfaces. This is the same condition that leads to lower baseflow in small headwater urbanized streams. Groffman et al. (2003) speculated on the effects of “urban hydrologic drought” on urban riparian systems. When riparian buffers are installed as GI, they may be affected by this urban hydrologic drought which can lead to 1) minimal effects on water quality, 2) low survival for riparian vegetation, and 3) invasive upland species, including exotics, in the riparian zone.

Riparian buffer restoration may have little effect on instream flows unless connections are reestablished between filtered or spread flows and the riparian system (Walsh et al., 2005). Restoration of riparian buffers in urban landscapes will either require active management to control terrestrial invasives, or restoration of alluvial aquifers, high water tables, and wet soils to encourage the survival of native species and the exclusion of exotics that can thrive in less wet environments (Auble et al., 1997).

Increasing infiltration through restoration or preservation of riparian buffers along smaller streams such as ephemeral waterways may have the ability to enhance local water resources (Gallo et al., 2013). Urban landscapes deliver more water to ephemeral channels due to decreased infiltration in the watershed. This provides an opportunity for focused infiltration within those ephemeral channels.

The interaction of nitrogen removal through denitrification and the restoration of infiltration to soil and groundwater in urban riparian zones has been better studied than other potential stormwater contaminants. Because biological denitrification occurs in organic rich soil, subsoil, or aquifer material which becomes saturated, nitrate removal in riparian buffers is dependent on the re-wetting of the soil and aquifer through enhanced infiltration. Groffman and Crawford (2003) reported studies on restoring denitrification potential in urban riparian zones and found that denitrification was enhanced in areas of high organic matter and higher soil moisture. They propose routing high  $\text{NO}_3^-$  runoff through these areas to enhance the denitrification capacity of urban riparian buffers. Palta et al. (2016) point out that in urban riparian areas, high denitrification may be found in areas of groundwater seeps or standing water. This supports the general idea that re-wetting urban riparian zones will be necessary to restore denitrification.

Unlike riparian buffers on smaller streams, larger rivers that were often the focus for the development of urban areas require different management approaches. For larger rivers, studies have shown that increasing infiltration of flood waters on floodplains for detention and infiltration is likely to have the largest effect on groundwater, rather than connecting the riparian buffer to direct stormwater runoff (Jacobson et al. 2015; McMillan and Noe, 2017). McMillan and Noe (2017) note that low floodplains (riparian zones) frequently inundated by upstream sources will provide greater water quality improvement than more extensive but higher floodplains. This is partly due to infiltration of the ponded water in the floodplain and recharge of alluvial groundwater. This is further justification for expanding stream corridors wherever possible to “incorporate the protection of natural systems” as green infrastructure (Collins et al., 2010b). A less manipulative approach to stream restoration with feasible steps that expand the stream corridor and limited channel manipulation (Palmer et al., 2014) will create opportunities for infiltration into alluvial groundwater.

Bank infiltration is a form of artificial groundwater recharge that has been applied in urban environments, and is a longstanding practice for pre-treatment of river water that is used for drinking water (Nutzmann et al., 2011; Schmidt, et al. 2003). The practice depends on the infiltration of surface water into streambanks and riverbeds driven by groundwater pumping from alluvial aquifers. By serving as “natural filtration,” the practice may have beneficial effects on surface water if the water is discharged back to surface sources. The induced recharge can also be used as either drinking water supply or to re-water floodplains.

### **7.2.11 Results from Reviews**

A review of stormwater infiltration practices on groundwater contamination was conducted by Pitt et al. (1999). This review found the most concerning contaminants in groundwater through stormwater infiltration were  $\text{Cl}^-$ , pesticides, PAHs, pathogens, and heavy metals. They found it was rare to have

groundwater contamination, except for Cl<sup>-</sup>, in residential areas because infiltration occurred through surface soils which retains most contaminants, and that contamination of the groundwater was more likely in commercial and industrial areas where subsurface infiltration was used.

In a review article by Kabir et al. (2014), the authors reviewed the literature concerning ion and metal pollution in urban GI systems. They concluded the current understanding of GI systems in relation to metals and nutrients rely on empirical principles. They state the need to do more comprehensive studies that use pathways, thermodynamics, kinetics, and microbial dynamics along with understanding the role of plants in the system. They also found there are high pollution loads and high ecological risk associated with the use of GI in urban areas. The authors also stated the effects of GI on the watershed scale require further research.

# 8.0 Conclusions/ Future Research Needs

## 8.1 Conclusions

Stormwater reclamation for eventual reuse is triggering a paradigm shift from stormwater seen as a contaminant and a flood risk to a resource that can solve these risks. GI design strategy retains storage, infiltrates runoff, and contributes to the renewed groundwater recharge to more closely resemble the hydrology before urban development. The disturbance of the natural hydrologic cycle due to urbanization is closely connected to deteriorating urban water quality. This creates an increased risk to groundwater quality because of new pathways for contaminant introduction into groundwater, chemicals associated from anthropogenic activities, and wastewater exposure. This literature review determined what research that has been done on GI practices with respect to groundwater quality and the risks and impacts to the subsurface environment. The issues addressed include: 1) contaminant risks that need further research, 2) new infrastructure that has not been researched in depth, and 3) determination of local considerations when planning for green infrastructure.

Any pollutant found in stormwater could be a potential groundwater contaminant when used with GI infiltration technology. GI can return the urban hydrology to a more natural hydrologic cycle through retention and infiltration methods. Surface and subsurface infiltration can influence the impact the infiltrating stormwater has on the groundwater chemistry. Retention techniques can influence the water table depth through mounding, which have been seen in restoration projects, bioretention cells, and regenerative stormwater conveyance systems. Concern with GI for stormwater infiltration include fluctuations in groundwater levels, limitations with large precipitation events, clogging, and soil limitations. The infiltration is dependent on the clogging rate of the infrastructure.

Depending on the water's chemical, biological, and physical conditions, there is the risk of potential contaminants leaching from native soils and geology. When it comes to managing water resources, the tendency for contaminants to move between the ground and surface water needs to be considered. Urbanization can introduce contaminants that are otherwise not an issue in natural stormwater hydrology. Groundwater can be contaminated by many constituents: nutrients, metals, dissolved minerals, pesticides, other organics, and pathogens; the sources of which include residues from automobiles, lawn treatments though fertilizers and pesticides, sewer overflows, and road deicing salts. Due to risks affecting groundwater quality, it is suggested that infiltrating GI not be implemented in areas with potentially high contaminant loading, i.e. recycling centers, gas stations, and brownfields. When infiltrating devices are installed and used for urban runoff, there are concerns as to how the soils interact with the stormwater runoff pollution while infiltrating into the subsurface, thus providing possible risks of groundwater quality impairment from areas with potentially high contaminant concentrations.

The chemical interactions between surface water and groundwater are controlled by the type of geologic materials present and the amount of time the water is in contact with these materials. The various chemical reactions that affect the biological and geochemical characteristics of the basin are acid-base reactions, precipitation and dissolution of minerals, sorption, ion exchange, oxidation-reduction reactions, biodegradation, and dissolution and exsolution of gases. It is concluded that when implementing green stormwater infrastructure for infiltration, the properties of the unsaturated and saturated zones interacting with the infiltrating water need to be considered. These considerations encompass the understanding of the native soil texture, structure, and organic matter content of the

unsaturated zone, as well as considering the porosity and permeability of the saturated zone and the flow of the groundwater. Kinetics and mixing relationships also require examination. Colloidal transport also needs to be considered as a mechanism that can transport contaminants through the soil, by either being a contaminant itself or having a contaminant sorb to a benign colloid. Colloids can be restricted by capture, sorption and static interaction. As discussed previously, colloid-facilitated transport could be an important mechanism for the movement of contaminants into groundwater (de Jonge et al., 2004).

The potential and actual impacts to groundwater quality as the results of GI practices were reviewed. The results presented were mixed; in some cases, there were impacts or potential impacts, and in other cases there were no impacts found. Many of the studies' results were problematic for several reasons. In most cases, the results—reflecting only what occurred in the vadose zone or the infrastructure—were extrapolated to predict what may occur to the groundwater. This extrapolation ignores other processes that could facilitate the transport of contaminants to the groundwater, such as preferential flow. Since there was no attempt made to measure concentrations of contaminants in aquifers or deeper in the vadose zone, there is no definitive evidence of changes in groundwater quality.

In studies that did include groundwater monitoring, it is unknown in some cases if the sampling strategy would detect changes in groundwater quality. Information on groundwater flow direction was not included, therefore the relationship of monitoring points to the potential transport of contaminants could not be ascertained. Another potential problem was that the studies did not account for lag between the time of water infiltration and the time it takes to transport the infiltrated water to the aquifer. In most studies, that sampling occurred at or very close to the precipitation event. Because lag time was not considered, transient changes to groundwater quality were not accounted for, even in systems that were monitored for decades.

The only system that consistently showed impacts to groundwater quality was ASR. The ASR impacts fell into one of two categories: unintended consequences, or the mixing of two waters with different composition and characteristics.

Simulation models can be an affordable way for predicting quality and quantity changes, as well as a decision-making tool for implementing green infrastructure. While there are many models in use for surface water and groundwater transport, there are few that integrate green infrastructure, and those that have do not address groundwater contaminant transport. Green infrastructure models have been implemented in various formats, but none specifically addressed groundwater contamination from this infrastructure. Problems associated with implementing models for assessing green infrastructure technologies and influence on groundwater include the amount of data available for calibration and validating these models, indicating a need for more field research to obtain this data.

Microbiological organisms such as bacteria, viruses, and parasites can be a contamination risk depending on the unsaturated and saturated zone conditions, incubation time, and native microbial population behavior. Microbial contaminants are a concern primarily if they present a public health threat from consuming contaminated groundwater, with the most common waterborne disease being acute gastrointestinal illness. While gut-associated microbial contaminants are not expected to grow and thrive within the groundwater environment, their rates of removal are affected by several, often interdependent, environmental factors. Research has shown there is a general trend of differential survival for the various contaminant organism types. Viruses tend to have the longest persistence times within any groundwater environment; enteric eukaryotes (*Cryptosporidium* spp. and *Giardia* spp.) and enteric bacteria typically have die-off rates of five to ten times, and over one hundred times larger than enteric viruses, respectively. Pathogen removal or die-off rates are typically reported based upon first

order decay models; however, field and laboratory experiments have shown that biphasic models better approximate the removal behavior of fecal eukaryotes and viruses within groundwater systems. Hence, these studies have shown that there is an initial rapid removal phase for the first few days after introduction, followed by a slower phase two to hundreds of times less than the initial phase that can lead to months or years of persistence.

In saturated zones, factors influencing pathogen survivals in groundwater are temperature, water chemistry, and biological processes. Aquifer hydrogeology can influence the mechanical filtration, adsorption, wedging, and straining processes that can remove pathogens. There is also the potential of competition for nutrients and predation by indigenous microorganisms can play a significant role in the removal of introduced enteric pathogens. In unsaturated zones, the same processes from the saturated zone can apply but the air phase within the unsaturated zones can create two new interfaces, air-water and air-sediment that do not exist in saturated conditions which can both adsorb and entrap organisms. The decreased moisture can subject microorganisms to die-off or inactivation through desiccation. The highest native microbial populations are going to be in the rooting zone of the soil profile. Below the rooting zone, microbial populations and activity decrease with depth.

Macrobiological organisms can enhance or cause complications with green infrastructure. Vegetation is often used to retain nutrients and metals, enhance ecosystem service, increase filtration, and mimic the natural hydrology. The selection of the plants is important because they need to survive potentially toxic contaminants and the perturbations of the GI systems. There are few studies on how various macroorganisms can influence the green infrastructure. Bioturbator species that live in the sediment can increase the possible risk of nutrient contamination, and burrowing activity of worms can increase the macropores in the sediment and influence the infiltration. Macrobiological organisms can enhance or cause complications for green infrastructure, but research on these effects is limited.

Urban riparian zones can function as green infrastructure, but few studies have been done on their influence on groundwater. Previous studies on riparian zone restoration show that they could be useful to restore denitrification to urban streams. By serving as “natural filtration,” the practice may have beneficial effects on surface water if the water is discharged back to surface sources. This induced recharge can also be used for either drinking water supply or to re-water floodplains. This is also a less manipulative, more feasible way to create opportunities for filtration into alluvial groundwater.

## **8.2 Future Research**

Analogous to what the Pitt et al. (1999) and the recent Kabir et al. (2014) reviews concluded, we concur that more research is required to understand the potential groundwater quality impacts that can result from the implementation of GI. Apart from conservative chemical species such as chloride, a more complete understanding of what conditions are likely to cause groundwater quality impairment is necessary to mitigate or prevent these potential impacts. This review also indicates there is an apparent risk to the vadose zone “quality.” Stormwater infiltration is causing the soil and vadose zone sediments to degrade, and the potential future impacts and risks to groundwater quality because of this are unknown—making long-term GI studies crucial.

Since land use and environmental conditions are likely to change, future groundwater risks are possible at many current GI sites if the infrastructure is not properly maintained. Further research is needed to determine the best monitoring methods for groundwater at these sites throughout their lifetime. Changing conditions will likely change the chemical and physical properties which can alter the retention properties in the soil/vadose zone. These potential land use changes and maintenance problems

need to be addressed in future research. Another issue encountered is that, once the GI system is no longer functional or is “decommissioned,” what practices should then be implemented to mitigate the potential environmental issues created by trapping the contaminants in the vadose zone. This emphasizes the need for long term monitoring methods that addresses placement of sampling points and timing of sampling to determine the long-term impacts to the subsurface. Currently GI performance standards are not included into the National Pollutant Discharge Elimination System (NPDES) permits, including impacts on groundwater. Including this into the NPDES system may be beneficial to protection groundwater quality.

Additional research is needed to understand the impacts and benefits that various macrobiological organisms have on GI, and how these affect the hydrology, fate, and transport of contaminants in GI systems. Vegetation is the most common addition to GI, but there is an inadequate understanding as to how this vegetation influences groundwater quality over time. Addressing whether preferential flow increases over time or if nutrient and metal concentrations change over time is a necessity. Previous studies on riparian buffer zones have shown various benefits to restoring these in non-GI situations, but further studies are needed to determine the benefits and potential issues with implementing them as part of urban GI.

Simulation modeling of GI systems needs to be addressed to help users understand the potential groundwater impacts. Further research of simulation models is needed to address the location and spacing of GI stormwater practices to determine if there are diminishing returns on the quantity of stormwater controls. Simulation models are necessary to determine how large GI projects can be designed to effectively reduce runoff and have the least environmental impact (Brown et al., 2012; Eckart et al., 2017). Research on the use of models to demonstrate how GI performs under different temporal scales, spatial scales, and climatic conditions is needed since there is a lack of data on the performance of these technologies. Simulation research and improvements in modeling techniques are also needed so that they can assist in understanding the role of GI in restoring the water balance, reducing contaminants over the long term, evaluating various GI performance, as well as acting as decision support tools (Dietz, 2007; Ahiablame et al., 2012; Fletcher et al., 2013; Eckart et al., 2017).

Overall, there are several research areas necessary for a better understanding of the risks of a GI infiltration technology that have been proposed as the result of this effort. There needs to be more investigations looking at the GI interactions on a longer temporal scale and wider spatial range. When implementing GI, the local geology, climate, hydrology, biology, geochemistry, type of infrastructure, and contaminant loads need to be carefully considered to reduce the risk to groundwater quality.

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# 10.0 Quality Assurance/ Quality Control

This research was conducted using an approved Quality Assurance Project Plan (QAPP), G-GWERD-0030990 v0, March 28, 2017. Unless otherwise indicated below, the data used to generate this report followed the QAPP.

- Peer reviewed data values from Toxnet or EPI Suite were preferentially used, if peer reviewed data was not available, then (est) was indicated after the data value in Table A4 (Appendix 1) to signify that the data quality was not known.

# Appendix 1

**Table A1. Possible stormwater organic contaminants. Criteria for the classification of mobility, solubility, and volatility can be found in Table A2.**

Compound	CAS #	Mobility <sup>A</sup>	Solubility <sup>B</sup>	Volatility <sup>C</sup>	Drinking Water Standard <sup>D</sup>	Superfund Regional Screening Levels for Groundwater Protection <sup>E</sup>	NY State Groundwater Effluent Limits <sup>F</sup>	Reference
					µg/L	µg/kg	µg/L	
1,1,1,2-tetrachloroethane	630-20-6	MM	S	V	-----	0.22	-----	29
1,1,1-trichloroethane	71-55-6	MM to M	S	V	MCL= 200	280	-----	29
1,1,2,2-tetrachloroethane	79-34-5	MM to M	S	MV	-----	0.03	-----	29
1,1,2-trichloroethane	79-00-5	MM to M	S	MV	MCL= 500	0.013	1	29
1,1-dichloroethane	75-34-3	M to HM	S	V	-----	0.78	-----	29
1,2,3-trichloropropane	96-18-4	M	S	MV	-----	0.00032	0.04	29
1,2,3-trimethylbenzene	526-73-8	SM to MM	SS	V	-----	1.5	-----	14
1,2,4-trichlorobenzene	120-82-1	BM to SM	SS	V	MCL= 70	1.2	-----	14
1,2,4-trimethylbenzene	95-63-6	MM	SS	V	-----	2.1	-----	14
1,2-dichlorobenzene	95-50-1	BM to MM	MS	V	MCL= 600	30	3	14
1,2-dichloroethane	107-06-2	M	S	V	MCL= 5	0.048	0.6	29
1,2-dichloropropane	78-87-5	M	S	V	MCL= 5	0.15	1	29
1,3-dichlorobenzene	541-73-1	BM to MM	MS	V	-----	-----	3	5,15
1,3-dichloropropane	142-28-9	MM	S	MV	-----	13	-----	29
1,4-dichlorobenzene	106-46-7	BM to MM	SS	V	MCL= 75	0.46	3	14,17
11-ketotestosterone	53187-98-7	MM to M	MS	NV	-----	-----	-----	31
17α-estradiol	57-91-0	BM to MM	SS	NV	-----	-----	-----	28
17α-ethynylestradiol	57-63-6	BM to MM	MS	NV	-----	-----	-----	28
17β-estradiol	50-28-2	BM to MM	SS	NV	-----	-----	-----	28

**Table A1. Possible stormwater organic contaminants. Criteria for the classification of mobility, solubility, and volatility can be found in Table A2.**

Compound	CAS #	Mobility <sup>A</sup>	Solubility <sup>B</sup>	Volatility <sup>C</sup>	Drinking Water Standard <sup>D</sup>	Superfund Regional Screening Levels for Groundwater Protection <sup>E</sup>	NY State Groundwater Effluent Limits <sup>F</sup>	Reference
					µg/L	µg/kg	µg/L	
19-norethisterone	68-22-4	MM	SS	NV	-----	-----	-----	40
1-acetyl-1-methyl-2-dimethyl-oxamoyl-2-phenylhydrazide (AMDOPH)	519-65-3	M to HM	S	NV	-----	-----	-----	1,28
1-chloro-2-methylpropane	513-36-0	MM to M	MS	MV	-----	-----	-----	1
1-chlorobutane	109-69-3	MM to M	S	V	-----	26	-----	1
1-chloropentane	543-59-9	MM	MS	V	-----	-----	-----	29
1H-benzotrazole	95-14-7	MM	VS	SV	-----	-----	-----	39
1-methylnaphthalene	90-12-0	IM to MM	SS	MV	-----	5.8	-----	25, 30
2, 4-D	94-75-9	MM to M	MS	NV	MCL= 70	-----	-----	5,8,15
2,2',3,4,4',5,5'-heptachlorobiphenyl (PCB 180)	35065-29-3	IM	NS	NV	MCL= 0.5	-----	-----	26
2,2',3,4,4',5-hexachlorobiphenyl (PCB 138)	35065-28-2	IM	NS	NV	MCL= 0.5	-----	-----	26
2,2',4,4',5,5'-hexachlorobiphenyl (PCB 153)	35065-27-1	IM	NS	NV	MCL= 0.5	-----	-----	26
2,2',5,5'-tetrachlorobiphenyl (PCB 52)	35693-99-3	BM	NS	SV	MCL= 0.5	-----	-----	26
2,3',4,4',5-pentachlorobiphenyl (PCB 118)	31508-00-6	IM	NS	SV	MCL= 0.5	-----	-----	26
2,4,4'-trichlorobiphenyl (PCB 28)	7012-37-5	BM	SS	SV	MCL= 0.5	-----	-----	26

**Table A1. Possible stormwater organic contaminants. Criteria for the classification of mobility, solubility, and volatility can be found in Table A2.**

Compound	CAS #	Mobility <sup>A</sup>	Solubility <sup>B</sup>	Volatility <sup>C</sup>	Drinking Water Standard <sup>D</sup>	Superfund Regional Screening Levels for Groundwater Protection <sup>E</sup>	NY State Groundwater Effluent Limits <sup>F</sup>	Reference
					µg/L	µg/kg	µg/L	
2,4,5,2',5'-pentachlorobiphenyl (PCB 101)	37680-73-2	IM to BM	NS	NV	MCL= 0.5	----	----	26
2-chloro-2-methylpropane	507-20-0	MM to M	S to MS	MV	----	----	----	1
2-chlorobutane	78-86-4	MM to M	S	MV	----	----	----	1
2-Chloromethylphenol	40053-98-3	MM	S	NV	----	----	----	29
2-methyl-4-chlorophenoxyacetic Acid	94-74-6	M	MS	NV	MCL= 60	0.2	0.44	39
2-methylnaphthalene	91-57-6	IM to MM	SS	MV	----	19	----	25,30
2-methylthiobenzothiazole	615-22-5	SM to MM	MS	NV	----	----	----	39
3-Chloromethylphenol	60760-06-7	MM	S	NV	----	----	----	29
3-β-coprostanol	360-68-9	IM	NS	NV to SV	----	----	----	31
4-Chloromethylphenol	35421-08-0	MM	S	NV	----	----	----	29
4'-hydroxy-diclofenac	64118-84-9	MM	SS	NV	----	----	----	33
4-methyl-1H-benzotriazole	29878-31-7	M	S	NV	----	----	----	39
4-Nitrophenol	100-02-7	MM to M	VS	NV	----	----	----	1
4-nonylphenol	104-40-5	BM	SS	MV	----	----	----	27,39,40
4-nonylphenoldiethoxylate	20427-84-9	SM	SS	NV	----	----	----	27
4-nonylphenolmonoethoxylate	104-35-8	SM	SS	NV	----	----	----	27
4-tert-Butylphenol	98-54-4	SM to MM	MS	NV	----	----	----	26
5-methyl-1H-benzotriazole	136-85-6	M	S	NV	----	550	----	39
Acebutolol	37517-30-9	M	MS	NV	----	----	----	37
Acenaphthene	83-32-9	IM to MM	SS	MV	----	550	----	26

**Table A1. Possible stormwater organic contaminants. Criteria for the classification of mobility, solubility, and volatility can be found in Table A2.**

Compound	CAS #	Mobility <sup>A</sup>	Solubility <sup>B</sup>	Volatility <sup>C</sup>	Drinking Water Standard <sup>D</sup>	Superfund Regional Screening Levels for Groundwater Protection <sup>E</sup>	NY State Groundwater Effluent Limits <sup>F</sup>	Reference
					µg/L	µg/kg	µg/L	
Acenaphthylene	208-96-8	IM to SM	SS	MV	-----	-----	-----	26
Aceslfame	33665-90-6	HM	VS	NV	-----	-----	-----	39
Acetaldehyde	75-07-0	HM	VS	MV	-----	0.38	8	38
Acetaminophen	103-90-2	M	VS	NV	-----	-----	-----	37
Acetophenone	98-86-2	MM to M	S	MV	-----	58	-----	17
Acrylamide	79-06-1	M	VS	NV	-----	0.011	-----	38
Alachlor	15972-60-8	SM to MM	MS	NV	MCL= 2	0.86	0.5	8
Aldrin	309-00-2	BM to MM	MS	MV	-----	0.75	Not Detectable	26
Alkylphenol ethoxylates	68412-54-4	HM	SS	VV	-----	-----	-----	28
Aminotriazole	61-82-5	M	VS	NV	-----	-----	-----	26
Amitriptyline	50-48-6	SM	SS	NV	-----	-----	-----	40
Amoxicillin	26787-78-0	MM to M	S	NV	-----	-----	-----	40
AMPA	74341-63-2	M to HM	VS	NV	-----	-----	-----	26
Androstenedione	63-05-8	SM	SS	NV	-----	-----	-----	31
Anthracene	120-12-7	SM	NS	V	-----	5800	-----	5
Anthraquinone	84-65-1	BM to SM	SS	NV	-----	14	-----	17
Atenolol	29122-68-7	M to HM	VS	NV	-----	-----	-----	37,40
Atrazine	1912-24-9	SM to M	SS	NV	MCL= 3	0.19	7.5	5,15
Atrazine amide-I	142179-76-8	M	S	NV	-----	-----	-----	1,28
Azithromycin	83905-01-5	SM	SS	NV	-----	-----	-----	17
Bentazone	25057-89-0	MM to M	MS	NV	-----	12	-----	1,28
Benz(a)anthracene	56-55-3	IM	NS	V	-----	12	-----	5,15

**Table A1. Possible stormwater organic contaminants. Criteria for the classification of mobility, solubility, and volatility can be found in Table A2.**

Compound	CAS #	Mobility <sup>A</sup>	Solubility <sup>B</sup>	Volatility <sup>C</sup>	Drinking Water Standard <sup>D</sup>	Superfund Regional Screening Levels for Groundwater Protection <sup>E</sup>	NY State Groundwater Effluent Limits <sup>F</sup>	Reference
					µg/L	µg/kg	µg/L	
Benzene	71-43-2	M	S	V	MCL= 5	0.23	1	29
Benzo(a)pyrene	50-32-8	IM	NS	NV	MCL= 0.2	4	Not Detectable	5
Benzo(e)pyrene	192-97-2	IM	NS	V	-----	-----	-----	5
Benzo(a)fluoranthene	203-33-8	IM	NS	SV	-----	-----	-----	5
Benzo(b)fluoranthene	205-99-2	IM	NS	NV	-----	41	-----	5
Benzo(b)naphtho(2,3-d)thiophene	243-46-9	IM	NS	NV	-----	-----	-----	5
Benzo(ghi)perylene	191-24-2	IM	NS	SV	-----	-----	-----	6
Benzo(j)fluoranthene	205-82-3	BM	NS	NV	-----	78	-----	5
Benzo(k)fluoranthene	207-08-9	IM to BM	NS	SV	-----	400	-----	5
Benzophenone	119-61-9	MM	MS	SV	-----	-----	-----	17,31
Benzothiazole	95-16-9	MM	S	SV	-----	-----	-----	39
Benzyl acetate	140-11-4	MM	S	MV	-----	-----	-----	28
Beta-sitosterol	83-46-5	IM	NS	SV	-----	-----	-----	31
Bezafibrate	41859-67-0	MM	SS	NV	-----	-----	-----	1
Bis(2-chloroethyl) ether	111-44-4	MM	VS	MV	-----	0.0036	1	5
Bis(2-chloroisopropyl) ether	39638-32-9	MM to M	SS	SV	-----	-----	-----	5
Bis(2-ethylhexyl)phthalate (DEHP)	117-81-7	IM to BM	NS	SV	MCL= 6	1300	5	1,5,15
Bis(tributyltin) oxide	56-35-9	IM to SM	SS	SV	-----	29000	-----	26
Bisphenol A	80-05-7	SM to MM	MS	NV	-----	5800	-----	31, 39,40
Bromochloroacetic acid	5589-96-8	HM	VS	NV	MCL= 60	-----	-----	34
Bromodichloroacetic acid	71133-14-7	M to HM	S	NV	MCL= 60	-----	-----	34

**Table A1. Possible stormwater organic contaminants. Criteria for the classification of mobility, solubility, and volatility can be found in Table A2.**

Compound	CAS #	Mobility <sup>A</sup>	Solubility <sup>B</sup>	Volatility <sup>C</sup>	Drinking Water Standard <sup>D</sup>	Superfund Regional Screening Levels for Groundwater Protection <sup>E</sup>	NY State Groundwater Effluent Limits <sup>F</sup>	Reference
					µg/L	µg/kg	µg/L	
Bromodichloromethane	75-27-4	MM to M	S	V	MCL= 80	0.036	-----	28
Butylbenzylphthalate (BBP)	85-68-7	SM	SS	SV	-----	230	-----	5
Butylated hydroxyanisole	25013-16-5	MM	MS	SV	-----	450	-----	17
Caffeine	58-08-2	SM to M	VS	NV	-----	-----	-----	24,28
CA-ibuprofen	-----	SM to M	S to MS	NV	-----	-----	-----	28
Carbamazepine	298-46-4	MM	SS	NV	-----	-----	-----	1
Carbamazepine-10,11-epoxid	36507-30-9	MM to M	MS	NV	-----	-----	-----	33
Carbendazim	10605-21-7	SM to MM	SS	NV	-----	-----	-----	39
Carbon tetrachloride	56-23-5	M	S to MS	V	MCL= 5	0.18	5	29
Carbozole	86-74-8	SM to MM	SS	SV	-----	-----	-----	17
Cefazolin	25953-19-9	M	MS	NV	-----	-----	-----	42
Cefriaxone	73384-59-5	MM to HM	MS	NV	-----	-----	-----	42
Ceftazidime	72558-82-8	BM to HM	MM	NV	-----	-----	-----	28
Chlordane	57-74-9	BM	NS	MV	MCL= 2	-----	0.05	1,5,15
Chlorfenvinphos	470-90-6	MM	MS	NV	-----	3.1	-----	26
Chloroacetic acid	79-11-8	M	VS	NV	MCL= 60	0.81	-----	38
Chlorodifluoromethane	75-45-6	HM	S	V	MCL= 80	4300	-----	28
Chloromethane	74-87-3	M	S	V	-----	4.9	-----	29
Chlorpyrifos	2921-88-2	BM to MM	SS	MV	-----	12	-----	12
Chlortetracycline	57-62-5	M to HM	MS	NV	-----	-----	-----	33
Cholesterol	57-88-5	BM	NS	MV	-----	-----	-----	31
Choline Chloride	67-48-1	HM	VS	NV	-----	-----	-----	38

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					µg/L	µg/kg	µg/L	
Chrysene	218-01-9	IM	NS	SV	-----	1200	-----	1
Cimetidine	51481-61-9	M	VS	NV	-----	-----	-----	17
Ciprofloxacin	85721-33-1	M to HM	VS	NV	-----	-----	-----	28
Cis-androsterone	53-41-8	SM to MM	SS	NV	-----	-----	-----	31
Citalopram	59729-33-8	BM to SM	SS	NV	-----	-----	-----	17
Clarithromycin	81103-11-9	MM	SS	NV	-----	-----	-----	37
Clofibric Acid	882-09-7	M	MS	NV	-----	-----	-----	1
Cocaine	50-36-2	MM	S	NV	-----	-----	-----	35
Codeine	76-57-3	MM	VS	NV	-----	-----	-----	17,40
Coronene	191-07-1	IM	NS	NV	-----	-----	-----	25
Cotinine	486-56-6	MM	VS	NV	-----	-----	-----	17
Cyanazine	21725-46-2	MM	MS	NV	-----	0.041	-----	1
Cyanazine amide	36576-42-8	M	S	NV	-----	-----	-----	1,28
Cyclophosphamide	50-18-0	M	VS	NV	-----	-----	-----	40
DDT	50-29-3	IM	NS	SV	-----	77	0.2	6
Debromodiphenyl ether	1163-19-5	IM	NS	NV	-----	7800	-----	29
Deethylcyanazine	36556-77-1	M to HM	S	NV	-----	-----	-----	1,28
Dehyronifedipine	67035-22-7	SM to MM	SS	NV	-----	-----	-----	17
Deisopropylatrazine	1007-28-9	M	MS	NV	-----	-----	-----	1,28
Desethylatrazine	6190-65-4	SM to M	S	NV	-----	-----	-----	1,28
Desethylsimazine	1007-28-9	M	MS	NV	-----	-----	-----	26
Diatrizate	117-96-4	M to HM	VS	NV	-----	-----	-----	39
Diazinon	333-41-5	SM to M	SS	NV	-----	6.5	0.7	5,12,15

**Table A1. Possible stormwater organic contaminants. Criteria for the classification of mobility, solubility, and volatility can be found in Table A2.**

Compound	CAS #	Mobility <sup>A</sup>	Solubility <sup>B</sup>	Volatility <sup>C</sup>	Drinking Water Standard <sup>D</sup>	Superfund Regional Screening Levels for Groundwater Protection <sup>E</sup>	NY State Groundwater Effluent Limits <sup>F</sup>	Reference
					µg/L	µg/kg	µg/L	
Dibenz([a,h]anthracene	53-70-3	IM	NS	SV	-----	13	-----	14,20,25,30
Dibenzofuran	132-64-9	SM	SS	MV	-----	15	-----	25
Dibenzothiophene	132-65-0	BM to SM	SS	MV	-----	12	-----	29
Dibromoacetic acid	631-64-1	HM	VS	NV	MCL= 60	-----	-----	34
Dibromochloroacetic acid	5278-95-5	M to HM	S	NV	MCL= 60	-----	-----	34
Dibromochloromethane	124-48-1	M	S	MV	MCL= 80	0.045	-----	28
Dibutyltin	1002-53-5	MM to M	S	MV	-----	-----	-----	26
Dichloroacetic acid	79-43-6	M	VS	NV	MCL= 60	0.31	-----	34
Dichloromethane	75-09-2	M to HM	VS	VV	MCL= 5	2.7	5	1
Dichlorprop	120-36-5	MM to M	MS	NV	-----	-----	-----	39
Diclofenac	15307-86-5	MM	SS	NV	-----	-----	-----	1
Dieldrin	60-57-1	BM to SM	NS	V	-----	0.069	0.004	26
Dienochlor	2227-17-0	IM	NS	MV	-----	-----	-----	29
Diethyl phthalate (DEP)	84-66-2	SM to M	S	SV	-----	610	-----	28
Dihydrotestosterone	521-18-6	SM to MM	SS	NV	-----	-----	-----	31
Diisobutyl phthalate (DiBP)	84-69-5	IM to SM	SS	SV	-----	-----	-----	28
Diisodecyl phthalate (DiDP)	26761-40-0	IM	SS	SV	-----	-----	-----	28
Diisononyl phthalate (DiNP)	28553-12-0	IM	SS	SV	-----	-----	-----	28
Diltiazem	42399-41-7	SM to MM	MS	NV	-----	-----	-----	17
Dimethenamid	87674-68-8	MM to M	MS	NV	-----	-----	-----	8
Dimethyl phthalate (DMP)	131-11-3	MM to M	S	SV	-----	-----	-----	28
Dimethyltin	23120-99-2	M to HM	VS	V	-----	-----	-----	26
Dimetridazole	551-92-8	M	VS	NV	-----	-----	-----	37

**Table A1. Possible stormwater organic contaminants. Criteria for the classification of mobility, solubility, and volatility can be found in Table A2.**

Compound	CAS #	Mobility <sup>A</sup>	Solubility <sup>B</sup>	Volatility <sup>C</sup>	Drinking Water Standard <sup>D</sup>	Superfund Regional Screening Levels for Groundwater Protection <sup>E</sup>	NY State Groundwater Effluent Limits <sup>F</sup>	Reference
					µg/L	µg/kg	µg/L	
Di-n-butyl phthalate (DBP)	84-74-2	SM	SS	SV	-----	230	50	28
Di-n-octyl phthalate (DnOP)	117-84-0	IM to BM	NS	SV	-----	5700	-----	28
Diocetyl tin	94410-05-6	BM to SM	SS to NS	V	-----	-----	-----	26
Diphenhydramine	58-73-1	MM	S	NV	-----	-----	-----	17,18
Dipropyl phthalate (DPP)	131-16-8	MM	SS	NV	-----	-----	-----	28
Diuron	330-54-1	MM to M	SS	NV	-----	1.5	-----	26
D-limonene	5989-27-5	SM	SS	V	-----	-----	-----	17
Endrin	72-20-8	BM	NS	SV	MCL= 2	9.2	Not Detectable	26
Epitestosterone	481-30-1	SM to MM	SS	NV	-----	-----	-----	31
Erythromycin	114-07-8	MM	SS	NV	-----	-----	-----	37,40
Estriol	50-27-1	SM	SS	NV	-----	-----	-----	28
Estrone	53-16-7	BM to MM	SS	NV	-----	-----	-----	28
Ethylbenzene	100-41-4	MM	MS	V	MCL= 700	1.7	-----	14,29
Ethylene thiourea	96-45-7	M	VS	NV	-----	0.036	Not Detectable	38
Flumequine	42835-25-6	BM to SM	S	NV	-----	-----	-----	37
Fluoranthrene	206-44-0	IM to BM	SS	V	-----	8900	-----	1,5,15
Fluorene	86-73-7	IM to SM	SS	V	-----	540	-----	5
Fluoxetine	54910-89-3	BM to SM	SS	NV	-----	-----	-----	17,21,23,32,40
Formaldehyde	50-00-0	HM	VS	SV	-----	80	8	38
Galaxolide	1222-05-5	BM	SS	MV	-----	-----	-----	28
Gemfibrozil	25812-30-0	MM	SS	NV	-----	-----	-----	28
Gestodene	60282-87-3	SM to MM	SS	NV	-----	-----	-----	38

**Table A1. Possible stormwater organic contaminants. Criteria for the classification of mobility, solubility, and volatility can be found in Table A2.**

Compound	CAS #	Mobility <sup>A</sup>	Solubility <sup>B</sup>	Volatility <sup>C</sup>	Drinking Water Standard <sup>D</sup>	Superfund Regional Screening Levels for Groundwater Protection <sup>E</sup>	NY State Groundwater Effluent Limits <sup>F</sup>	Reference
					µg/L	µg/kg	µg/L	
Glyphosate	1071-83-6	SM	VS	NV	MCL= 700	880	-----	26
Hydroxyatrazine	67-68-5	MM to HM	VS	NV	-----	-----	-----	1,28
Hydroxydeethylatrazine	19988-24-0	MM to HM	VS	NV	-----	-----	-----	1,28
Hydroxydeisopropylatrazine	7313-54-4	MM to HM	VS	NV	-----	-----	-----	1,28
Ibuprofen	15687-27-1	SM	SS	SV	-----	-----	-----	28
Ifosfamide	3778-73-2	M	S	NV	-----	-----	-----	40
Indeno pyrene	193-39-5	IM	NS	SV	-----	240	-----	6
Indole	120-72-9	MM	S	SV	-----	-----	-----	17
Indomethacin	53-86-1	SM to MM	SS	NV	-----	-----	-----	1
Iohexol	66108-95-0	M to HM	MS	NV	-----	-----	-----	39
Iomeprol	78649-41-9	M to HM	MS	NV	-----	-----	-----	39
Iopamidol	60166-93-0	M to HM	VS	NV	-----	-----	-----	39
Iopromide	73334-07-3	M to HM	SS	NV	-----	-----	-----	1
Isophorone	78-59-1	MM	VS	SV	-----	26	-----	17
Isopropyl chloride	75-29-6	M	S	V	-----	-----	-----	29
Isoproturon	34123-59-6	MM	MS	NV	-----	-----	-----	26
Isoquinoline	119-65-3	SM to M	S	NV	-----	-----	-----	17
Ketoprofen	22071-15-4	MM	MS	NV	-----	-----	-----	28
Lincomycin	154-21-2	M	MS	NV	-----	-----	-----	37
Lindane	58-89-9	BM to HM	SS	V	MCL= 0.2	0.24	0.05	1,5,15
Lomefloxacin	98079-51-7	M to HM	VS	NV	-----	-----	-----	35
Malathion	121-75-5	BM to MM	MS	NV	-----	10	7	5
Mecoprop	93-65-2	M to HM	MS	NV	-----	0.46	-----	1,28

**Table A1. Possible stormwater organic contaminants. Criteria for the classification of mobility, solubility, and volatility can be found in Table A2.**

Compound	CAS #	Mobility <sup>A</sup>	Solubility <sup>B</sup>	Volatility <sup>C</sup>	Drinking Water Standard <sup>D</sup>	Superfund Regional Screening Levels for Groundwater Protection <sup>E</sup>	NY State Groundwater Effluent Limits <sup>F</sup>	Reference
					µg/L	µg/kg	µg/L	
Metalaxyl	57837-19-1	MM to M	S	NV	-----	33	-----	8
Metaldehyde	108-62-3	MM	MS	MV	-----	-----	-----	26
Metazachlor	67129-08-2	SM to MM	MS	NV	-----	-----	-----	8
Metformin	1115-70-4	M	VS	NV	-----	-----	-----	38
Methyl dihydrojasmonate	24851-98-7	MM	SS	NV	-----	-----	-----	28
Methyl salicylate	119-36-8	MM to M	S	SV	-----	-----	-----	17
Methyl tert-butyl ether (MTBE)	1634-04-4	M to HM	VS	MV	-----	3.2	-----	1
Methylparaben	99-76-3	M	S	NV	-----	-----	-----	38
Methyltriclosan	4640-01-1	BM	SS	NV to SV	-----	-----	-----	39
Metolachlor	51218-45-2	SM to M	MS	NV	-----	320	10	8
Metoprolol	37350-58-6	M	VS	NV	-----	-----	-----	37,40
Metronidazole	443-48-1	M	VS	NV	-----	-----	-----	37,38
Miconazole	22916-47-8	IM to BM	NS	NV	-----	-----	-----	17
Monobromoacetic acid	79-08-3	M	VS	NV	MCL= 60	-----	-----	34
Monobutyltin	78763-54-9	MM to M	SS	MV	-----	-----	-----	26
Monochloroacetic acid	79-11-8	M	VS	NV	MCL= 60	0.81	-----	34
Musk ambrette	83-66-9	SM	SS	SV	-----	-----	-----	38
Musk ketone	81-14-1	SM	SS	NV	-----	-----	-----	38
Musk xylene	81-15-2	BM to SM	SS	NV	-----	-----	-----	38
m-xylene	108-38-3	MM	MS	V	MCL= 10000	19	-----	14,29
N,N-Diethyl-3-methylbenzamide (DEET)	134-62-3	MM	MS	NV	-----	-----	-----	39

**Table A1. Possible stormwater organic contaminants. Criteria for the classification of mobility, solubility, and volatility can be found in Table A2.**

Compound	CAS #	Mobility <sup>A</sup>	Solubility <sup>B</sup>	Volatility <sup>C</sup>	Drinking Water Standard <sup>D</sup>	Superfund Regional Screening Levels for Groundwater Protection <sup>E</sup>	NY State Groundwater Effluent Limits <sup>F</sup>	Reference
					µg/L	µg/kg	µg/L	
N4-acetyl-Sulfamethazine	100-90-3	MM to HM	VS	NV	-----	-----	-----	33
Nalidixic Acid	389-08-2	M	SS	NV	-----	-----	-----	37
Naphthalene	91-20-3	IM to MM	SS	VV	-----	0.54	-----	5,15
Naproxen	22204-53-1	MM	SS	NV	-----	-----	-----	28
Nitrilotriacetic acid	139-13-9	HM to MM	VS		-----	-----	-----	38
Nonylphenol	25154-52-3	BM	SS	SV	-----	-----	-----	28
Nonylphenol ethoxylate	9016-45-9	SM	SS	NV	-----	-----	-----	28
Norfloxacin	70458-96-7	BM	MS	NV	-----	-----	-----	37,42
N-propyl chloride	540-54-5	M	S	V	-----	-----	-----	29
Octylethoxylate	9002-93-1	SM to MM	SS	NV	-----	-----	-----	28
Octylphenol	949-13-3	BM	SS	NV	-----	-----	-----	28
Ofloxacin	82419-36-1	BM	VS	NV	-----	-----	-----	37
OH-ibuprofen	51146-55-5	M	S	NV	-----	-----	-----	28
op' DDA	3424-82-6	SM to MM	SS	NV	-----	-----	-----	1,28
Oxazepam	604-75-1	MM	SS	NV	-----	-----	-----	38
o-xylene	95-47-6	MM to M	MS	V	MCL= 10000	19	-----	14,29
Oxytetracycline	79-57-2	BM to MM	MS	NV	-----	-----	-----	33,42
Para-cresol	106-44-5	MM to M	VS	SV	-----	150	-----	17
Para-tert-octylphenol	140-66-9	BM	SS	SV	-----	-----	-----	26,39,40
Pentachlorophenol	87-86-5	IM to SM	SS	NV	MCL= 1	0.4	-----	1,5
Pentoxifylline	6493-05-6	M	MS	NV	-----	-----	-----	37

**Table A1. Possible stormwater organic contaminants. Criteria for the classification of mobility, solubility, and volatility can be found in Table A2.**

Compound	CAS #	Mobility <sup>A</sup>	Solubility <sup>B</sup>	Volatility <sup>C</sup>	Drinking Water Standard <sup>D</sup>	Superfund Regional Screening Levels for Groundwater Protection <sup>E</sup>	NY State Groundwater Effluent Limits <sup>F</sup>	Reference
					µg/L	µg/kg	µg/L	
Perfluorobutanesulfonic Acid (PFBS)	375-73-5	MM to M	MS	MV	----	21	----	37,40
Perfluorooctanionic Acid (PFOA)	335-67-1	MM to M	S	V	----	----	----	37,40
Perfluorodecanoic Acid (PFDA)	335-76-2	IM to SM	NS	V	----	----	----	37
Perfluorododecanoic Acid (PFDoA)	307-55-1	IM to BM	NS	V	----	----	----	37
Perfluoroheptanoic Acid (PFHpA)	375-85-9	BM to M	SS	V	----	----	----	37
Perfluorohexanesulfonic Acid (PFHxS)	355-46-4	HM	SS	MV	----	----	----	37
Perfluorohexanoic Acid (PFHxA)	307-24-4	BM to M	VS	SV	----	----	----	37
Perfluorononanoic Acid (PFNA)	375-95-1	IM to MM	NS	MV	----	----	----	37,40
Perfluorooctanesulfonic Acid (PFOS)	1763-23-1	BM to MM	NS	NV	----	----	----	37,40
Perfluoroundecanoic Acid (PFUnA)	4234-23-5	SM to HM	NS	NV	----	----	----	37
Perylene	198-55-0	IM	NS	SV	----	----	----	25
Phenanthrene	85-01-8	IM to SM	SS	V	----	----	----	1,5,15
Phenol	108-95-2	SM	VS	MV	----	330	----	1
Pipemidic Acid	51940-44-4	M to HM	VS	NV	----	----	----	37
pp' DDA	83-05-6	SM to MM	SS	NV	----	----	----	1,28
Primidone	125-33-7	MM to M	MS	NV	----	----	----	1
Progesterone	57-83-0	SM	SS	NV	----	----	----	38

**Table A1. Possible stormwater organic contaminants. Criteria for the classification of mobility, solubility, and volatility can be found in Table A2.**

Compound	CAS #	Mobility <sup>A</sup>	Solubility <sup>B</sup>	Volatility <sup>C</sup>	Drinking Water Standard <sup>D</sup>	Superfund Regional Screening Levels for Groundwater Protection <sup>E</sup>	NY State Groundwater Effluent Limits <sup>F</sup>	Reference
					µg/L	µg/kg	µg/L	
Propazine	139-40-2	MM to M	SS	NV	-----	30	-----	1,28
Propranolol	525-66-6	MM	MS	NV	-----	-----	-----	31,39,40
Propylparaben	94-13-3	MM	MS	NV	-----	-----	-----	38
Propyphenazone	479-92-5	MM to M	MS	NV	-----	-----	-----	1
p-xylene	106-42-3	MM	MS	V	MCL= 10000	19	-----	14,29
Pyrene	129-00-0	IM to SM	SS	V	-----	1300	-----	5,15
Roxithromycin	80214-83-1	SM to HM	NS	NV	-----	-----	-----	37
Salicylic acid	69-72-7	MM	S	NV	-----	-----	-----	28
Sertaline	79617-96-2	IM to SM	S	NV	-----	-----	-----	17
Simazine	122-34-9	SM to M	SS	NV	MCL= 4	0.3	0.5	1,28
Skatole	83-34-1	MM	MS	SV	-----	-----	-----	17
Sodium alkylbenzene sulfonate	68411-30-3	-----	-----	-----	-----	-----	-----	16,22
Sodium decylbenzenesulfonate	1322-98-1	SM to MM	MS	NV	-----	-----	-----	16,22
Sodium dodecylbenzenesulfonate	25155-30-0	BM to MM	MS	NV	-----	-----	-----	16,22
Sodium N-tridecylbenzenesulfonate	26248-24-8	BM to MM	SS	NV	-----	-----	-----	16,22
Sodium Undecylbenzenesulfonate	27636-75-5	SM to MM	SS	NV	-----	-----	-----	16,22
Sotalol	3930-20-9	M to HM	MS	NV	-----	-----	-----	37
Sucralose	56038-13-2	M	VS	NV	-----	-----	-----	39
Sulfadiazine	68-35-9	M	SS	NV	-----	-----	-----	37,42

**Table A1. Possible stormwater organic contaminants. Criteria for the classification of mobility, solubility, and volatility can be found in Table A2.**

Compound	CAS #	Mobility <sup>A</sup>	Solubility <sup>B</sup>	Volatility <sup>C</sup>	Drinking Water Standard <sup>D</sup>	Superfund Regional Screening Levels for Groundwater Protection <sup>E</sup>	NY State Groundwater Effluent Limits <sup>F</sup>	Reference
					µg/L	µg/kg	µg/L	
Sulfadimethoxine	122-11-2	MM to M	MS	NV	-----	-----	-----	33
Sulfamethazine	57-68-1	MM to M	S	NV	-----	-----	-----	33,37
Sulfamethoxazole	723-46-6	M	MS	NV	-----	-----	-----	1
Sulfamonomethoxine	1220-83-3	M	S	NV	-----	-----	-----	37
Sulfaniamide	63-74-1	M	S	NV	-----	-----	-----	32
Sulfathiazole	72-14-0	MM to M	MS	NV	-----	-----	-----	33,37
Terbutylazine	5915-41-3	MM	SS	NV	-----	-----	-----	8
Terbutryn	886-50-0	BM to MM	SS	NV	-----	1.9	-----	39
Tert-Amyl chloride	594-36-5	MM to M	MS	MV	-----	-----	-----	17
Tertrabutyltin	1461-25-2	IM to BM	NS	V	-----	-----	-----	26
Testosterone	58-22-0	SM	SS	NV	-----	-----	-----	31
Tetrachloroethylene	127-18-4	MM	MS	V	MCL= 5	1.8	-----	26
Tetracycline	60-54-8	M to HM	MS	NV	-----	-----	-----	33,42
Thiabendazole	148-79-8	SM	SS	NV	-----	-----	-----	17
Toluene	108-88-3	MM to M	MS	V	MCL= 1000	76	-----	26
Tolyfluanid	731-27-1	SM	SS	SV	-----	-----	-----	29
Tonalide	21145-77-7	BM to SM	SS	MV	-----	-----	-----	28
Tramadol	27203-92-5	MM	S	NV	-----	-----	-----	38
Tribromoacetic acid	75-96-7	HM	VS	NV	-----	-----	-----	34
Tribromomethane	75-25-2	MM	S	MV	MCL= 80	2.4	-----	28
Tributyl phosphate	126-73-8	SM	MS	SV	-----	25	-----	39
Tributyltin	688-73-3	SM	SS	V	-----	8.2	-----	26,29

**Table A1. Possible stormwater organic contaminants. Criteria for the classification of mobility, solubility, and volatility can be found in Table A2.**

Compound	CAS #	Mobility <sup>A</sup>	Solubility <sup>B</sup>	Volatility <sup>C</sup>	Drinking Water Standard <sup>D</sup>	Superfund Regional Screening Levels for Groundwater Protection <sup>E</sup>	NY State Groundwater Effluent Limits <sup>F</sup>	Reference
					µg/L	µg/kg	µg/L	
Trichloroacetic acid	76-03-9	MM	VS	NV	MCL= 60	0.22	-----	34
Trichloroethylene	79-01-6	SM to M	S	V	MCL= 5	0.1	5	29
Trichloromethane	67-66-3	MM to M	S	V	MCL= 80	0.061	7	28
Triclosan	3380-34-5	BM to SM	SS	NV	-----	-----	-----	28
Tricyclohexyltin chloride	3091-32-5	IM	SS to NS	SV	-----	-----	-----	26
Tricyclohexyltin Hydroxide	13121-70-5	IM to BM	SS to NS	NV	-----	-----	-----	26
Triethylphosphate	78-40-0	M	VS	NV	-----	-----	-----	39
Trifluoromethane	75-46-7	M	S	V	MCL= 80	-----	-----	28
Triiodomethane	75-47-8	M	MS to SS	MV	MCL= 80	-----	-----	28
Triisobutyl phosphate	126-71-6	SM	MS to SS	NV	-----	-----	-----	39
Trimethoprim	738-70-5	M	MS	NV	-----	-----	-----	17,37
Trimethyltin Chloride	1066-45-1	M to HM	VS	MV	-----	-----	-----	26
Tri-n-Butyltin Hydride	688-73-3	SM	SS	V	-----	8.2	-----	26
Trioctyltin chloride	2587-76-0	IM	NS	MV	-----	-----	-----	26
Triphenyl phosphate	115-86-6	SM	SS	SV	-----	-----	-----	39
Triphenylphosphine oxide	791-28-6	SM to MM	MS to SS	NV	-----	150	-----	39
Triphenyltin Acetate	900-95-8	BM to M	SS	NV	-----	-----	-----	26
Triphenyltin chloride	639-58-7	IM to SM	SS	NV	-----	-----	-----	26
Triphenyltin hydride	892-20-6	IM to SM	SS	NV	-----	-----	-----	26
Triphenyltin hydroxide	76-87-9	SM	SM	NV	-----	-----	-----	26
Tris(1,3-dichloroisopropyl)phosphate	13674-87-8	SM	SS	NV	-----	800	-----	39
Tris(2-butoxyethyl)phosphate	78-51-3	SM	S	NV	-----	-----	-----	31,39

**Table A1. Possible stormwater organic contaminants. Criteria for the classification of mobility, solubility, and volatility can be found in Table A2.**

Compound	CAS #	Mobility <sup>A</sup>	Solubility <sup>B</sup>	Volatility <sup>C</sup>	Drinking Water Standard <sup>D</sup>	Superfund Regional Screening Levels for Groundwater Protection <sup>E</sup>	NY State Groundwater Effluent Limits <sup>F</sup>	Reference
					µg/L	µg/kg	µg/L	
Tris(2-chloroethyl)phosphate	115-96-8	MM	S	SV	-----	3.8	-----	39
Tylosin	1401-69-0	BM to MM	S	NV	-----	-----	-----	37
Valsartan	137862-53-4	BM to MM	SS	NV	-----	-----	-----	18,41
Venlafaxine	93413-69-5	MM	MS	NV	-----	-----	-----	17,32,40
Xylene	1330-20-7	MM	MS	V	MCL=10000	19	-----	26
α-BHC	319-84-6	SM	SS	SV	-----	-----	0.01	1
α-Endosulfan	959-98-8	SM	SS	NV	-----	-----	-----	1
α-Hexachloro-cyclohexane	319-84-6	SM	SS	SV	-----	0.041	0.01	1

<sup>A</sup>Mobility Classes: HM= Highly Mobile, M= Mobile, MM= Moderately Mobile, SM= Slightly Mobile, BM= Hardly Mobile, IM= Immobile;

<sup>B</sup>Solubility Classes: NS= Negligible Soluble, SS= Slightly Soluble, MS= Moderately Soluble, S= Soluble, VS= Very Soluble;

<sup>C</sup>Volatility Classes: VV= Very Volatile, V= Volatile, MV= Moderately Volatile, SV= Slightly Volatile, NV= Nonvolatile;

<sup>D</sup>Safe Drinking Water Act, 42 U.S.C. §300f et seq. (1974): <https://www.epa.gov/laws-regulations/summary-safe-drinking-water-act>;

<sup>E</sup>Regional Screening Levels for Chemical Contaminants at Superfund Sites as of May 2018, Risk-based SSL for Protection of Groundwater; <https://www.epa.gov/risk/regional-screening-levels-rsls-generic-tables>; and

<sup>F</sup>Rules and Regulations of the State of New York: 6CRR-NY 703.6 Groundwater effluent limitations for discharges to class GA waters.

References:

1. US EPA, 1983; 2. Chang et al., 1990; 3. Ehrenfeld et al., 1991; 4. Mineart et al., 1994; 5. Pitt et al., 1994; 6. Bannerman et al., 1996; 7. Bell et al., 1995; 8. Buchelli et al., 1998; 9. Ellis, J.B., 2000; 10. Horsley, 2000; 11. Maryland Department of Environment, 2000; 12. Schueler and Holland, 2000; 13. Breault and Granato, 2003; 14. Granato et al., 2003; 15. Clark and Pitt, 2007; 16. Eriksson et al., 2007; 17. Nilsen et al., 2007; 18. Kasprzyk-Hordern et al., 2008; 19. Whittemore, 2008; 20. Diblasi et al., 2009; 21. Radjenović et al., 2009; 22. Hartmann et al., 2005; 23. Metcalfe et al., 2010; 24. Sui et al., 2010; 25. Watts et al., 2010; 26. Zgheib et al., 2012; 27. Writer et al., 2011; 28. Barceló, 2012; 29. Gasperi et al., 2012; 30. Mahler et al., 2012; 31. Philips et al., 2012; 32. Sprague, 2012; 33. Krein et al., 2013; 34. Nguyen et al., 2013; 35. Dodder et al., 2014; 36. Leisenring et al., 2014; 37. Lin et al., 2015; 38. Lopez et al., 2015; 39. Launay et al., 2016 40. Wilkinson et al., 2016; 41. McEachran et al., 2017; 42. Zhang et al., 2016; 43. National Stormwater Quality Database, Version 3.0

**Table A2. Classification criteria used for solubility, volatility, and mobility in Table A1.**

Solubility Classifications <sup>1</sup>		Volatility from Water to the gas phase Classifications <sup>1</sup>		Mobility Classifications <sup>2</sup>	
Solubility, log C <sub>w</sub> <sup>3</sup> (mg/L)	Classification	Log K <sub>H</sub> <sup>4</sup> (atm-m <sup>3</sup> /mole)	Classification	Log K <sub>oc</sub> (L/kg)	Classification
<-1	Negligible Soluble	>-1	Very Volatile	<1	Highly Mobile
-1 – 2	Slightly Soluble	-1 – -3	Volatile	1 – 2	Mobile
2 – 3	Moderately Soluble	-3 – -5	Moderately Volatile	2 – 3	Moderately Mobile
3 – 4	Soluble	-5 – -7	Slightly Volatile	3 – 4	Slightly Mobile
>4	Very Soluble	<-7	Nonvolatile	4 – 5	Hardly Mobile
				>5	Immobile

<sup>1</sup> U.S. EPA. 2012.

<sup>2</sup> FAO. 2000.

<sup>3</sup> C<sub>w</sub>= water solubility.

<sup>4</sup> K<sub>H</sub>= Henry's Law Constant.

**Table A3. Possible Inorganic contaminants in stormwater.**

Compound	CAS #	Drinking Water Standard <sup>A</sup>	Superfund Regional Screening Levels for Groundwater Protection <sup>B</sup>	NY State Groundwater Effluent Limits <sup>C</sup>	Reference
		µg/L	mg/kg	µg/L	
Aluminum (Al)	7429-90-5	sMCL=50 to 200	3000	2000	9
Ammonia (NH <sub>3</sub> )	7664-41-7	-----	-----	-----	2,12
Antimony (Sb)	7440-36-0	-----	0.035	6	1,13
Arsenic (As)	7440-38-2	MCL= 10	0.0015	50	1,15,36
Barium (Ba)	7440-39-3	MCL= 2000	16	2000	19
Beryllium (Be)	7440-41-7	MCL= 4	1.9	-----	1
Biological Oxygen Demand (BOD)	-----	-----	-----	-----	2,12
Boron (B)	7440-42-8	-----	1.3	-----	13
Bromide (Br)	24959-67-9	-----	-----	-----	13
Cadmium (Cd)	7440-43-9	MCL= 5	0.069	10	1,5,11,12,15
Calcium (Ca)	7440-70-2	-----	-----	-----	13
Cerium (Ce)	7440-45-1	-----	-----	-----	13
Chloride (Cl <sup>-</sup> )	16887-00-6	sMCL=250000	-----	500000	3,5,11,15
Chromium (Cr)	7440-47-3	MCL= 100	4000000	-----	1,5,10,15
Cobalt (Co)	7440-48-4	-----	0.027	-----	13
Chemical Oxygen Demand (COD)	-----	-----	-----	-----	2,5,12
Copper (Cu)	7440-50-8	TT action level= 1300; sMCL=1000	2.8	400	1,2,11,12
Cyanide (CN <sup>-</sup> )	57-12-5	MCL= 200	0.0015	400	1
Fluoride (F <sup>-</sup> )	16984-48-8	MCL=4000; sMCL=2000	12	3000	19
Iodide (I <sup>-</sup> )	20461-54-5	-----	-----	-----	13
Iron (Fe)	7439-89-6	sMCL=300	35	600	2,19
Lead (Pb)	7439-92-1	TT action level= 15	14	50	1,2,5,11,13,15
Lithium (Li)	7439-93-2	-----	1.2	-----	13
Magnesium (Mg)	7439-95-4	-----	-----	-----	13

**Table A3. Possible Inorganic contaminants in stormwater.**

Compound	CAS #	Drinking Water Standard <sup>A</sup>	Superfund Regional Screening Levels for Groundwater Protection <sup>B</sup>	NY State Groundwater Effluent Limits <sup>C</sup>	Reference
		µg/L	mg/kg	µg/L	
Manganese (Mn)	7439-96-5	sMCL=50	2.8	600	9
Mercury (Hg)	7439-97-6	MCL= 2	0.0033	1.4	19
Molybdenum (Mo)	7439-98-7	-----	0.2	-----	13
Nickel (Ni)	7440-02-0	-----	2.6	200	1,15
Nitrate (NO <sub>3</sub> <sup>-</sup> )	14797-55-88	MCL= 10000 (as N)	-----	20000	2,5,12,15
Nitrite (NO <sub>2</sub> <sup>-</sup> )	14797-65-0	MCL= 1000 (as N)	-----	2000	2,5,12,15
Palladium (Pd)	7440-05-03	-----	-----	-----	13
Phosphate (PO <sub>4</sub> <sup>3-</sup> )	14265-44-2	-----	-----	-----	2,43
Platinum (Pt)	7440-06-4	-----	-----	-----	9
Potassium (K)	7440-09-7	-----	-----	-----	13
Rhodium (Rh)	7440-16-6	-----	-----	-----	9
Selenium (Se)	7782-49-2	MCL= 50	0.052	20	1,13
Silicon (Si)	7440-21-3	-----	-----	-----	13
Silver (Ag)	7440-22-4	sMCL=100	0.08	100	19
Sodium (Na)	7440-23-5	-----	-----	-----	13
Strontium (Sr)	7440-24-6	-----	42	-----	13
Sulfate (SO <sub>4</sub> <sup>2-</sup> )	14808-79-8	sMCL=250000	-----	500000	19
Tin (Sn)	7440-31-5	-----	-----	-----	13
Titanium (Ti)	7440-32-6	-----	-----	-----	13
Total Dissolved Solids (TDS)	-----	sMCL=500000	-----	-----	36
Total Kjeldahl N	-----	-----	-----	-----	2,12,43
Total Nitrogen	-----	-----	-----	-----	11,12
Total Organic Carbon	-----	-----	-----	-----	2,11
Total Phosphorus	-----	-----	-----	-----	11,12,15
Total Suspended Solids	-----	-----	-----	-----	2,7,11,12,15
Tungsten (W)	7440-33-7	-----	-----	-----	13

**Table A3. Possible Inorganic contaminants in stormwater.**

Compound	CAS #	Drinking Water Standard <sup>A</sup>	Superfund Regional Screening Levels for Groundwater Protection <sup>B</sup>	NY State Groundwater Effluent Limits <sup>C</sup>	Reference
		µg/L	mg/kg	µg/L	
Turbidity	-----	-----	-----	-----	36
Vanadium (V)	7440-62-2	-----	8.6	-----	13
Zinc (Zn)	7440-66-6	sMCL=5000	37	5000	1,2,5,11,12,15

<sup>A</sup>Safe Drinking Water Act, 42 U.S.C. §300f et seq. (1974): <https://www.epa.gov/laws-regulations/summary-safe-drinking-water-act>;

<sup>B</sup>Regional Screening Levels for Chemical Contaminants at Superfund Sites as of May 2018, Risk-based SSL for Protection of Groundwater: <https://www.epa.gov/risk/regional-screening-levels-rsls-generic-tables>; and

<sup>C</sup>Rules and Regulations of the State of New York: 6CRR-NY 703.6 Groundwater effluent limitations for discharges to class GA waters.

References:

1. US EPA, 1983; 2. Chang et al., 1990; 3. Ehrenfeld et al., 1991; 4. Mineart et al., 1994; 5. Pitt et al., 1994; 6. Bannerman et al., 1996; 7. Bell et al., 1995; 8. Buchelli et al., 1998; 9. Ellis, J.B., 2000; 10. Horsley, 2000; 11. Maryland Department of Environment, 2000; 12. Schueler and Holland, 2000; 13. Breault and Granato, 2003; 14. Granato et al., 2003; 15. Clark and Pitt, 2007; 16. Eriksson et al., 2007; 17. Nilsen et al., 2007; 18. Kasprzyk-Hordern et al., 2008; 19. Whittemore, 2008; 20. Diblasi et al., 2009; 21. Radjenović et al., 2009; 22. Hartmann et al., 2005; 23. Metcalfe et al., 2010; 24. Sui et al., 2010; 25. Watts et al., 2010; 26. Zgheib et al., 2012; 27. Writer et al., 2011; 28. Barceló, 2012; 29. Gasperi et al., 2012; 30. Mahler et al., 2012; 31. Philips et al., 2012; 32. Sprague, 2012; 33. Krein et al., 2013; 34. Nguyen et al., 2013; 35. Dodder et al., 2014; 36. Leisenring et al., 2014; 37. Lin et al., 2015; 38. Lopez et al., 2015; 39. Launay et al., 2016 40. Wilkinson et al., 2016; 41. McEachran et al., 2017; 42. Zhang et al., 2016; 43. National Stormwater Quality Database, Version 3.0

**Table A4. Select properties of organic contaminants potentially found in stormwater.**

Compound	CAS #	Chemical Formula	Molecular Weight	T <sub>m</sub> <sup>1</sup>	T <sub>b</sub> <sup>2</sup>	Log C <sub>w</sub> <sup>3</sup>
Units			g/mol	°C	°C	mg/L
1,1,1,2-tetrachloroethane	630-20-6	C <sub>2</sub> H <sub>2</sub> Cl <sub>4</sub>	167.85	-70.2 <sup>A</sup>	130.2 <sup>A</sup>	3.03 <sup>A</sup>
1,1,1-trichloroethane	71-55-6	C <sub>2</sub> H <sub>3</sub> Cl <sub>3</sub>	133.40	-30 <sup>A</sup>	74 <sup>A</sup>	3.11 <sup>A</sup>
1,1,2,2-tetrachloroethane	79-34-5	C <sub>2</sub> H <sub>2</sub> Cl <sub>4</sub>	167.85	-42.3 <sup>A</sup>	146 <sup>A</sup>	3.45 <sup>A</sup>
1,1,2-trichloroethane	79-00-5	C <sub>2</sub> H <sub>3</sub> Cl <sub>3</sub>	133.40	-35 <sup>A</sup>	113 - 114 <sup>A</sup>	3.66 <sup>A</sup>
1,1-dichloroethane	75-34-3	C <sub>2</sub> H <sub>4</sub> Cl <sub>2</sub>	98.96	-96.93 <sup>A</sup>	57.4 <sup>A</sup>	3.7 <sup>A</sup>
1,2,3-trichloropropane	96-18-4	C <sub>3</sub> H <sub>5</sub> Cl <sub>3</sub>	147.43	-13.9 <sup>A</sup>	158 <sup>A</sup>	3.24 <sup>A</sup>
1,2,3-trimethylbenzene	526-73-8	C <sub>9</sub> H <sub>12</sub>	120.19	-25.4 <sup>A</sup>	176.12 <sup>A</sup>	1.88 <sup>A</sup>
1,2,4-trichlorobenzene	120-82-1	C <sub>6</sub> H <sub>3</sub> Cl <sub>3</sub>	181.45	16.9 <sup>A</sup>	213.5 <sup>A</sup>	1.69 <sup>A</sup>
1,2,4-trimethylbenzene	95-63-6	C <sub>9</sub> H <sub>12</sub>	120.19	-43.77 <sup>A</sup>	168.89 <sup>A</sup>	1.76 <sup>A</sup>
1,2-dichlorobenzene	95-50-1	C <sub>6</sub> H <sub>4</sub> Cl <sub>2</sub>	147.01	-16.7 <sup>A</sup>	180.1 <sup>A</sup>	2.19 <sup>A</sup>
1,2-dichloroethane	107-06-2	C <sub>2</sub> H <sub>4</sub> Cl <sub>2</sub>	98.96	-35.3 <sup>A</sup>	83.5 <sup>A</sup>	3.93 <sup>A</sup>
1,2-dichloropropane	78-87-5	C <sub>3</sub> H <sub>6</sub> Cl <sub>2</sub>	112.98	-100.4 <sup>A</sup>	96.4 <sup>A</sup>	3.45 <sup>A</sup>
1,3-dichlorobenzene	541-73-1	C <sub>6</sub> H <sub>4</sub> Cl <sub>2</sub>	147.00	-24.8 <sup>A</sup>	173 <sup>A</sup>	2.1 <sup>A</sup>
1,3-dichloropropane	142-28-9	C <sub>3</sub> H <sub>6</sub> Cl <sub>2</sub>	112.98	-99.5 <sup>A</sup>	120.4 <sup>A</sup>	3.44 <sup>A</sup>
1,4-dichlorobenzene	106-46-7	C <sub>6</sub> H <sub>4</sub> Cl <sub>2</sub>	147.01	53.09 <sup>A</sup>	174 <sup>A</sup>	1.88 <sup>A</sup>
11-keto testosterone	53187-98-7	C <sub>19</sub> H <sub>26</sub> O <sub>3</sub>	302.41	173.50 (est) <sup>C</sup>	422.75 (est) <sup>C</sup>	2.94 (est) <sup>C</sup>
17α-estradiol	57-91-0	C <sub>18</sub> H <sub>24</sub> O <sub>2</sub>	272.39	221.5 <sup>C</sup>	395.47 (est) <sup>C</sup>	1.91 <sup>C</sup>
17α-ethynyl estradiol	57-63-6	C <sub>20</sub> H <sub>24</sub> O <sub>2</sub>	296.40	183 <sup>C</sup>	411.21 (est) <sup>C</sup>	2.07 <sup>C</sup>
17β-estradiol	50-28-2	C <sub>18</sub> H <sub>24</sub> O <sub>2</sub>	272.39	221.5 <sup>C</sup>	395.47 (est) <sup>C</sup>	1.91 <sup>C</sup>
19-norethisterone	68-22-4	C <sub>20</sub> H <sub>26</sub> O <sub>2</sub>	298.42	204 <sup>A</sup>	400.27 (est) <sup>C</sup>	0.85 <sup>A</sup>
1-acetyl-1-methyl-2-dimethyl-oxamoyl-2-phenylhydrazide	519-65-3	C <sub>13</sub> H <sub>17</sub> N <sub>3</sub> O <sub>3</sub>	263.29	178.74 (est) <sup>C</sup>	434.89 (est) <sup>C</sup>	3.64 (est) <sup>C</sup>
1-chloro-2-methylpropane	513-36-0	C <sub>4</sub> H <sub>9</sub> Cl	92.57	-130.3 <sup>C</sup>	68.5 <sup>C</sup>	2.97 (est) <sup>C</sup>
1-chlorobutane	109-69-3	C <sub>4</sub> H <sub>9</sub> Cl	92.57	-123.1 <sup>A</sup>	78.5 <sup>A</sup>	3.04 <sup>A</sup>

**Table A4. Select properties of organic contaminants potentially found in stormwater.**

Compound	CAS #	Chemical Formula	Molecular Weight	T <sub>m</sub> <sup>1</sup>	T <sub>b</sub> <sup>2</sup>	Log C <sub>w</sub> <sup>3</sup>
Units			g/mol	°C	°C	mg/L
1-chloropentane	543-59-9	C <sub>5</sub> H <sub>11</sub> Cl	106.59	-99 <sup>A</sup>	107.8 <sup>A</sup>	2.29 <sup>A</sup>
1H-benzotrazole	95-14-7	C <sub>6</sub> H <sub>5</sub> N <sub>3</sub>	119.13	98.5 <sup>A</sup>	204 <sup>A</sup>	4.3 <sup>A</sup>
1-methylnaphthalene	90-12-0	C <sub>11</sub> H <sub>10</sub>	142.20	-30.43 <sup>A</sup>	244 <sup>A</sup>	1.41 <sup>A</sup>
2, 4-D	94-75-7	C <sub>8</sub> H <sub>6</sub> Cl <sub>2</sub> O <sub>3</sub>	221.03	138 <sup>A</sup>	160 <sup>C</sup>	2.73 - 2.83 <sup>A</sup>
2,2',3,4,4',5,5'-heptachlorobiphenyl (PCB 180)	35065-29-3	C <sub>12</sub> H <sub>3</sub> Cl <sub>7</sub>	395.32	163.56 (est) <sup>C</sup>	415.60 (est) <sup>C</sup>	-3.55 (est) <sup>C</sup>
2,2',3,4,4',5-hexachlorobiphenyl (PCB 138)	35065-28-2	C <sub>12</sub> H <sub>4</sub> Cl <sub>6</sub>	360.88	146.34 (est) <sup>C</sup>	396.90 (est) <sup>C</sup>	-2.63 <sup>C</sup>
2,2',4,4',5,5'-hexachlorobiphenyl (PCB 153)	35065-27-1	C <sub>12</sub> H <sub>4</sub> Cl <sub>6</sub>	360.88	146.34 (est) <sup>C</sup>	396.90 (est) <sup>C</sup>	-2.89 <sup>C</sup>
2,2',5,5'-tetrachlorobiphenyl (PCB 52)	35693-99-3	C <sub>12</sub> H <sub>6</sub> Cl <sub>4</sub>	291.99	122.32 (est) <sup>C</sup>	359.51 (est) <sup>C</sup>	-1.07 <sup>C</sup>
2,3',4,4',5-pentachlorobiphenyl (PCB 118)	31508-00-6	C <sub>12</sub> H <sub>5</sub> Cl <sub>5</sub>	326.44	134.60 (est) <sup>C</sup>	378.21 (est) <sup>C</sup>	-2.15 <sup>C</sup>
2,4,4'-trichlorobiphenyl (PCB 28)	7012-37-5	C <sub>12</sub> H <sub>7</sub> Cl <sub>3</sub>	257.54	100.85 (est) <sup>C</sup>	340.70 (est) <sup>C</sup>	-0.47 <sup>C</sup>
2,4,5,2',5'-pentachlorobiphenyl (PCB 101)	37680-73-2	C <sub>12</sub> H <sub>5</sub> Cl <sub>5</sub>	326.44	134.60 (est) <sup>C</sup>	378.21 (est) <sup>C</sup>	-1.87 <sup>C</sup>
2-chloro-2-methylpropane	507-20-0	C <sub>4</sub> H <sub>9</sub> Cl	92.57	-26 <sup>C</sup>	50 <sup>C</sup>	3.00 <sup>C</sup>
2-chlorobutane	78-86-4	C <sub>4</sub> H <sub>9</sub> Cl	92.57	-140 <sup>C</sup>	68 <sup>C</sup>	3.10 <sup>C</sup>
2-Chloromethylphenol	40053-98-3	C <sub>7</sub> H <sub>7</sub> ClO	142.58	45.6 (est) <sup>C</sup>	242 (est) <sup>C</sup>	3.52 (est) <sup>C</sup>
2-methyl-4-chlorophenoxyacetic Acid	94-74-6	C <sub>9</sub> H <sub>9</sub> ClO <sub>3</sub>	200.62	118 - 119 <sup>A</sup>	286.74 <sup>C</sup>	2.8 <sup>A</sup>
2-methylnaphthalene	91-57-6	C <sub>11</sub> H <sub>10</sub>	142.2	34.6 <sup>A</sup>	241.1 <sup>A</sup>	1.39 <sup>A</sup>
2-methylthiobenzothiazole	615-22-5	C <sub>8</sub> H <sub>7</sub> NS <sub>2</sub>	181.28	44 <sup>C</sup>	310.03 (est) <sup>C</sup>	2.04 <sup>C</sup>
3-Chloromethylphenol	60760-06-7	C <sub>7</sub> H <sub>7</sub> ClO	142.58	45.6 (est) <sup>C</sup>	242 (est) <sup>C</sup>	3.52 (est) <sup>C</sup>
3-β-coprostanol	360-68-9	C <sub>27</sub> H <sub>48</sub> O	388.68	185.5 <sup>C</sup>	428.87 (est) <sup>C</sup>	-3.47 (est) <sup>C</sup>
4-Chloromethylphenol	35421-08-0	C <sub>7</sub> H <sub>7</sub> ClO	142.58	45.6 (est) <sup>C</sup>	242 (est) <sup>C</sup>	3.52 (est) <sup>C</sup>
4'-hydroxy-diclofenac	64118-84-9	C <sub>14</sub> H <sub>9</sub> Cl <sub>2</sub> NO <sub>3</sub>	310.13	193.26 (est) <sup>C</sup>	458.61 (est) <sup>C</sup>	1.25 <sup>C</sup>

**Table A4. Select properties of organic contaminants potentially found in stormwater.**

Compound	CAS #	Chemical Formula	Molecular Weight	T <sub>m</sub> <sup>1</sup>	T <sub>b</sub> <sup>2</sup>	Log C <sub>w</sub> <sup>3</sup>
Units			g/mol	°C	°C	mg/L
4-methyl-1H-benzotriazole	29878-31-7	C <sub>7</sub> H <sub>7</sub> N <sub>3</sub>	133.15	76 - 87 <sup>C</sup>	160 <sup>C</sup>	3.49 (est) <sup>C</sup>
4-Nitrophenol	100-02-7	C <sub>6</sub> H <sub>5</sub> NO <sub>3</sub>	139.11	113 - 114 <sup>A</sup>	279 <sup>A</sup>	4.19 <sup>A</sup>
4-nonylphenol	104-40-5	C <sub>15</sub> H <sub>24</sub> O	220.36	42 <sup>C</sup>	293 - 297 <sup>C</sup>	0.85 <sup>A</sup>
4-nonylphenoldiethoxylate	20427-84-3	C <sub>19</sub> H <sub>32</sub> O <sub>3</sub>	308.46	140.16 (est) <sup>C</sup>	404.90 (est) <sup>C</sup>	0.02 (est) <sup>C</sup>
4-nonylphenolmonoethoxylate	104-35-8	C <sub>17</sub> H <sub>28</sub> O <sub>2</sub>	264.40	116.18 (est) <sup>C</sup>	369.64 (est) <sup>C</sup>	0.04 (est) <sup>C</sup>
4-tert-Butylphenol	98-54-4	C <sub>10</sub> H <sub>14</sub> O	150.22	98 <sup>C</sup>	237 <sup>C</sup>	2.63 <sup>C</sup>
5-methyl-1H-benzotriazole	136-85-6	C <sub>7</sub> H <sub>7</sub> N <sub>3</sub>	133.15	80 - 82 <sup>C</sup>	210 - 212 (est) <sup>C</sup>	3.49 (est) <sup>C</sup>
Acebutolol	37517-30-9	C <sub>18</sub> H <sub>28</sub> N <sub>2</sub> O <sub>4</sub>	336.43	119 - 123 <sup>C</sup>	504.10 (est) <sup>C</sup>	2.41 <sup>C</sup>
Acenaphthene	83-32-9	C <sub>12</sub> H <sub>10</sub>	154.21	93 <sup>A</sup>	277.5 <sup>A</sup>	0.59 <sup>A</sup>
Acenaphthylene	208-96-8	C <sub>12</sub> H <sub>8</sub>	152.21	89.4 <sup>A</sup>	280 <sup>A</sup>	0.59 <sup>A</sup>
Acesulfame	33665-90-6	C <sub>4</sub> H <sub>4</sub> KNO <sub>4</sub> S	201.24	123.2 <sup>A</sup>	357.88 (est) <sup>C</sup>	5.77 <sup>A</sup>
Acetaldehyde	75-07-0	C <sub>2</sub> H <sub>4</sub> O	44.05	-123.4 <sup>A</sup>	20.8 <sup>A</sup>	6 <sup>A</sup>
Acetaminophen	103-90-2	C <sub>8</sub> H <sub>9</sub> NO <sub>2</sub>	151.16	168 <sup>A</sup>	340.65 (est) <sup>C</sup>	4.15 <sup>A</sup>
Acetophenone	98-86-2	C <sub>8</sub> H <sub>8</sub> O	120.15	20.5 <sup>A</sup>	202 <sup>A</sup>	3.79 <sup>A</sup>
Acrylamide	79-06-1	C <sub>3</sub> H <sub>5</sub> NO	71.08	84.5 <sup>A</sup>	192.6 <sup>A</sup>	5.57 <sup>A</sup>
Alachlor	15972-60-8	C <sub>14</sub> H <sub>20</sub> ClNO <sub>2</sub>	269.77	40 - 41 <sup>A</sup>	100 <sup>C</sup>	2.38 <sup>A</sup>
Aldrin	309-00-2	C <sub>12</sub> H <sub>8</sub> Cl <sub>6</sub>	364.90	104 <sup>A</sup>	329.77 (est) <sup>C</sup>	2.23 <sup>A</sup>
Alkylphenol ethoxylate	68412-54-4	C <sub>15</sub> H <sub>24</sub> O(C <sub>2</sub> H <sub>4</sub> O) <sub>9</sub>	617.60	140.16 (est) <sup>C</sup>	404.90 (est) <sup>C</sup>	0.04 - 0.02 (est) <sup>C</sup>
Aminotriazole	61-82-5	C <sub>2</sub> H <sub>4</sub> N <sub>4</sub>	84.08	159 <sup>A</sup>	258.30 (est) <sup>C</sup>	5.45 <sup>A</sup>
Amitriptyline	50-48-6	C <sub>20</sub> H <sub>23</sub> N	277.40	196.5 <sup>C</sup>	381.64 (est) <sup>C</sup>	0.99 <sup>A</sup>
Amoxicillin	26787-78-0	C <sub>16</sub> H <sub>19</sub> N <sub>3</sub> O <sub>5</sub> S	365.40	329.94 (est) <sup>C</sup>	657.45 (est) <sup>C</sup>	3.54 <sup>A</sup>
AMPA	74341-63-2	C <sub>7</sub> H <sub>10</sub> N <sub>2</sub> O <sub>4</sub>	186.17	437.63 (est) <sup>C</sup>	297.85 (est) <sup>C</sup>	4.33 (est) <sup>C</sup>
Androstenedione	63-05-8	C <sub>19</sub> H <sub>26</sub> O <sub>2</sub>	286.40	173-174 <sup>A</sup>	200 <sup>A</sup>	1.76 <sup>A</sup>
Anthracene	120-12-7	C <sub>14</sub> H <sub>10</sub>	178.23	216 <sup>A</sup>	341.3 <sup>A</sup>	-1.36 <sup>A</sup>

**Table A4. Select properties of organic contaminants potentially found in stormwater.**

Compound	CAS #	Chemical Formula	Molecular Weight	T <sub>m</sub> <sup>1</sup>	T <sub>b</sub> <sup>2</sup>	Log C <sub>w</sub> <sup>3</sup>
Units			g/mol	°C	°C	mg/L
Anthraquinone	84-65-1	C <sub>14</sub> H <sub>8</sub> O <sub>2</sub>	208.22	286 <sup>A</sup>	377 <sup>A</sup>	0.13 <sup>A</sup>
Atenolol	29122-68-7	C <sub>14</sub> H <sub>22</sub> N <sub>2</sub> O <sub>3</sub>	266.33	146 - 148 <sup>A</sup>	438.63 (est) <sup>C</sup>	4.12 <sup>A</sup>
Atrazine	1912-24-9	C <sub>8</sub> H <sub>14</sub> ClN <sub>5</sub>	215.68	173 <sup>A</sup>	205 <sup>A</sup>	1.54 <sup>A</sup>
Atrazine amide-I	142179-76-8	C <sub>7</sub> H <sub>10</sub> ClN <sub>5</sub> O	215.64	159.27 (est) <sup>C</sup>	385.84 (est) <sup>C</sup>	3.43 - 3.72 (est) <sup>C</sup>
Azithromycin	83905-01-5	C <sub>38</sub> H <sub>72</sub> N <sub>2</sub> O <sub>12</sub>	748.98	113-115 <sup>A</sup>	846.60 (est) <sup>C</sup>	0.37 <sup>A</sup>
Bentazone	25057-89-0	C <sub>10</sub> H <sub>12</sub> N <sub>2</sub> O <sub>3</sub> S	240.28	139.4 -141 <sup>A</sup>	415.31 (est) <sup>C</sup>	2.70 <sup>A</sup>
Benz(a)anthracene	56-55-3	C <sub>18</sub> H <sub>12</sub>	228.29	155 - 157 <sup>A</sup>	437.6 <sup>A</sup>	-2.03 <sup>A</sup>
Benzene	71-43-2	C <sub>6</sub> H <sub>6</sub>	78.11	5.6 <sup>A</sup>	80.08 <sup>A</sup>	3.25 <sup>A</sup>
Benzo(a)fluoranthene	203-33-8	C <sub>20</sub> H <sub>12</sub>	252.31	169.41 <sup>C</sup>	442.75 <sup>C</sup>	-1.69 <sup>C</sup>
Benzo(a)pyrene	50-32-8	C <sub>20</sub> H <sub>12</sub>	252.32	179 <sup>A</sup>	495 <sup>C</sup>	-2.79 <sup>A</sup>
Benzo(b)fluoranthene	205-99-2	C <sub>20</sub> H <sub>12</sub>	252.32	168.4 <sup>A</sup>	442.75 (est) <sup>C</sup>	-2.82 <sup>A</sup>
Benzo(e)pyrene	192-97-2	C <sub>20</sub> H <sub>12</sub>	252.32	177.5 <sup>C</sup>	310 - 312 <sup>C</sup>	-2.25 <sup>C</sup>
Benzo(j)fluoranthene	205-82-3	C <sub>20</sub> H <sub>12</sub>	252.31	166 <sup>C</sup>	442.75 (est) <sup>C</sup>	-1.97 <sup>C</sup>
Benzo(k)fluoranthene	207-08-9	C <sub>20</sub> H <sub>12</sub>	252.32	217 <sup>A</sup>	480 <sup>A</sup>	-2.12 <sup>A</sup>
Benzo(b)naphtho(2,3-d)thiophene	243-46-9	C <sub>16</sub> H <sub>10</sub> S	234.32	148.75 (est) <sup>C</sup>	404.91 (est) <sup>C</sup>	-1.21 <sup>C</sup>
Benzo(ghi)perylene	191-24-2	C <sub>22</sub> H <sub>12</sub>	276.34	278.3 <sup>A</sup>	550 <sup>A</sup>	-3.59 <sup>A</sup>
Benzophenone	119-61-9	C <sub>13</sub> H <sub>10</sub> O	182.22	48.5 <sup>A</sup>	305.9 <sup>A</sup>	2.14 <sup>A</sup>
Benzothiazole	95-16-9	C <sub>7</sub> H <sub>5</sub> NS	135.19	2 <sup>A</sup>	227 - 228 <sup>A</sup>	3.23 <sup>C</sup>
Benzyl acetate	140-11-4	C <sub>9</sub> H <sub>10</sub> O <sub>2</sub>	150.18	-51 <sup>A</sup>	213 <sup>A</sup>	3.21 <sup>C</sup>
Beta-sitosterol	83-46-5	C <sub>29</sub> H <sub>50</sub> O	414.72	147 <sup>C</sup>	448.98 (est) <sup>C</sup>	-4.33 (est) <sup>C</sup>
Bezafibrate	41859-67-0	C <sub>19</sub> H <sub>20</sub> ClNO <sub>4</sub>	361.82	186 <sup>C</sup>	538.10 (est) <sup>C</sup>	0.09 (est) <sup>C</sup>
Bis(2-chloroethyl) ether	111-44-4	C <sub>4</sub> H <sub>8</sub> Cl <sub>2</sub> O	143.01	-51.9 <sup>A</sup>	178.5 <sup>A</sup>	4.01 <sup>A</sup>
Bis(2-chloroisopropyl) ether	39638-32-9	C <sub>6</sub> H <sub>12</sub> Cl <sub>2</sub> O	171.06	-96.8 - 101.8 <sup>A</sup>	187.3 <sup>A</sup>	1.60 <sup>C</sup>
Bis(2-ethylhexyl) phthalate	117-81-7	C <sub>24</sub> H <sub>38</sub> O <sub>4</sub>	390.56	-55 <sup>A</sup>	384 <sup>A</sup>	-0.57 <sup>A</sup>

**Table A4. Select properties of organic contaminants potentially found in stormwater.**

Compound	CAS #	Chemical Formula	Molecular Weight	T <sub>m</sub> <sup>1</sup>	T <sub>b</sub> <sup>2</sup>	Log C <sub>w</sub> <sup>3</sup>
Units			g/mol	°C	°C	mg/L
Bis(tributyltin) oxide	56-35-9	C <sub>24</sub> H <sub>54</sub> OSn <sub>2</sub>	596.11	45 <sup>A</sup>	180 <sup>C</sup>	0.6 <sup>A</sup>
Bisphenol A	80-05-7	C <sub>15</sub> H <sub>16</sub> O <sub>2</sub>	228.29	153 <sup>A</sup>	360.5 <sup>A</sup>	2.48 <sup>A</sup>
Bromochloroacetic acid	5589-96-8	C <sub>2</sub> H <sub>2</sub> BrClO <sub>2</sub>	173.39	31.5 <sup>C</sup>	215 <sup>A</sup>	5.40 <sup>A</sup>
Bromodichloroacetic acid	71133-14-7	C <sub>2</sub> HBrCl <sub>2</sub> O <sub>2</sub>	207.83	47.94 (est) <sup>C</sup>	234.61 (est) <sup>C</sup>	3.69 <sup>A</sup>
Bromodichloromethane	75-27-4	CHBrCl <sub>2</sub>	163.80	-57 <sup>A</sup>	90 <sup>A</sup>	3.6 <sup>A</sup>
Butyl benzyl phthalate	85-68-7	C <sub>19</sub> H <sub>20</sub> O <sub>4</sub>	312.35	-35 <sup>A</sup>	370 <sup>A</sup>	0.43 <sup>A</sup>
Butylated hydroxyanisole	25013-16-5	C <sub>11</sub> H <sub>16</sub> O <sub>2</sub>	180.25	51 <sup>A</sup>	268 <sup>A</sup>	2.32 <sup>A</sup>
Caffeine	58-08-2	C <sub>8</sub> H <sub>10</sub> N <sub>4</sub> O <sub>2</sub>	194.19	236.2 <sup>A</sup>	178 <sup>A</sup>	4.33 <sup>A</sup>
CA-ibuprofen	15935-54-3	C <sub>13</sub> H <sub>16</sub> O <sub>4</sub>	236.26	152.42 (est) <sup>C</sup>	395.40 (est) <sup>C</sup>	2.32 - 3.16 (est) <sup>C</sup>
Carbamazepine	298-46-4	C <sub>15</sub> H <sub>12</sub> N <sub>2</sub> O	236.27	190.2 <sup>A</sup>	410.02 (est) <sup>C</sup>	1.26 <sup>A</sup>
Carbamazepine-10,11-epoxid	36507-30-9	C <sub>15</sub> H <sub>12</sub> N <sub>2</sub> O	252.27	175.10 (est) <sup>C</sup>	419.72 (est) <sup>C</sup>	2.44 (est) <sup>C</sup>
Carbendazim	10605-21-7	C <sub>9</sub> H <sub>9</sub> N <sub>3</sub> O <sub>2</sub>	191.19	300 <sup>A</sup>	404.73 (est) <sup>C</sup>	0.9 <sup>A</sup>
Carbon tetrachloride	56-23-5	CCl <sub>4</sub>	153.81	-23 <sup>A</sup>	76.8 <sup>A</sup>	2.9 - 3.06 <sup>A</sup>
Carbozole	86-74-8	C <sub>12</sub> H <sub>9</sub> N	167.21	245 <sup>A</sup>	354.6 <sup>A</sup>	0.26 <sup>A</sup>
Cefazolin	25953-19-9	C <sub>14</sub> H <sub>14</sub> N <sub>8</sub> O <sub>4</sub> S <sub>3</sub>	454.51	198 - 200 <sup>A</sup>	757.72 (est) <sup>C</sup>	2.32 <sup>A</sup>
Cefriaxone	73384-59-5	C <sub>18</sub> H <sub>18</sub> N <sub>8</sub> O <sub>7</sub> S <sub>3</sub>	554.58	349.84 (est) <sup>C</sup>	915.45 (est) <sup>C</sup>	2.36 (est) <sup>C</sup>
Ceftazidime	72558-82-8	C <sub>22</sub> H <sub>22</sub> N <sub>6</sub> O <sub>7</sub> S <sub>2</sub>	546.57	349.84 (est) <sup>C</sup>	852.33 (est) <sup>C</sup>	2.60 <sup>C</sup>
Chlordane	57-74-9	C <sub>10</sub> H <sub>6</sub> Cl <sub>8</sub>	409.78	104 - 107 <sup>A</sup>	175 <sup>C</sup>	-1.25 <sup>A</sup>
Chlorfenvinphos	470-90-6	C <sub>12</sub> H <sub>14</sub> Cl <sub>3</sub> O <sub>4</sub> P	359.56	-23 - -19 <sup>A</sup>	167-170 <sup>C</sup>	2.09 <sup>A</sup>
Chloroacetic acid	79-11-8	C <sub>2</sub> H <sub>3</sub> ClO <sub>2</sub>	94.50	50 - 63 <sup>A</sup>	189.3 <sup>A</sup>	5.93 <sup>A</sup>
Chlorodifluoromethane	75-45-6	CHClF <sub>2</sub>	86.47	-157.42 <sup>A</sup>	-40.8 <sup>A</sup>	3.44 <sup>A</sup>
Chloromethane	74-87-3	CH <sub>3</sub> Cl	50.49	-97.6 <sup>A</sup>	-23.7 <sup>A</sup>	3.7 <sup>A</sup>
Chlorpyrifos	2921-88-2	C <sub>9</sub> H <sub>11</sub> Cl <sub>3</sub> NO <sub>3</sub> PS	350.59	41 - 42 <sup>A</sup>	160 (decomp) <sup>A</sup>	-0.39 - 0.14 <sup>A</sup>
Chlortetracycline	57-62-5	C <sub>22</sub> H <sub>23</sub> ClN <sub>2</sub> O <sub>8</sub>	478.88	168.5 <sup>C</sup>	764.02 (est) <sup>C</sup>	2.79 <sup>C</sup>

**Table A4. Select properties of organic contaminants potentially found in stormwater.**

Compound	CAS #	Chemical Formula	Molecular Weight	T <sub>m</sub> <sup>1</sup>	T <sub>b</sub> <sup>2</sup>	Log C <sub>w</sub> <sup>3</sup>
Units			g/mol	°C	°C	mg/L
Cholesterol	57-88-5	C <sub>27</sub> H <sub>46</sub> O	386.65	148.5 <sup>A</sup>	360 <sup>A</sup>	-1.02 <sup>A</sup>
Choline Chloride	67-48-1	C <sub>5</sub> H <sub>14</sub> NOCl	139.63	305 <sup>A</sup>	380.89 (est) <sup>C</sup>	6 (est) <sup>C</sup>
Chrysene	218-01-9	C <sub>18</sub> H <sub>12</sub>	228.29	255 <sup>A</sup>	448 <sup>A</sup>	-2.7 <sup>A</sup>
Cimetidine	51481-61-9	C <sub>10</sub> H <sub>16</sub> N <sub>6</sub> S	252.34	141 - 143 <sup>A</sup>	480.81 (est) <sup>C</sup>	4.02 <sup>C</sup>
Ciprofloxacin	85721-33-1	C <sub>17</sub> H <sub>18</sub> FN <sub>3</sub> O <sub>3</sub>	331.35	225 - 257 <sup>A</sup>	566.55 (est) <sup>C</sup>	4.48 <sup>A</sup>
Cis-androsterone	53-41-8	C <sub>19</sub> H <sub>30</sub> O <sub>2</sub>	290.45	178 <sup>C</sup>	386.13 (est) <sup>C</sup>	1.50 <sup>C</sup>
Citalopram	59729-33-8	C <sub>20</sub> H <sub>21</sub> FN <sub>2</sub> O	324.39	164.03 (est) <sup>C</sup>	178 <sup>C</sup>	1.49 (est) <sup>C</sup>
Clarithromycin	81103-11-9	C <sub>38</sub> H <sub>69</sub> NO <sub>13</sub>	747.95	217 - 225 <sup>A</sup>	842.47 (est) <sup>C</sup>	0.23 <sup>A</sup>
Clofibric Acid	882-09-7	C <sub>10</sub> H <sub>11</sub> ClO <sub>3</sub>	214.65	118 - 119 <sup>C</sup>	321.21 (est) <sup>C</sup>	2.77 <sup>C</sup>
Cocaine	50-36-2	C <sub>17</sub> H <sub>21</sub> NO <sub>4</sub>	303.36	98 <sup>A</sup>	362.63 (est) <sup>C</sup>	3.26 <sup>A</sup>
Codeine	76-57-3	C <sub>18</sub> H <sub>21</sub> NO <sub>3</sub>	299.36	157.5 <sup>A</sup>	250 <sup>A</sup>	4.08 <sup>C</sup>
Coronene	191-07-1	C <sub>24</sub> H <sub>12</sub>	300.36	437.3 <sup>C</sup>	525 <sup>C</sup>	-3.55 <sup>C</sup>
Cotinine	486-56-6	C <sub>10</sub> H <sub>12</sub> N <sub>2</sub> O	176.22	41 <sup>C</sup>	250 <sup>C</sup>	6 <sup>A</sup>
Cyanazine	21725-46-2	C <sub>9</sub> H <sub>13</sub> CIN <sub>6</sub>	240.69	167.5 - 169 <sup>A</sup>	369.47 (est) <sup>C</sup>	2.23 <sup>A</sup>
Cyanazine amide	36576-42-8	C <sub>9</sub> H <sub>15</sub> CIN <sub>6</sub> O	258.71	176.52 (est) <sup>C</sup>	422.76 (est) <sup>C</sup>	3.31 <sup>C</sup>
Cyclophosphamide	50-18-0	C <sub>7</sub> H <sub>15</sub> Cl <sub>2</sub> N <sub>2</sub> O <sub>2</sub> P	261.09	49.5 - 53 <sup>A</sup>	359.82 (est) <sup>C</sup>	4.6 <sup>A</sup>
Debromodiphenyl ether	1163-19-5	C <sub>12</sub> Br <sub>10</sub> O	959.17	305 <sup>A</sup>	425 <sup>A</sup>	-4 <sup>A</sup>
Deethylcyanazine	36556-77-1	C <sub>7</sub> H <sub>11</sub> CIN <sub>6</sub> O	230.66	175.43 (est) <sup>C</sup>	420.44 (est) <sup>C</sup>	3.04 <sup>C</sup>
Dehyronifedipine	67035-22-7	C <sub>17</sub> H <sub>16</sub> N <sub>2</sub> O <sub>6</sub>	344.32	178.45 (est) <sup>C</sup>	451.04 (est) <sup>C</sup>	0.75 (est) <sup>C</sup>
Deisopropylatrazine	1007-28-9	C <sub>5</sub> H <sub>8</sub> CIN <sub>5</sub>	173.60	112.32 (est) <sup>C</sup>	304.59 (est) <sup>C</sup>	2.83 <sup>C</sup>
Desethylatrazine	6190-65-4	C <sub>6</sub> H <sub>10</sub> CIN <sub>5</sub>	187.63	136 <sup>A</sup>	310.23 (est) <sup>C</sup>	3.51 <sup>A</sup>
Diatrizoate	117-96-4	C <sub>11</sub> H <sub>9</sub> I <sub>3</sub> N <sub>2</sub> O <sub>4</sub>	613.92	> 300 <sup>A</sup>	654.65 (est) <sup>C</sup>	5.7 <sup>A</sup>
Diazinon	333-41-5	C <sub>12</sub> H <sub>21</sub> N <sub>2</sub> O <sub>3</sub> PS	304.35	87.58 <sup>C</sup>	125 <sup>A</sup>	1.60 - 1.78 <sup>A</sup>
Dibenz(a,h)anthracene	53-70-3	C <sub>22</sub> H <sub>14</sub>	278.35	269 <sup>A</sup>	524 <sup>A</sup>	-2.6 <sup>A</sup>

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Compound	CAS #	Chemical Formula	Molecular Weight	T <sub>m</sub> <sup>1</sup>	T <sub>b</sub> <sup>2</sup>	Log C <sub>w</sub> <sup>3</sup>
Units			g/mol	°C	°C	mg/L
Dibenzofuran	132-64-9	C <sub>12</sub> H <sub>8</sub> O	168.19	86.5 <sup>A</sup>	287 <sup>A</sup>	0.49 <sup>A</sup>
Dibenzothiophene	132-65-0	C <sub>12</sub> H <sub>8</sub> S	184.26	99.5 <sup>A</sup>	332.5 <sup>A</sup>	0.17 <sup>A</sup>
Dibromoacetic acid	631-64-1	C <sub>2</sub> H <sub>2</sub> Br <sub>2</sub> O <sub>2</sub>	217.84	48 <sup>A</sup>	232 - 234 <sup>A</sup>	6.32 <sup>A</sup>
Dibromochloroacetic acid	5278-95-5	C <sub>2</sub> HBr <sub>2</sub> ClO <sub>2</sub>	252.29	68.10 (est) <sup>C</sup>	263.59 (est) <sup>C</sup>	3.38 <sup>A</sup>
Dibromochloromethane	124-48-1	CHBr <sub>2</sub> Cl	208.28	-20 <sup>A</sup>	121.3 - 125 <sup>A</sup>	3.43 <sup>A</sup>
Dibutyltin	1002-53-5	C <sub>8</sub> H <sub>18</sub> Sn	232.94	-7.44 (est) <sup>C</sup>	174.74 (est) <sup>C</sup>	3.20 <sup>C</sup>
Dichloroacetic acid	79-43-6	C <sub>2</sub> H <sub>2</sub> Cl <sub>2</sub> O <sub>2</sub>	128.94	-4 - 9.7 <sup>A</sup>	194 <sup>A</sup>	4.58 <sup>C</sup>
Dichlorodiphenyltrichloroethane (DDT)	50-29-3	C <sub>14</sub> H <sub>9</sub> Cl <sub>5</sub>	354.49	108.5 <sup>A</sup>	260 <sup>A</sup>	-2.26 <sup>A</sup>
Dichloromethane	75-09-2	CH <sub>2</sub> Cl <sub>2</sub>	84.93	-95 <sup>A</sup>	39.75 <sup>A</sup>	4.12 <sup>A</sup>
Dichloroprop	120-36-5	C <sub>9</sub> H <sub>6</sub> Cl <sub>2</sub> O <sub>3</sub>	235.06	117.5 <sup>A</sup>	334.17 (est) <sup>C</sup>	2.54 <sup>A</sup>
Diclofenac	15307-86-5	C <sub>14</sub> H <sub>11</sub> Cl <sub>2</sub> NO <sub>2</sub>	296.15	156 - 158 <sup>A</sup>	423.77 (est) <sup>C</sup>	0.37 <sup>A</sup>
Dieldrin	60-57-1	C <sub>12</sub> H <sub>8</sub> Cl <sub>6</sub> O	380.91	175.5 <sup>A</sup>	330 <sup>C</sup>	-0.71 <sup>A</sup>
Dienochlor	2227-17-0	C <sub>10</sub> Cl <sub>10</sub>	474.61	122 - 123 <sup>A</sup>	250 <sup>A</sup>	-1.6 <sup>A</sup>
Diethyl phthalate	84-66-2	C <sub>12</sub> H <sub>14</sub> O <sub>4</sub>	222.24	-40.5 <sup>A</sup>	295 <sup>A</sup>	3.03 <sup>A</sup>
Dihydrotestosterone	521-18-6	C <sub>19</sub> H <sub>30</sub> O <sub>2</sub>	290.44	181 <sup>C</sup>	386.13 (est) <sup>C</sup>	1.62 <sup>C</sup>
Diisobutyl phthalate	84-69-5	C <sub>16</sub> H <sub>22</sub> O <sub>4</sub>	278.34	-64 <sup>A</sup>	296.5 <sup>A</sup>	0.79 <sup>A</sup>
Diisodecyl phthalate	26761-40-0	C <sub>28</sub> H <sub>46</sub> O <sub>4</sub>	446.66	-50 <sup>A</sup>	463.36 (est) <sup>C</sup>	-0.55 <sup>A</sup>
Diisononyl phthalate	28553-12-0	C <sub>26</sub> H <sub>42</sub> O <sub>4</sub>	418.61	-48 <sup>A</sup>	440.16 (est) <sup>C</sup>	-0.7 <sup>A</sup>
Diltiazem	42399-41-7	C <sub>22</sub> H <sub>26</sub> N <sub>2</sub> O <sub>4</sub> S	414.52	212 <sup>A</sup>	540.45 (est) <sup>C</sup>	2.67 <sup>A</sup>
Dimethenamid	87674-68-8	C <sub>12</sub> H <sub>18</sub> ClNO <sub>2</sub> S	275.79	138.56 (est) <sup>C</sup>	127 <sup>C</sup>	2.39 <sup>C</sup>
Dimethyl phthalate	131-11-3	C <sub>10</sub> H <sub>10</sub> O <sub>4</sub>	194.18	5.5 <sup>A</sup>	283.7 <sup>A</sup>	3.6 <sup>A</sup>
Dimethyltin	23120-99-2	C <sub>2</sub> H <sub>6</sub> Sn	148.78	-82.83 (est) <sup>C</sup>	32.31 (est) <sup>C</sup>	4.78 (est) <sup>C</sup>
Dimetridazole	551-92-8	C <sub>5</sub> H <sub>7</sub> N <sub>3</sub> O <sub>2</sub>	141.13	138.5 <sup>C</sup>	296.07 (est) <sup>C</sup>	4.26 <sup>C</sup>
Di-n-butyl phthalate	84-74-2	C <sub>16</sub> H <sub>22</sub> O <sub>4</sub>	278.35	-35 <sup>A</sup>	340 <sup>A</sup>	1.05 <sup>A</sup>

**Table A4. Select properties of organic contaminants potentially found in stormwater.**

Compound	CAS #	Chemical Formula	Molecular Weight	T <sub>m</sub> <sup>1</sup>	T <sub>b</sub> <sup>2</sup>	Log C <sub>w</sub> <sup>3</sup>
Units			g/mol	°C	°C	mg/L
Di-n-octyl phthalate	117-84-0	C <sub>24</sub> H <sub>38</sub> O <sub>4</sub>	390.56	25 <sup>A</sup>	220 - 248 <sup>C</sup>	-1.66 <sup>A</sup>
Diocetyl tin	94410-05-6	C <sub>16</sub> H <sub>34</sub> Sn	345.15	78.31 (est) <sup>C</sup>	314.03 (est) <sup>C</sup>	-2.37 - -0.02 (est) <sup>C</sup>
Diphenhydramine	58-73-1	C <sub>17</sub> H <sub>21</sub> NO	255.36	168 <sup>C</sup>	150 - 165 <sup>C</sup>	3.49 <sup>A</sup>
Dipropyl phthalate	131-16-8	C <sub>14</sub> H <sub>18</sub> O <sub>4</sub>	250.29	18.19 (est) <sup>C</sup>	317.5 <sup>C</sup>	1.58 <sup>C</sup>
Diuron	330-54-1	C <sub>9</sub> H <sub>10</sub> Cl <sub>2</sub> N <sub>2</sub> O	233.09	158 - 159 <sup>A</sup>	180 - 190 (decomp) <sup>A</sup>	1.62 <sup>A</sup>
D-limonene	5989-27-5	C <sub>10</sub> H <sub>16</sub>	136.24	-74 <sup>A</sup>	177.6 <sup>A</sup>	1.14 <sup>A</sup>
Endrin	72-20-8	C <sub>12</sub> H <sub>8</sub> Cl <sub>6</sub> O	380.91	~200 <sup>A</sup>	330 <sup>C</sup>	-0.60 <sup>A</sup>
Epitestosterone	481-30-1	C <sub>19</sub> H <sub>28</sub> O <sub>2</sub>	288.43	155 <sup>C</sup>	390.02 (est) <sup>C</sup>	1.83 <sup>C</sup>
Erythromycin	114-07-8	C <sub>37</sub> H <sub>67</sub> NO <sub>13</sub>	733.94	191 <sup>A</sup>	853.10 (est) <sup>C</sup>	0.62 <sup>A</sup>
Estriol	50-27-1	C <sub>18</sub> H <sub>24</sub> O <sub>3</sub>	288.39	288 <sup>A</sup>	431.81 (est) <sup>C</sup>	1.44 <sup>A</sup>
Estrone	53-16-7	C <sub>18</sub> H <sub>22</sub> O <sub>2</sub>	270.37	260.2 <sup>A</sup>	154 <sup>C</sup>	1.48 <sup>A</sup>
Ethylbenzene	100-41-4	C <sub>8</sub> H <sub>10</sub>	106.17	-94.95 <sup>A</sup>	136.2 <sup>A</sup>	2.23 <sup>A</sup>
Ethylene thiourea	96-45-7	C <sub>3</sub> H <sub>6</sub> N <sub>2</sub> S	102.16	203 <sup>A</sup>	347.18 (est) <sup>C</sup>	4.3 <sup>A</sup>
Flumequine	42835-25-6	C <sub>14</sub> H <sub>12</sub> FO <sub>3</sub>	261.25	253 - 255 <sup>A</sup>	402.68 (est) <sup>C</sup>	3.34 <sup>A</sup>
Fluoranthene	206-44-0	C <sub>16</sub> H <sub>10</sub>	202.26	110.2 <sup>A</sup>	384 <sup>A</sup>	-0.6 <sup>A</sup>
Fluorene	86-73-7	C <sub>13</sub> H <sub>10</sub>	166.22	114.76 <sup>A</sup>	294 <sup>A</sup>	0.23 <sup>A</sup>
Fluoxetine	54910-89-3	C <sub>17</sub> H <sub>18</sub> F <sub>3</sub> NO	309.33	105.27 (est) <sup>C</sup>	347.23 (est) <sup>C</sup>	1.78 <sup>C</sup>
Formaldehyde	50-00-0	CH <sub>2</sub> O	30.03	-92 <sup>A</sup>	-19.5 <sup>A</sup>	5.6 <sup>A</sup>
Galaxolide	1222-05-5	C <sub>18</sub> H <sub>26</sub> O	258.41	-5 <sup>C</sup>	325 <sup>C</sup>	0.24 <sup>A</sup>
Gemfibrozil	25812-30-0	C <sub>15</sub> H <sub>22</sub> O <sub>3</sub>	250.34	62 <sup>A</sup>	159 <sup>A</sup>	1.04 <sup>A</sup>
Gestodene	60282-87-3	C <sub>21</sub> H <sub>26</sub> O <sub>2</sub>	310.43	197.9 <sup>A</sup>	413.40 (est) <sup>C</sup>	0.91 <sup>A</sup>
Glyphosate	1071-83-6	C <sub>3</sub> H <sub>8</sub> NO <sub>5</sub> P	169.07	230 <sup>A</sup>	417.49 (est) <sup>C</sup>	4.02 <sup>A</sup>
Hydroxyatrazine	67-68-5	C <sub>8</sub> H <sub>15</sub> N <sub>5</sub> O	197.24	164.13 (est) <sup>C</sup>	409.52 (est) <sup>C</sup>	6.00 (est) <sup>C</sup>
Hydroxydeethylatrazine	19988-24-0	C <sub>6</sub> H <sub>11</sub> N <sub>5</sub> O	169.19	173.57 (est) <sup>C</sup>	416.44 (est) <sup>C</sup>	5.42 <sup>C</sup>

**Table A4. Select properties of organic contaminants potentially found in stormwater.**

Compound	CAS #	Chemical Formula	Molecular Weight	T <sub>m</sub> <sup>1</sup>	T <sub>b</sub> <sup>2</sup>	Log C <sub>w</sub> <sup>3</sup>
Units			g/mol	°C	°C	mg/L
Hydroxydeisopropylatrazine	7313-54-4	C <sub>5</sub> H <sub>9</sub> N <sub>5</sub> O	155.16	171.42 (est) <sup>C</sup>	411.84 (est) <sup>C</sup>	5.84 <sup>C</sup>
Ibuprofen	15687-27-1	C <sub>13</sub> H <sub>18</sub> O <sub>2</sub>	206.29	75 - 77 <sup>A</sup>	323.11 (est) <sup>C</sup>	1.32 <sup>A</sup>
Ifosfamide	3778-73-2	C <sub>7</sub> H <sub>15</sub> Cl <sub>2</sub> N <sub>2</sub> O <sub>2</sub> P	261.10	39 - 41 <sup>A</sup>	353.49 (est) <sup>C</sup>	3.58 <sup>C</sup>
Indeno pyrene	193-39-5	C <sub>22</sub> H <sub>12</sub>	276.33	164 <sup>A</sup>	536 <sup>A</sup>	-3.16 <sup>A</sup>
Indole	120-72-9	C <sub>8</sub> H <sub>7</sub> N	117.15	52 <sup>A</sup>	253 <sup>A</sup>	3.55 <sup>A</sup>
Indomethacin	53-86-1	C <sub>19</sub> H <sub>16</sub> ClNO <sub>4</sub>	357.79	155 - 162 <sup>A</sup>	514.50 (est) <sup>C</sup>	-0.03 <sup>A</sup>
Iohexol	66108-95-0	C <sub>19</sub> H <sub>26</sub> I <sub>3</sub> N <sub>3</sub> O <sub>9</sub>	821.14	174 - 180 <sup>C</sup>	960.86 (est) <sup>C</sup>	2.03 <sup>C</sup>
Iomeprol	78649-41-9	C <sub>17</sub> H <sub>22</sub> I <sub>3</sub> N <sub>3</sub> O <sub>8</sub>	777.09	349.84 (est) <sup>C</sup>	904.94 (est) <sup>C</sup>	2.10 <sup>C</sup>
Iopamidol	60166-93-0	C <sub>17</sub> H <sub>22</sub> I <sub>3</sub> N <sub>3</sub> O <sub>8</sub>	777.08	300 <sup>A</sup>	942.21 (est) <sup>C</sup>	5.08 <sup>A</sup>
Iopromide	73334-07-3	C <sub>18</sub> H <sub>24</sub> I <sub>3</sub> N <sub>3</sub> O <sub>8</sub>	791.12	349.84 (est) <sup>C</sup>	885.14 (est) <sup>C</sup>	1.38 <sup>C</sup>
Isophorone	78-59-1	C <sub>9</sub> H <sub>14</sub> O	138.21	-8.1 <sup>A</sup>	215.32 <sup>A</sup>	4.08 <sup>A</sup>
Isopropyl chloride	75-29-6	C <sub>3</sub> H <sub>7</sub> Cl	78.54	-117.2 <sup>A</sup>	35.7 <sup>A</sup>	3.48 <sup>A</sup>
Isoproturon	34123-59-6	C <sub>12</sub> H <sub>18</sub> N <sub>2</sub> O	206.30	158 <sup>C</sup>	344.22 (est) <sup>C</sup>	2.16 <sup>C</sup>
Isoquinoline	119-65-3	C <sub>9</sub> H <sub>7</sub> N	129.16	26.47 <sup>C</sup>	243.2 <sup>C</sup>	3.19 <sup>C</sup>
Ketoprofen	22071-15-4	C <sub>16</sub> H <sub>14</sub> O <sub>3</sub>	254.29	94 <sup>C</sup>	403.57 (est) <sup>C</sup>	2.08 <sup>C</sup>
Lincomycin	154-21-2	C <sub>18</sub> H <sub>34</sub> N <sub>2</sub> O <sub>6</sub> S	406.54	135 - 148 <sup>A</sup>	606.28 (est) <sup>C</sup>	2.97 <sup>A</sup>
Lindane	58-89-9	C <sub>6</sub> H <sub>6</sub> Cl <sub>6</sub>	290.83	112.5 <sup>A</sup>	311 <sup>A</sup>	0.86 <sup>A</sup>
Lomefloxacin	98079-51-7	C <sub>17</sub> H <sub>19</sub> F <sub>2</sub> N <sub>3</sub> O <sub>3</sub>	351.36	239 - 240.5 <sup>C</sup>	558.54 (est) <sup>C</sup>	4.07 - 4.44 (est) <sup>C</sup>
Malathion	121-75-5	C <sub>10</sub> H <sub>19</sub> O <sub>6</sub> PS <sub>2</sub>	330.36	3.85 <sup>A</sup>	156 - 157 <sup>C</sup>	2.15 <sup>A</sup>
Mecoprop	93-65-2	C <sub>10</sub> H <sub>11</sub> ClO <sub>3</sub>	214.60	93 - 95 <sup>A</sup>	298 <sup>C</sup>	2.94 <sup>A</sup>
Metalaxyl	57837-19-1	C <sub>15</sub> H <sub>21</sub> NO <sub>4</sub>	279.33	71 - 72 <sup>A</sup>	295.9 <sup>A</sup>	3.92 <sup>A</sup>
Metaldehyde	108-62-3	C <sub>8</sub> H <sub>16</sub> O <sub>4</sub>	176.21	246.2 <sup>A</sup>	115 <sup>A</sup>	2.35 <sup>A</sup>
Metazachlor	67129-08-2	C <sub>14</sub> H <sub>16</sub> ClN <sub>3</sub> O	277.75	85 <sup>C</sup>	411.60 (est) <sup>C</sup>	2.40 <sup>C</sup>
Metformin	1115-70-4	C <sub>4</sub> H <sub>11</sub> N <sub>5</sub>	165.63	218 - 232 <sup>A</sup>	280.57 (est) <sup>C</sup>	6.03 <sup>A</sup>

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Compound	CAS #	Chemical Formula	Molecular Weight	T <sub>m</sub> <sup>1</sup>	T <sub>b</sub> <sup>2</sup>	Log C <sub>w</sub> <sup>3</sup>
Units			g/mol	°C	°C	mg/L
Methyl dihydrojasmonate	24851-98-7	C <sub>13</sub> H <sub>22</sub> O <sub>3</sub>	226.32	-10 <sup>C</sup>	309.32 (est) <sup>C</sup>	1.96 (est) <sup>C</sup>
Methyl salicylate	119-36-8	C <sub>8</sub> H <sub>8</sub> O <sub>3</sub>	152.15	-8.6 <sup>A</sup>	220 - 224 <sup>A</sup>	3.87 <sup>A</sup>
Methyl tert-butyl ether (MTBE)	1634-04-4	C <sub>5</sub> H <sub>12</sub> O	88.15	-108.6 <sup>A</sup>	55 <sup>A</sup>	4.71 <sup>A</sup>
Methylparaben	99-76-3	C <sub>8</sub> H <sub>8</sub> O <sub>3</sub>	152.15	125.2 <sup>A</sup>	270.5 <sup>A</sup>	3.4 <sup>A</sup>
Methyltriclosan	4640-01-1	C <sub>13</sub> H <sub>9</sub> Cl <sub>3</sub> O <sub>2</sub>	303.56	123.54 (est) <sup>C</sup>	362.44 (est) <sup>C</sup>	-0.39 (est) <sup>C</sup>
Metolachlor	51218-45-2	C <sub>15</sub> H <sub>22</sub> ClNO <sub>2</sub>	283.80	-62.1 <sup>A</sup>	282 <sup>C</sup>	2.72 <sup>A</sup>
Metoprolol	37350-58-6	C <sub>15</sub> H <sub>25</sub> NO <sub>3</sub>	267.37	116.15 (est) <sup>C</sup>	362.44 (est) <sup>C</sup>	6 <sup>A</sup>
Metronidazole	443-48-1	C <sub>6</sub> H <sub>9</sub> N <sub>3</sub> O <sub>3</sub>	171.15	158 - 160 <sup>A</sup>	357.30 (est) <sup>C</sup>	4.04 <sup>A</sup>
Miconazole	22916-47-8	C <sub>18</sub> H <sub>14</sub> Cl <sub>4</sub> N <sub>2</sub> O	416.13	215.55 (est) <sup>C</sup>	506.31 (est) <sup>C</sup>	-1.95 (est) <sup>C</sup>
Monobromoacetic acid	79-08-3	C <sub>2</sub> H <sub>3</sub> BrO <sub>2</sub>	138.95	49 <sup>A</sup>	208 <sup>A</sup>	6.24 <sup>A</sup>
Monobutyltin	78763-54-9	C <sub>4</sub> H <sub>10</sub> Sn	176.83	-47.79 (est) <sup>C</sup>	85.67 (est) <sup>C</sup>	1.93 <sup>C</sup>
Monochloroacetic acid	79-11-8	C <sub>2</sub> H <sub>3</sub> ClO <sub>2</sub>	94.49	50 - 63 <sup>A</sup>	189.3 <sup>A</sup>	5.93 <sup>A</sup>
Musk ambrette	83-66-9	C <sub>12</sub> H <sub>16</sub> N <sub>2</sub> O <sub>5</sub>	268.27	85 <sup>A</sup>	185 <sup>A</sup>	0.38 <sup>A</sup>
Musk ketone	81-14-1	C <sub>14</sub> H <sub>18</sub> N <sub>2</sub> O <sub>5</sub>	294.30	135.5 <sup>A</sup>	401.75 (est) <sup>C</sup>	-0.41 <sup>A</sup>
Musk xylene	81-15-2	C <sub>12</sub> H <sub>15</sub> N <sub>3</sub> O <sub>6</sub>	297.26	110 <sup>A</sup>	411.56 (est) <sup>C</sup>	-0.33 <sup>A</sup>
m-xylene	108-38-3	C <sub>8</sub> H <sub>10</sub>	106.17	-47.85 <sup>A</sup>	139.1 <sup>A</sup>	2.21 <sup>A</sup>
N,N-Diethyl-3-methylbenzamide (DEET)	134-62-3	C <sub>12</sub> H <sub>17</sub> NO	191.27	-45 <sup>C</sup>	290 <sup>C</sup>	2.96 <sup>A</sup>
N4-acetyl-Sulfamethazine	100-90-3	C <sub>14</sub> H <sub>16</sub> N <sub>4</sub> O <sub>3</sub> S	320.37	225.71 (est) <sup>C</sup>	528.06 (est) <sup>C</sup>	3.06 <sup>C</sup>
Nalidixic Acid	389-08-2	C <sub>12</sub> H <sub>12</sub> N <sub>2</sub> O <sub>3</sub>	232.24	229 - 230 <sup>A</sup>	397.21 (est) <sup>C</sup>	2 <sup>A</sup>
Naphthalene	91-20-3	C <sub>10</sub> H <sub>8</sub>	128.17	80.2 <sup>A</sup>	217.9 <sup>A</sup>	1.49 <sup>A</sup>
Naproxen	22204-53-1	C <sub>14</sub> H <sub>14</sub> O <sub>3</sub>	230.26	155 <sup>A</sup>	379.70 (est) <sup>C</sup>	1.2 <sup>A</sup>
Nitritotriacetic acid	139-13-9	C <sub>6</sub> H <sub>9</sub> NO <sub>6</sub>	191.14	242 <sup>A</sup>	429.28 (est) <sup>C</sup>	4.77 <sup>A</sup>
Nonylphenol	25154-52-3	C <sub>15</sub> H <sub>24</sub> O	220.35	42 <sup>C</sup>	293 - 297 <sup>A</sup>	0.7 <sup>A</sup>
Nonylphenol ethoxylates	9016-45-9	C <sub>19</sub> H <sub>32</sub> O <sub>3</sub>	308.46	140.16 (est) <sup>C</sup>	404.90 (est) <sup>C</sup>	0.02 (est) <sup>C</sup>

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Compound	CAS #	Chemical Formula	Molecular Weight	T <sub>m</sub> <sup>1</sup>	T <sub>b</sub> <sup>2</sup>	Log C <sub>w</sub> <sup>3</sup>
Units			g/mol	°C	°C	mg/L
Norfloxacin	70458-96-7	C <sub>16</sub> H <sub>18</sub> FN <sub>3</sub> O <sub>3</sub>	319.33	220 - 221 <sup>A</sup>	552.97 (est) <sup>C</sup>	2.45 <sup>A</sup>
N-propyl chloride	540-54-5	C <sub>3</sub> H <sub>7</sub> Cl	78.54	-122.8 <sup>A</sup>	46.6 <sup>A</sup>	3.43 <sup>A</sup>
Octylethoxylate	9002-93-1	C <sub>14</sub> H <sub>22</sub> O	250.38	-4 <sup>C</sup>	120 <sup>C</sup>	0.73 (est) <sup>C</sup>
Octylphenol	949-13-3	C <sub>13</sub> H <sub>22</sub> O	206.33	82.77 (est) <sup>C</sup>	310.93 (est) <sup>C</sup>	0.49 (est) <sup>C</sup>
Ofloxacin	82419-36-1	C <sub>18</sub> H <sub>20</sub> FN <sub>3</sub> O <sub>4</sub>	361.37	250 - 257 <sup>A</sup>	576.13 (est) <sup>C</sup>	4.03 <sup>A</sup>
OH-ibuprofen	51146-55-5	C <sub>13</sub> H <sub>18</sub> O <sub>3</sub>	222.29	122.61 (est) <sup>C</sup>	354.25 (est) <sup>C</sup>	3.47 <sup>C</sup>
op' DDA	34113-46-7	C <sub>14</sub> H <sub>10</sub> Cl <sub>2</sub> O <sub>2</sub>	281.13	145.16 (est) <sup>C</sup>	395.09 (est) <sup>C</sup>	0.53 - 0.88 (est) <sup>C</sup>
Oxazepam	604-75-1	C <sub>15</sub> H <sub>11</sub> ClN <sub>2</sub> O <sub>2</sub>	286.72	205 - 206 <sup>A</sup>	491.71 (est) <sup>C</sup>	1.3 <sup>A</sup>
o-xylene	95-47-6	C <sub>8</sub> H <sub>10</sub>	106.17	-25.16 <sup>A</sup>	144.5 <sup>A</sup>	2.25 <sup>A</sup>
Oxytetracycline	79-57-2	C <sub>22</sub> H <sub>24</sub> N <sub>2</sub> O <sub>9</sub>	460.43	184.5 <sup>A</sup>	781.66 (est) <sup>C</sup>	2.5 <sup>A</sup>
Para-cresol	106-44-5	C <sub>7</sub> H <sub>8</sub> O	108.13	34.77 <sup>A</sup>	201.9 <sup>A</sup>	4.33 <sup>A</sup>
Para-tert-octylphenol	140-66-9	C <sub>14</sub> H <sub>22</sub> O	206.36	84 - 85 <sup>A</sup>	158 <sup>C</sup>	0.68 <sup>C</sup>
Pentachlorophenol	87-86-5	C <sub>6</sub> HCl <sub>5</sub> O	266.32	174 <sup>A</sup>	309 - 310 <sup>A</sup>	1.15 <sup>A</sup>
Pentoxifylline	6493-05-6	C <sub>13</sub> H <sub>18</sub> N <sub>4</sub> O <sub>3</sub>	278.31	217.99 (est) <sup>C</sup>	511.53 (est) <sup>C</sup>	2.66 <sup>C</sup>
Perfluorobutanesulfonic Acid (PFBS)	375-73-5	C <sub>4</sub> HF <sub>9</sub> O <sub>3</sub> S	300.10	36.86 (est) <sup>C</sup>	210 - 212 <sup>A</sup>	2.54 <sup>A</sup>
Perfluorooctanionic Acid (PFOA)	335-67-1	C <sub>8</sub> HF <sub>15</sub> O <sub>2</sub>	414.07	54.3 <sup>A</sup>	192 <sup>A</sup>	3.52 <sup>A</sup>
Perfluorodecanoic Acid (PFDA)	335-76-2	C <sub>10</sub> HF <sub>19</sub> O <sub>2</sub>	514.09	50.26 (est) <sup>C</sup>	239.28 (est) <sup>C</sup>	-2.09 (est) <sup>C</sup>
Perfluorododecanoic Acid (PFDoA)	307-55-1	C <sub>12</sub> HF <sub>23</sub> O <sub>2</sub>	614.10	108 <sup>C</sup>	249 <sup>C</sup>	-3.89 (est) <sup>C</sup>
Perfluoroheptanoic Acid (PFHpA)	375-85-9	C <sub>7</sub> HF <sub>13</sub> O <sub>2</sub>	364.06	30 <sup>A</sup>	175 <sup>A</sup>	0.56 <sup>A</sup>
Perfluorohexanesulfonic Acid (PFHxS)	355-46-4	C <sub>6</sub> HF <sub>13</sub> O <sub>3</sub> S	400.11	41.25 (est) <sup>C</sup>	238 - 239 <sup>A</sup>	0.79 <sup>A</sup>
Perfluorohexanoic Acid (PFHxA)	307-24-4	C <sub>6</sub> HF <sub>11</sub> O <sub>2</sub>	314.05	23.07 (est) <sup>C</sup>	157 <sup>A</sup>	4.2 <sup>A</sup>
Perfluorononanoic Acid (PFNA)	375-95-1	C <sub>9</sub> HF <sub>17</sub> O <sub>2</sub>	464.08	38.94 (est) <sup>C</sup>	221.92 (est) <sup>C</sup>	-1.20 <sup>A</sup>
Perfluorooctanesulfonic Acid (PFOS)	1763-23-1	C <sub>8</sub> HF <sub>17</sub> O <sub>3</sub> S	500.13	51.90 (est) <sup>C</sup>	249 <sup>A</sup>	-2.49 <sup>A</sup>

**Table A4. Select properties of organic contaminants potentially found in stormwater.**

Compound	CAS #	Chemical Formula	Molecular Weight	T <sub>m</sub> <sup>1</sup>	T <sub>b</sub> <sup>2</sup>	Log C <sub>w</sub> <sup>3</sup>
Units			g/mol	°C	°C	mg/L
Perfluoroundecanoic Acid (PFUnA)	4234-23-5	C <sub>11</sub> HF <sub>21</sub> O <sub>2</sub>	564.09	349.84 (est) <sup>C</sup>	917.72 (est) <sup>C</sup>	-1.72 (est) <sup>C</sup>
Perylene	198-55-0	C <sub>20</sub> H <sub>12</sub>	2252.32	273 - 274 <sup>A</sup>	350 - 400 <sup>A</sup>	-3.4 <sup>A</sup>
Phenanthrene	85-01-8	C <sub>14</sub> H <sub>10</sub>	178.23	99 <sup>A</sup>	338.4 <sup>A</sup>	0.04 <sup>A</sup>
Phenol	108-95-2	C <sub>6</sub> H <sub>5</sub> OH	94.11	40.91 <sup>A</sup>	181.75 <sup>A</sup>	4.92 <sup>A</sup>
Pipemidic Acid	51940-44-4	C <sub>14</sub> H <sub>17</sub> N <sub>5</sub> O <sub>3</sub>	303.32	253 - 255 <sup>C</sup>	566.52 (est) <sup>C</sup>	6 <sup>C</sup>
pp' DDA	83-05-6	C <sub>14</sub> H <sub>10</sub> Cl <sub>2</sub> O <sub>2</sub>	281.13	167 - 169 <sup>C</sup>	395.09 (est) <sup>C</sup>	0.53 - 0.76 (est) <sup>C</sup>
Primidone	125-33-7	C <sub>12</sub> H <sub>14</sub> N <sub>2</sub> O <sub>2</sub>	218.26	281 - 282 <sup>A</sup>	463.54 (est) <sup>C</sup>	2.7 <sup>A</sup>
Progesterone	57-83-0	C <sub>21</sub> H <sub>30</sub> O <sub>2</sub>	314.46	129 <sup>A</sup>	396.19 (est) <sup>C</sup>	0.94 <sup>A</sup>
Propazine	139-40-2	C <sub>9</sub> H <sub>16</sub> ClN <sub>5</sub>	229.71	229.7 <sup>A</sup>	318.46 (est) <sup>C</sup>	0.93 <sup>A</sup>
Propranolol	525-66-6	C <sub>16</sub> H <sub>21</sub> NO <sub>2</sub>	259.34	96 <sup>C</sup>	386.48 (est) <sup>C</sup>	2.36 <sup>C</sup>
Propylparaben	94-13-3	C <sub>10</sub> H <sub>12</sub> O <sub>3</sub>	180.20	96.1 <sup>A</sup>	285.14 (est) <sup>C</sup>	2.7 <sup>A</sup>
Propyphenazone	479-92-5	C <sub>14</sub> H <sub>18</sub> N <sub>2</sub> O	230.31	103 <sup>C</sup>	364.43 (est) <sup>C</sup>	2.82 <sup>C</sup>
p-xylene	106-42-3	C <sub>8</sub> H <sub>10</sub>	106.17	13.3 <sup>A</sup>	138.3 <sup>A</sup>	2.21 <sup>A</sup>
Pyrene	129-00-0	C <sub>16</sub> H <sub>10</sub>	202.25	150.62 <sup>A</sup>	394 <sup>A</sup>	-0.87 <sup>A</sup>
Roxithromycin	80214-83-1	C <sub>41</sub> H <sub>75</sub> N <sub>2</sub> O <sub>15</sub>	837.05	349.84 (est) <sup>C</sup>	917.72 (est) <sup>C</sup>	-1.72 (est) <sup>C</sup>
Salicylic acid	69-72-7	C <sub>7</sub> H <sub>6</sub> O <sub>3</sub>	138.12	159 <sup>A</sup>	211 <sup>A</sup>	3.35 <sup>A</sup>
Sertraline	79617-96-2	C <sub>17</sub> H <sub>17</sub> Cl <sub>2</sub> N	306.23	243 - 245 <sup>A</sup>	387.42 (est) <sup>C</sup>	3.58 <sup>A</sup>
Simazine	122-34-9	C <sub>7</sub> H <sub>12</sub> ClN <sub>5</sub>	201.66	225 - 227 <sup>A</sup>	307.45 (est) <sup>C</sup>	0.79 <sup>A</sup>
Skatole	83-34-1	C <sub>9</sub> H <sub>9</sub> N	131.18	95 <sup>A</sup>	265 - 266 <sup>A</sup>	2.7 <sup>A</sup>
Sodium decylbenzenesulfonate	1322-98-1	C <sub>16</sub> H <sub>25</sub> NaO <sub>3</sub> S	321.44	276.79 (est) <sup>C</sup>	637.41 (est) <sup>C</sup>	2.25 (est) <sup>C</sup>
Sodium dodecylbenzenesulfonate	25155-30-0	C <sub>18</sub> H <sub>29</sub> NaO <sub>3</sub> S	348.48	287.63 (est) <sup>C</sup>	660.62 (est) <sup>C</sup>	2.90 <sup>A</sup>
Sodium N-tridecylbenzenesulfonate	26248-24-8	C <sub>19</sub> H <sub>31</sub> NaO <sub>3</sub> S	362.50	293.05 (est) <sup>C</sup>	672.22 (est) <sup>C</sup>	0.74 <sup>C</sup>
Sodium Undecylbenzenesulfonate	27636-75-5	C <sub>17</sub> H <sub>27</sub> NaO <sub>3</sub> S	334.45	282.21 (est) <sup>C</sup>	649.01 (est) <sup>C</sup>	1.74 (est) <sup>C</sup>
Sotalol	3930-20-9	C <sub>12</sub> H <sub>20</sub> N <sub>2</sub>	272.36	206.5 - 207 <sup>C</sup>	407.42 (est) <sup>C</sup>	2.74 (est) <sup>C</sup>

**Table A4. Select properties of organic contaminants potentially found in stormwater.**

Compound	CAS #	Chemical Formula	Molecular Weight	T <sub>m</sub> <sup>1</sup>	T <sub>b</sub> <sup>2</sup>	Log C <sub>w</sub> <sup>3</sup>
Units			g/mol	°C	°C	mg/L
Sucralose	56038-13-2	C <sub>12</sub> H <sub>19</sub> Cl <sub>3</sub> O <sub>8</sub>	397.64	130 <sup>A</sup>	551.89 (est) <sup>C</sup>	4.36 <sup>A</sup>
Sulfadiazine	68-35-9	C <sub>10</sub> H <sub>10</sub> N <sub>4</sub> O <sub>2</sub> S	250.78	255.5 <sup>C</sup>	428 (est) <sup>C</sup>	1.89 <sup>C</sup>
Sulfadimethoxine	122-11-2	C <sub>12</sub> H <sub>14</sub> N <sub>4</sub> O <sub>4</sub> S	310.33	203.5 <sup>C</sup>	475.30 (est) <sup>C</sup>	2.54 <sup>C</sup>
Sulfamethazine	57-68-1	C <sub>12</sub> H <sub>14</sub> N <sub>4</sub> O <sub>2</sub> S	278.33	176 <sup>A</sup>	451.19 (est) <sup>C</sup>	3.18 <sup>A</sup>
Sulfamethoxazole	723-46-6	C <sub>10</sub> H <sub>11</sub> N <sub>3</sub> O <sub>3</sub> S	253.28	167 <sup>A</sup>	414.01 (est) <sup>C</sup>	2.79 <sup>A</sup>
Sulfamonomethoxine	1220-83-3	C <sub>11</sub> H <sub>12</sub> N <sub>4</sub> O <sub>3</sub> S	280.30	190.01 (est) <sup>C</sup>	451.65 (est) <sup>C</sup>	3.60 - 3.88 (est) <sup>C</sup>
Sulfanilamide	63-74-1	C <sub>6</sub> H <sub>8</sub> N <sub>2</sub> O <sub>2</sub> S	172.20	165.5 <sup>A</sup>	341.95 (est) <sup>C</sup>	3.92 <sup>A</sup>
Sulfathiazole	72-14-0	C <sub>9</sub> H <sub>9</sub> N <sub>3</sub> O <sub>2</sub> S <sub>2</sub>	255.32	175 (form a); 202 (form b) <sup>A</sup>	428.28 (est) <sup>C</sup>	2.57 <sup>A</sup>
Terbutylazine	5915-41-3	C <sub>9</sub> H <sub>16</sub> ClN <sub>5</sub>	229.71	175.5 <sup>A</sup>	321.23 (est) <sup>C</sup>	0.95 <sup>A</sup>
Terbutryne	886-50-0	C <sub>10</sub> H <sub>19</sub> N <sub>5</sub> S	241.36	104 <sup>A</sup>	154 - 160 <sup>A</sup>	1.4 <sup>A</sup>
Tert-Amyl chloride	594-36-5	C <sub>5</sub> H <sub>11</sub> Cl	106.59	-73.5 <sup>C</sup>	85.5 <sup>C</sup>	2.90 <sup>C</sup>
Tertrabutyltin	1461-25-2	C <sub>16</sub> H <sub>36</sub> Sn	347.17	-97 <sup>A</sup>	145 <sup>A</sup>	-4.19 - -1.94 (est) <sup>C</sup>
Testosterone	58-22-0	C <sub>19</sub> H <sub>28</sub> O <sub>2</sub>	288.42	151 <sup>A</sup>	390.02 (est) <sup>C</sup>	1.37 <sup>A</sup>
Tetrachloroethylene	127-18-4	C <sub>2</sub> Cl <sub>4</sub>	165.83	-22.3 <sup>A</sup>	121.3 <sup>A</sup>	2.31 <sup>A</sup>
Tetracycline	60-54-8	C <sub>22</sub> H <sub>24</sub> N <sub>2</sub> O <sub>8</sub>	444.44	170 - 175 <sup>A</sup>	745.32 (est) <sup>C</sup>	2.36 <sup>A</sup>
Thiabendazole	148-79-8	C <sub>10</sub> H <sub>7</sub> N <sub>3</sub> S	201.25	30 4- 305 <sup>A</sup>	443.05 (est) <sup>C</sup>	1.7 <sup>A</sup>
Toluene	108-88-3	C <sub>7</sub> H <sub>8</sub>	92.14	-94.9 <sup>A</sup>	110.6 <sup>A</sup>	2.72 <sup>A</sup>
Tolyfluanid	731-27-1	C <sub>10</sub> H <sub>13</sub> Cl <sub>2</sub> FN <sub>2</sub> O <sub>2</sub> S <sub>2</sub>	347.24	93 <sup>A</sup>	Decomp <sup>A</sup>	-0.05 <sup>A</sup>
Tonalide	21145-77-7	C <sub>18</sub> H <sub>26</sub> O	258.41	54.5 <sup>A</sup>	331.88 (est) <sup>C</sup>	0.1 <sup>A</sup>
Tramadol	27203-92-5	C <sub>16</sub> H <sub>25</sub> NO <sub>2</sub>	263.38	115.18 (est) <sup>C</sup>	355.82 (est) <sup>C</sup>	3.06 <sup>A</sup>
Tribromoacetic acid	75-96-7	C <sub>2</sub> HBr <sub>3</sub> O <sub>2</sub>	296.74	129 - 135 <sup>A</sup>	245 <sup>A</sup>	5.3 <sup>A</sup>
Tribromomethane	75-25-2	CBr <sub>3</sub>	251.72	8.69 <sup>A</sup>	149.1 <sup>A</sup>	3.49 <sup>A</sup>
Tributyl phosphate	126-73-8	C <sub>12</sub> H <sub>27</sub> O <sub>4</sub> P	266.32	<-80 <sup>A</sup>	289 <sup>A</sup>	2.45 <sup>A</sup>
Tributyltin	688-73-3	C <sub>12</sub> H <sub>28</sub> Sn	291.07	28.89 (est) <sup>C</sup>	80 <sup>C</sup>	0.64 <sup>C</sup>

**Table A4. Select properties of organic contaminants potentially found in stormwater.**

Compound	CAS #	Chemical Formula	Molecular Weight	T <sub>m</sub> <sup>1</sup>	T <sub>b</sub> <sup>2</sup>	Log C <sub>w</sub> <sup>3</sup>
Units			g/mol	°C	°C	mg/L
Trichloroacetic acid	76-03-9	C <sub>2</sub> HCl <sub>3</sub> O <sub>2</sub>	163.38	57.5 <sup>A</sup>	195.5 <sup>A</sup>	4.73 <sup>A</sup>
Trichloroethylene	79-01-6	C <sub>2</sub> HCl <sub>3</sub>	131.40	-84.7 <sup>A</sup>	87.2 <sup>A</sup>	3.11 <sup>A</sup>
Trichloromethane	67-66-3	CHCl <sub>3</sub>	119.38	-63.41 <sup>A</sup>	61.17 <sup>A</sup>	3.9 <sup>A</sup>
Triclosan	3380-34-5	C <sub>12</sub> H <sub>7</sub> Cl <sub>3</sub> O <sub>2</sub>	289.54	54 - 57.3 <sup>A</sup>	280 - 290 <sup>A</sup>	1 <sup>A</sup>
Tricyclohexyltin chloride	3091-32-5	C <sub>18</sub> H <sub>33</sub> ClSn	403.62	129.5 <sup>C</sup>	370.70 (est) <sup>C</sup>	-2.59 - -0.98 (est) <sup>C</sup>
Tricyclohexyltin Hydroxide	13121-70-5	C <sub>18</sub> H <sub>35</sub> OSn	385.17	195 <sup>A</sup>	390.72 (est) <sup>C</sup>	-2.92 - 0.69 (est) <sup>C</sup>
Triethylphosphate	78-40-0	C <sub>6</sub> H <sub>15</sub> O <sub>4</sub> P	182.15	-56.4 <sup>A</sup>	215.5 <sup>A</sup>	5.7 <sup>A</sup>
Trifluoromethane	75-46-7	CF <sub>3</sub>	69.01	-155.18 <sup>A</sup>	-82 <sup>A</sup>	3.61 <sup>A</sup>
Triiodomethane	75-47-8	CHI <sub>3</sub>	393.73	119 <sup>A</sup>	218 <sup>A</sup>	2 <sup>A</sup>
Triisobutyl phosphate	126-71-6	C <sub>12</sub> H <sub>27</sub> O <sub>4</sub> P	266.32	16.43 (est) <sup>C</sup>	264 <sup>C</sup>	1.21 - 2.68 (est) <sup>C</sup>
Trimethoprim	738-70-5	C <sub>14</sub> H <sub>18</sub> N <sub>4</sub> O <sub>3</sub>	290.32	199 - 203 <sup>C</sup>	449.23 (est) <sup>C</sup>	2.6 <sup>A</sup>
Trimethyltin Chloride	1066-45-1	C <sub>3</sub> H <sub>9</sub> SnCl	199.27	37.5 <sup>A</sup>	154 - 156 <sup>A</sup>	4.30 - 4.40 (est) <sup>C</sup>
Tri-n-Butyltin Hydride	688-73-3	C <sub>12</sub> H <sub>28</sub> Sn	291.06	28.89 (est) <sup>C</sup>	112.5 - 113.5 <sup>A</sup>	0.64 <sup>C</sup>
Trioctyltin chloride	2587-76-0	C <sub>24</sub> H <sub>51</sub> ClSn	493.82	424 (est) <sup>C</sup>	154.33 (est) <sup>C</sup>	-6.15 - -6.31 (est) <sup>C</sup>
Triphenyl phosphate	115-86-6	C <sub>18</sub> H <sub>15</sub> O <sub>4</sub> P	326.29	50.5 <sup>A</sup>	245 <sup>A</sup>	0.28 <sup>A</sup>
Triphenylphosphine oxide	791-28-6	C <sub>18</sub> H <sub>15</sub> OP	278.29	156.5 <sup>C</sup>	>360 <sup>C</sup>	1.80 - 2.31 (est) <sup>C</sup>
Triphenyltin Acetate	900-95-8	C <sub>20</sub> H <sub>18</sub> O <sub>2</sub> Sn	409.07	122 - 123 <sup>A</sup>	406.83 (est) <sup>C</sup>	0.95 <sup>A</sup>
Triphenyltin chloride	639-58-7	C <sub>18</sub> H <sub>15</sub> ClSn	385.47	103.5 <sup>A</sup>	240 <sup>A</sup>	1.60 <sup>A</sup>
Triphenyltin hydride	892-20-6	C <sub>18</sub> H <sub>16</sub> Sn	351.03	94.01 (est) <sup>C</sup>	358.52 (est) <sup>C</sup>	-0.86 - 0.079 (est) <sup>C</sup>
Triphenyltin hydroxide	76-87-9	C <sub>18</sub> H <sub>16</sub> Sn	367.03	119 <sup>A</sup>	409.44 (est) <sup>C</sup>	0.79 <sup>A</sup>
Tris(1,3-dichloroisopropyl)phosphate	13674-87-8	C <sub>9</sub> H <sub>15</sub> Cl <sub>6</sub> O <sub>4</sub> P	430.89	27 <sup>C</sup>	236 - 237 <sup>A</sup>	0.85 <sup>A</sup>
Tris(2-butoxyethyl)phosphate	78-51-3	C <sub>18</sub> H <sub>39</sub> O <sub>7</sub> P	395.48	-70 <sup>A</sup>	215 - 228 <sup>A</sup>	3.04 <sup>A</sup>
Tris(2-chloroethyl)phosphate	115-96-8	C <sub>6</sub> H <sub>12</sub> Cl <sub>3</sub> O <sub>4</sub> P	285.48	-55 <sup>A</sup>	330 <sup>A</sup>	3.89 <sup>A</sup>

**Table A4. Select properties of organic contaminants potentially found in stormwater.**

Compound	CAS #	Chemical Formula	Molecular Weight	T <sub>m</sub> <sup>1</sup>	T <sub>b</sub> <sup>2</sup>	Log C <sub>w</sub> <sup>3</sup>
Units			g/mol	°C	°C	mg/L
Tylosin	1401-69-0	C <sub>46</sub> H <sub>77</sub> NO <sub>17</sub>	916.10	128 - 132 <sup>A</sup>	-47.4 <sup>A</sup>	3.7 <sup>A</sup>
Valsartan	137862-53-4	C <sub>24</sub> H <sub>29</sub> N <sub>5</sub> O <sub>3</sub>	435.50	116 - 117 <sup>A</sup>	674.38 (est) <sup>C</sup>	0.147 <sup>A</sup>
Venlafaxine	93413-69-5	C <sub>17</sub> H <sub>27</sub> NO <sub>2</sub>	277.40	118.80 (est) <sup>C</sup>	363.80 (est) <sup>C</sup>	2.43 <sup>A</sup>
Xylene	1330-20-7	C <sub>8</sub> H <sub>10</sub>	106.17	-47.4 <sup>A</sup>	137.2 - 160 <sup>A</sup>	2.03 <sup>A</sup>
α-BHC	319-84-6	C <sub>6</sub> H <sub>6</sub> Cl <sub>6</sub>	296.77	157.4 <sup>A</sup>	287.78 <sup>A</sup>	0.3 <sup>A</sup>
α-Endosulfan	959-98-8	C <sub>9</sub> H <sub>6</sub> Cl <sub>6</sub> O <sub>3</sub> S	406.90	106 <sup>C</sup>	401.28 (est) <sup>C</sup>	0.17 <sup>C</sup>
α-Hexachloro-cyclohexane	319-84-6	C <sub>6</sub> H <sub>6</sub> Cl <sub>6</sub>	290.83	157.4 <sup>A</sup>	287.78 <sup>A</sup>	0.3 <sup>A</sup>

<sup>1</sup>Melting Point

<sup>2</sup>Boiling Point

<sup>3</sup>C<sub>w</sub> is the solubility in water

<sup>4</sup>K<sub>H</sub> is the Henry's Law constant

<sup>5</sup>K<sub>OC</sub> is the organic carbon – water partitioning coefficient

<sup>6</sup>K<sub>ow</sub> is the octanol – water partitioning coefficient

<sup>7</sup>P<sup>0</sup> is the vapor pressure

<sup>8</sup>pKa is the acid dissociation constant

<sup>A</sup>Toxnet: <https://toxnet.nlm.nih.gov/newtoxnet/hsdb.htm> - Only values that were peer reviewed were used unless denoted by est for estimated value

<sup>C</sup>EPI Suite version 4.1: U.S. EPA. Office of Pollution Prevention and Toxics and Syracuse Research Corporation. Copyright 2000 – 2012. Peer reviewed values were used when available, otherwise the values obtained are delineated using est

Decomp= decomposes

est= estimated value

**Table A4. Select properties of organic contaminants potentially found in stormwater.**

Compound	CAS #	Log K <sub>H</sub> <sup>4</sup>	Log K <sub>OC</sub> <sup>5</sup>	Log K <sub>OW</sub> <sup>6</sup>	Log P <sup>0,7</sup>	pKa <sup>8</sup>
Units		L m <sup>3</sup> /mol	L/kg	L/kg	atm	
1,1,1,2-tetrachloroethane	630-20-6	-2.6 <sup>A</sup>	2.6 <sup>A</sup>	2.93 <sup>A</sup>	-1.80 <sup>A</sup>	-----
1,1,1-trichloroethane	71-55-6	-1.79 <sup>A</sup>	1.82 - 2.18 <sup>A</sup>	2.49 <sup>A</sup>	-0.79 <sup>A</sup>	-----
1,1,2,2-tetrachloroethane	79-34-5	-3.38 <sup>A</sup>	1.9 - 2.37 <sup>A</sup>	2.39 <sup>A</sup>	-2.12 <sup>A</sup>	-----
1,1,2-trichloroethane	79-00-5	-3.08 <sup>A</sup>	1.92 - 2.32 <sup>A</sup>	1.89 <sup>A</sup>	-1.52 <sup>A</sup>	-----
1,1-dichloroethane	75-34-3	-2.25 <sup>A</sup>	0.96 - 1.48 <sup>A</sup>	1.79 <sup>A</sup>	-0.53 <sup>A</sup>	-----
1,2,3-trichloropropane	96-18-4	-3.46 <sup>A</sup>	1.89 - 1.98 <sup>A</sup>	2.27 <sup>A</sup>	-2.32 <sup>A</sup>	-----
1,2,3-trimethylbenzene	526-73-8	-2.36 <sup>A</sup>	2.8 - 3.04 <sup>A</sup>	3.66 <sup>A</sup>	-2.65 <sup>A</sup>	-----
1,2,4-trichlorobenzene	120-82-1	-2.85 <sup>A</sup>	3.1 - 4.03 <sup>A</sup>	4.02 <sup>A</sup>	-3.22 <sup>A</sup>	-----
1,2,4-trimethylbenzene	95-63-6	-2.21 <sup>A</sup>	2.73 <sup>A</sup>	3.78 <sup>A</sup>	-2.56 <sup>A</sup>	-----
1,2-dichlorobenzene	95-50-1	-2.82 <sup>A</sup>	2.45 - 4.3 <sup>A</sup>	3.43 <sup>A</sup>	-2.74 <sup>A</sup>	-----
1,2-dichloroethane	107-06-2	-2.93 <sup>A</sup>	1.52 <sup>A</sup>	1.48 <sup>A</sup>	-0.99 <sup>A</sup>	-----
1,2-dichloropropane	78-87-5	-2.55 <sup>A</sup>	1.67 <sup>A</sup>	1.98 <sup>A</sup>	-1.16 <sup>A</sup>	-----
1,3-dichlorobenzene	541-73-1	-2.55 <sup>A</sup>	2.47 - 4.7 <sup>A</sup>	3.53 <sup>A</sup>	-2.55 <sup>A</sup>	-----
1,3-dichloropropane	142-28-9	-3.01 <sup>A</sup>	2.46 <sup>A</sup>	2 <sup>A</sup>	-1.62 <sup>A</sup>	-----
1,4-dichlorobenzene	106-46-7	-2.57 <sup>A</sup>	2.44 - 4.8 <sup>A</sup>	3.44 <sup>A</sup>	-2.64 <sup>A</sup>	-----
11-keto testosterone	53187-98-7	-14.87 (est) <sup>C</sup>	1.97 - 2.42 (est) <sup>C</sup>	1.92 (est) <sup>C</sup>	-11.83 (est) <sup>C</sup>	-----
17 $\alpha$ -estradiol	57-91-0	-14.85 - -13.44 (est) <sup>C</sup>	2.90 - 4.19 (est) <sup>C</sup>	4.01 <sup>C</sup>	-11.58 (est) <sup>C</sup>	-----
17 $\alpha$ -ethynyl estradiol	57-63-6	-14.10 (est) <sup>C</sup>	2.71 - 4.65 (est) <sup>C</sup>	3.67 <sup>C</sup>	-11.59 (est) <sup>C</sup>	-----
17 $\beta$ -estradiol	50-28-2	-14.85 - -13.44 (est) <sup>C</sup>	2.90 - 4.19 (est) <sup>C</sup>	4.01 <sup>C</sup>	-11.58 (est) <sup>C</sup>	-----
19-norethisterone	68-22-4	-9.24 <sup>A</sup>	2.34 <sup>A</sup>	2.97 <sup>A</sup>	-9.39 <sup>A</sup>	-----
1-acetyl-1-methyl-2-dimethyl-oxamoyl-2-phenylhydrazide	519-65-3	-16.93 (est) <sup>C</sup>	-0.05 - 1.00 (est) <sup>C</sup>	-0.76 (est) <sup>C</sup>	-11.42 (est) <sup>C</sup>	-----
1-chloro-2-methylpropane	513-36-0	-4.71 <sup>C</sup>	1.78 - 2.16 (est) <sup>C</sup>	2.49 (est) <sup>C</sup>	-0.71 <sup>C</sup>	-----
1-chlorobutane	109-69-3	-1.78 <sup>A</sup>	1.97 - 2.01 <sup>A</sup>	2.39 <sup>A</sup>	-0.88 <sup>A</sup>	-----

**Table A4. Select properties of organic contaminants potentially found in stormwater.**

Compound	CAS #	Log K <sub>H</sub> <sup>4</sup>	Log K <sub>oc</sub> <sup>5</sup>	Log K <sub>ow</sub> <sup>6</sup>	Log P <sup>0,7</sup>	pKa <sup>8</sup>
Units		L m <sup>3</sup> /mol	L/kg	L/kg	atm	
1-chloropentane	543-59-9	-1.55 <sup>A</sup>	2.38 <sup>A</sup>	2.73 <sup>A</sup>	-1.39 <sup>A</sup>	-----
1H-benzotrazole	95-14-7	-6.5 <sup>A</sup>	2.16 <sup>A</sup>	1.44 <sup>A</sup>	-4.28 <sup>A</sup>	8.37 <sup>A</sup>
1-methylnaphthalene	90-12-0	-3.29 <sup>A</sup>	2.76 - 5.78 <sup>A</sup>	3.87 <sup>A</sup>	-4.06 <sup>A</sup>	-----
2, 4-D	94-75-7	-7.01 <sup>A</sup>	1.30 - 2.13 <sup>A</sup>	2.81 <sup>A</sup>	-9.71 <sup>A</sup>	2.73 <sup>A</sup>
2,2',3,4,4',5,5'-heptachlorobiphenyl (PCB 180)	35065-29-3	-8 <sup>C</sup>	5.54 - 5.64 (est) <sup>C</sup>	8.27 (est) <sup>C</sup>	-8.89 <sup>C</sup>	-----
2,2',3,4,4',5-hexachlorobiphenyl (PCB 138)	35065-28-2	-7.68 <sup>C</sup>	5.19 - 5.33 (est) <sup>C</sup>	7.44 <sup>C</sup>	-8.30 <sup>C</sup>	-----
2,2',4,4',5,5'-hexachlorobiphenyl (PCB 153)	35065-27-1	-7.64 <sup>C</sup>	5.32 - 5.36 (est) <sup>C</sup>	6.34 <sup>C</sup>	-8.35 <sup>C</sup>	-----
2,2',5,5'-tetrachlorobiphenyl (PCB 52)	35693-99-3	-6.70 <sup>C</sup>	4.43 - 4.89 (est) <sup>C</sup>	6.09 <sup>C</sup>	-7.96 (est) <sup>C</sup>	-----
2,3',4,4',5-pentachlorobiphenyl (PCB 118)	31508-00-6	-6.54 <sup>C</sup>	5.01 - 5.11 (est) <sup>C</sup>	7.12 <sup>C</sup>	-7.93 <sup>C</sup>	-----
2,4,4'-trichlorobiphenyl (PCB 28)	7012-37-5	-6.70 <sup>C</sup>	4.18 - 4.68 (est) <sup>C</sup>	5.62 <sup>C</sup>	-6.28 <sup>C</sup>	-----
2,4,5,2',5'-pentachlorobiphenyl (PCB 101)	37680-73-2	-7.05 <sup>C</sup>	4.83 - 5.11 (est) <sup>C</sup>	5.68 <sup>C</sup>	-7.48 <sup>C</sup>	-----
2-chloro-2-methylpropane	507-20-0	-4.72 (est) <sup>C</sup>	1.64 - 2.13 (est) <sup>C</sup>	2.45 (est) <sup>C</sup>	-0.40 <sup>C</sup>	-----
2-chlorobutane	78-86-4	-4.62 <sup>C</sup>	1.78 - 2.02 (est) <sup>C</sup>	2.33 <sup>C</sup>	-0.69 <sup>C</sup>	-----
2-Chloromethylphenol	40053-98-3	-10.31 - -9.66 (est) <sup>C</sup>	2.37 - 2.77 (est) <sup>C</sup>	2.31 (est) <sup>C</sup>	-4.83 (est) <sup>C</sup>	-----
2-methyl-4-chlorophenoxyacetic Acid	94-74-6	-9.32 <sup>A</sup>	1.7 - 1.79 <sup>A</sup>	3.25 <sup>A</sup>	-8.11 <sup>A</sup>	3.13 <sup>A</sup>
2-methylnaphthalene	91-57-6	-3.29 <sup>A</sup>	3.00 - 5.96 <sup>A</sup>	3.86 <sup>A</sup>	-4.14 <sup>A</sup>	-----
2-methylthiobenzothiazole	615-22-5	-10.96 (est) <sup>C</sup>	2.73 - 3.41 (est) <sup>C</sup>	3.15 <sup>C</sup>	-6.00 (est) <sup>C</sup>	-----
3-Chloromethylphenol	60760-06-7	-10.31 - -9.66 (est) <sup>C</sup>	2.37 - 2.77 (est) <sup>C</sup>	2.31 (est) <sup>C</sup>	-4.83 (est) <sup>C</sup>	-----
3-β-coprostanol	360-68-9	-7.38 - -6.79 (est) <sup>C</sup>	5.39 - 6.19 (est) <sup>C</sup>	8.82 (est) <sup>C</sup>	-12.14 (est) <sup>C</sup>	-----
4-Chloromethylphenol	35421-08-0	-10.31 - -9.66	2.37 - 2.76 (est) <sup>C</sup>	2.31 (est) <sup>C</sup>	-4.83 (est) <sup>C</sup>	-----

**Table A4. Select properties of organic contaminants potentially found in stormwater.**

Compound	CAS #	Log K <sub>H</sub> <sup>4</sup>	Log K <sub>oc</sub> <sup>5</sup>	Log K <sub>ow</sub> <sup>6</sup>	Log P <sup>0,7</sup>	pKa <sup>8</sup>
Units		L m <sup>3</sup> /mol (est) <sup>C</sup>	L/kg	L/kg	atm	
4'-hydroxy-diclofenac	64118-84-9	-18.31 (est) <sup>C</sup>	2.33 - 2.77 (est) <sup>C</sup>	3.70 <sup>C</sup>	-11.97 (est) <sup>C</sup>	-----
4-methyl-1H-benzotriazole	29878-31-7	-9.79 (est) <sup>C</sup>	1.94 (est) <sup>C</sup>	1.71 (est) <sup>C</sup>	-3.01 (est) <sup>C</sup>	-----
4-Nitrophenol	100-02-7	-7.89 <sup>A</sup>	1.2 - 2.7 <sup>A</sup>	1.91 <sup>A</sup>	-6.89 <sup>A</sup>	7.15 <sup>A</sup>
4-nonylphenol	104-40-5	-4.47 <sup>A</sup>	4.51 <sup>A</sup>	5.76 <sup>A</sup>	-5.97 <sup>A</sup>	-----
4-nonylphenoldiethoxylate	20427-84-3	-11.58 (est) <sup>C</sup>	3.39 - 3.41 (est) <sup>C</sup>	5.30 (est) <sup>C</sup>	-5.16 (est) <sup>C</sup>	-----
4-nonylphenolmonoethoxylate	104-35-8	-9.78 - -8.90 (est) <sup>C</sup>	3.48 - 3.66 (est) <sup>C</sup>	5.58 (est) <sup>C</sup>	-9.63 (est) <sup>C</sup>	-----
4-tert-Butylphenol	98-54-4	-8.92 <sup>C</sup>	2.92 - 3.11 (est) <sup>C</sup>	3.31 <sup>C</sup>	-4.30 <sup>C</sup>	-----
5-methyl-1H-benzotriazole	136-85-6	-9.79 (est) <sup>C</sup>	1.93 - 1.94 (est) <sup>C</sup>	1.71 (est) <sup>C</sup>	-6.39 (est) <sup>C</sup>	-----
Acebutolol	37517-30-9	-22.52 (est) <sup>C</sup>	1.40 - 1.64 (est) <sup>C</sup>	1.71 <sup>C</sup>	-13.67 (est) <sup>C</sup>	-----
Acenaphthene	83-32-9	-3.74 <sup>A</sup>	2.97, 3.59, 5.33, 5.87 <sup>A</sup>	3.92 <sup>A</sup>	-5.54 <sup>A</sup>	-----
Acenaphthylene	208-96-8	-3.9 <sup>A</sup>	3.75 - 6.21 <sup>A</sup>	3.93 <sup>A</sup>	-5.20 <sup>A</sup>	-----
Acesulfame	33665-90-6	-8.02 <sup>A</sup>	0.6 <sup>A</sup>	-1.33 <sup>A</sup>	-7.93 <sup>A</sup>	5.67 <sup>A</sup>
Acetaldehyde	75-07-0	-4.18 <sup>A</sup>	0 <sup>A</sup>	-0.34 <sup>A</sup>	0.07 <sup>A</sup>	13.57 <sup>A</sup>
Acetaminophen	103-90-2	-9.06 <sup>A</sup>	1.32 <sup>A</sup>	0.46 <sup>A</sup>	-7.08 <sup>A</sup>	9.38 <sup>A</sup>
Acetophenone	98-86-2	-4.98 <sup>A</sup>	1 - 2.43 <sup>A</sup>	1.58 <sup>A</sup>	-3.28 <sup>A</sup>	-----
Acrylamide	79-06-1	-8.77 <sup>A</sup>	1.7 <sup>A</sup>	-0.67 <sup>A</sup>	-5.05 <sup>A</sup>	-----
Alachlor	15972-60-8	-8.08 <sup>A</sup>	2.08 - 3.33 <sup>A</sup>	3.52 <sup>A</sup>	-7.54 <sup>A</sup>	-----
Aldrin	309-00-2	-4.36 <sup>A</sup>	2.60 - 4.45 <sup>A</sup>	6.50 <sup>A</sup>	-6.80 <sup>A</sup>	-----
Alkylphenol ethoxylate	68412-54-4	3.39 - 3.41 (est) <sup>C</sup>	-11.59 (est) <sup>C</sup>	5.30 (est) <sup>C</sup>	-13.92 (est) <sup>C</sup>	-----
Aminotriazole	61-82-5	-12.66 <sup>A</sup>	1.06 - 1.71 <sup>A</sup>	-0.97 <sup>A</sup>	-9.24 <sup>A</sup>	4.2, 10.7 <sup>A</sup>
Amitriptyline	50-48-6	-7.16 <sup>A</sup>	3.55 <sup>A</sup>	4.92 <sup>A</sup>	-9.32 <sup>A</sup>	9.76 <sup>A</sup>
Amoxicillin	26787-78-0	-20.6 <sup>A</sup>	2 <sup>A</sup>	0.87 <sup>A</sup>	-19.21 <sup>A</sup>	3.2, 11.7 <sup>A</sup>
AMPA	74341-63-2	-18.30 (est) <sup>C</sup>	-1.82 - 1.00 (est) <sup>C</sup>	-3.52 (est) <sup>C</sup>	-11.93 (est) <sup>C</sup>	-----

**Table A4. Select properties of organic contaminants potentially found in stormwater.**

Compound	CAS #	Log K <sub>H</sub> <sup>4</sup>	Log K <sub>OC</sub> <sup>5</sup>	Log K <sub>OW</sub> <sup>6</sup>	Log P <sup>0,7</sup>	pKa <sup>8</sup>
Units		L m <sup>3</sup> /mol	L/kg	L/kg	atm	
Androstenedione	63-05-8	-7.43 <sup>A</sup>	3.79 <sup>A</sup>	2.75 <sup>A</sup>	-11.02 <sup>A</sup>	-----
Anthracene	120-12-7	-1.64 <sup>A</sup>	3.41 - 3.93 <sup>A</sup>	4.45 <sup>A</sup>	-8.07 <sup>A</sup>	-----
Anthraquinone	84-65-1	-7.63 <sup>A</sup>	3.44 - 4.24 <sup>A</sup>	3.39 <sup>A</sup>	-9.82 <sup>A</sup>	-----
Atenolol	29122-68-7	-20.86 (est) <sup>C</sup>	0.61 - 1.83 (est) <sup>C</sup>	0.16 <sup>A</sup>	-12.00 (est) <sup>C</sup>	9.6 <sup>A</sup>
Atrazine	1912-24-9	-8.59 <sup>A</sup>	1.73 - 3.07 <sup>A</sup>	2.61 <sup>A</sup>	-9.42 <sup>A</sup>	1.60 <sup>A</sup>
Atrazine amide-I	142179-76-8	-10.09 - -11.42 (est) <sup>C</sup>	1.46 - 1.54 (est) <sup>C</sup>	1.32 (est) <sup>C</sup>	-8.99 (est) <sup>C</sup>	-----
Azithromycin	83905-01-5	-28.28 <sup>A</sup>	3.49 <sup>A</sup>	4.02 <sup>A</sup>	-26.47 <sup>A</sup>	8.74 <sup>A</sup>
Bentazone	25057-89-0	-8.66 <sup>A</sup>	1.12 - 2.25 <sup>A</sup>	2.80 <sup>A</sup>	-8.34 <sup>A</sup>	3.30 <sup>A</sup>
Benz(a)anthracene	56-55-3	-2.24 <sup>A</sup>	5.74 - 6.27 <sup>A</sup>	5.76 <sup>A</sup>	-9.56 <sup>A</sup>	-----
Benzene	71-43-2	-2.25 <sup>A</sup>	1.93 <sup>A</sup>	2.13 <sup>A</sup>	-0.91 <sup>A</sup>	-----
Benzo(a)fluoranthene	203-33-8	-9.18 <sup>C</sup>	5.02, 5.78 (est) <sup>C</sup>	5.78 <sup>C</sup>	-10.49 <sup>C</sup>	-----
Benzo(a)pyrene	50-32-8	-2.92 <sup>A</sup>	6.0 - 8.17 <sup>A</sup>	6.13 <sup>A</sup>	-11.14 <sup>A</sup>	-----
Benzo(b)fluoranthene	205-99-2	-6.18 <sup>A</sup>	6.60 - 7.96 <sup>A</sup>	5.78, 6.6 <sup>A</sup>	-9.18 <sup>A</sup>	-----
Benzo(e)pyrene	192-97-2	-9.52 <sup>C</sup>	5.59 - 5.78 (est) <sup>C</sup>	6.44 <sup>C</sup>	11.12 <sup>C</sup>	-----
Benzo(j)fluoranthene	205-82-3	-9.58 - -9.09 (est) <sup>C</sup>	5.30 - 5.78 (est) <sup>C</sup>	6.11 (est) <sup>C</sup>	-10.46 (est) <sup>C</sup>	-----
Benzo(k)fluoranthene	207-08-9	-6.23 <sup>A</sup>	5.61 - 8.44 <sup>A</sup>	6.11 <sup>A</sup>	-11.9 <sup>A</sup>	-----
Benzo(b)naphtho(2,3-d)thiophene	243-46-9	-8.57 (est) <sup>C</sup>	4.63 - 4.99 (est) <sup>C</sup>	5.34 (est) <sup>C</sup>	-9.34 (est) <sup>C</sup>	-----
Benzo(ghi)perylene	191-24-2	-6.48 <sup>A</sup>	4.61 - 7.08 <sup>A</sup>	6.63 <sup>A</sup>	-12.88 <sup>A</sup>	-----
Benzophenone	119-61-9	-5.72 <sup>A</sup>	2.63 - 2.71 <sup>A</sup>	3.18 <sup>A</sup>	-5.6 <sup>A</sup>	-----
Benzothiazole	95-16-9	-6.43 <sup>A</sup>	2.47 <sup>A</sup>	2.01 <sup>A</sup>	-4.74 <sup>A</sup>	-----
Benzyl acetate	140-11-4	-4.96 <sup>A</sup>	2.45 <sup>A</sup>	1.96 <sup>A</sup>	-3.63 <sup>A</sup>	-----
Beta-sitosterol	83-46-5	-6.53 (est) <sup>C</sup>	5.85 - 6.69 (est) <sup>C</sup>	9.65 (est) <sup>C</sup>	-12.31 (est) <sup>C</sup>	-----
Bezafibrate	41859-67-0	-17.67 (est) <sup>C</sup>	2.31 - 2.62 (est) <sup>C</sup>	4.25 (est) <sup>C</sup>	-13.10 (est) <sup>C</sup>	-----
Bis(2-chloroethyl) ether	111-44-4	-4.54 <sup>A</sup>	2.08 <sup>A</sup>	1.29 <sup>A</sup>	-2.69 <sup>A</sup>	-----

**Table A4. Select properties of organic contaminants potentially found in stormwater.**

Compound	CAS #	Log $K_H^4$	Log $K_{oc}^5$	Log $K_{ow}^6$	Log $P^{0,7}$	pKa <sup>8</sup>
Units		L m <sup>3</sup> /mol	L/kg	L/kg	atm	
Bis(2-chloroisopropyl) ether	39638-32-9	-6.48 (est) <sup>C</sup>	1.66 - 2.90 (est) <sup>C</sup>	3.73 (est) <sup>C</sup>	-3.03 - -2.95 <sup>A</sup>	-----
Bis(2-ethylhexyl) phthalate	117-81-7	-6.57 <sup>A</sup>	4.94 - 6.23 <sup>A</sup>	7.6 <sup>A</sup>	-9.73 <sup>A</sup>	-----
Bis(tributyltin) oxide	56-35-9	-6.89 <sup>A</sup>	3.33 - 7.41 (est) <sup>C</sup>	3.84 <sup>A</sup>	-7.99 <sup>C</sup>	-----
Bisphenol A	80-05-7	-10.4 <sup>A</sup>	2.06 - 3.59 <sup>A</sup>	3.32 <sup>A</sup>	-10.28 <sup>A</sup>	9.6 <sup>A</sup>
Bromochloroacetic acid	5589-96-8	-7.66 <sup>A</sup>	0.28 <sup>A</sup>	0.61 <sup>A</sup>	-3.74 <sup>A</sup>	1.40 <sup>A</sup>
Bromodichloroacetic acid	71133-14-7	-8.1 <sup>A</sup>	0.51 - 1.00 (est) <sup>C</sup>	1.53 <sup>A</sup>	-4.33 <sup>A</sup>	0.03 <sup>A</sup>
Bromodichloromethane	75-27-4	-2.67 <sup>A</sup>	1.72 - 2.4 <sup>A</sup>	2 <sup>A</sup>	-1.18 <sup>A</sup>	-----
Butyl benzyl phthalate	85-68-7	-5.89 <sup>A</sup>	3.21 - 4.00 <sup>A</sup>	4.73 <sup>A</sup>	-7.97 <sup>A</sup>	-----
Butylated hydroxyanisole	25013-16-5	-5.92 <sup>A</sup>	2.92 <sup>A</sup>	3.5 <sup>A</sup>	-5.51 <sup>A</sup>	-----
Caffeine	58-08-2	-10.96 <sup>A</sup>	1.85 - 3.89 <sup>A</sup>	-0.07 <sup>A</sup>	-8.93 <sup>A</sup>	0.7, 14.0 <sup>A</sup>
CA-ibuprofen	15935-54-3	-15.07 (est) <sup>C</sup>	1.25 - 3.10 (est) <sup>C</sup>	1.97 (est) <sup>C</sup>	-9.15 (est) <sup>C</sup>	-----
Carbamazepine	298-46-4	-9.96 <sup>A</sup>	2.71 <sup>A</sup>	2.45 <sup>A</sup>	-9.63 <sup>A</sup>	13.9 <sup>A</sup>
Carbamazepine-10,11-epoxid	36507-30-9	-15.16 (est) <sup>C</sup>	1.31 - 2.51 (est) <sup>C</sup>	0.95 (est) <sup>C</sup>	-10.00 (est) <sup>C</sup>	-----
Carbendazim	10605-21-7	-11.82 <sup>A</sup>	2.09 - 3.45 <sup>A</sup>	1.52 <sup>A</sup>	-12.01 <sup>A</sup>	4.2, 4.29 <sup>A</sup>
Carbon tetrachloride	56-23-5	-1.56 <sup>A</sup>	1.85 <sup>A</sup>	2.83 <sup>A</sup>	-0.82 <sup>A</sup>	-----
Carbazole	86-74-8	-6.94 <sup>A</sup>	2.06 - 3.7 <sup>A</sup>	3.72 <sup>A</sup>	-8.70 <sup>A</sup>	-6 <sup>A</sup>
Cefazolin	25953-19-9	-22.7 <sup>A</sup>	1.08 <sup>A</sup>	-0.58 <sup>A</sup>	-20.71 <sup>A</sup>	3.6 <sup>A</sup>
Ceftriaxone	73384-59-5	-33.07 (est) <sup>C</sup>	-1.15 - 2.94 (est) <sup>C</sup>	-1.37 (est) <sup>C</sup>	-29.75 (est) <sup>C</sup>	-----
Ceftazidime	72558-82-8	-36.61 (est) <sup>C</sup>	-0.65 - 4.57 (est) <sup>C</sup>	-1.60 <sup>C</sup>	-23.52 (est) <sup>C</sup>	-----
Chlordane	57-74-9	-4.31	4.30 - 4.88 <sup>A</sup>	6.16 <sup>A</sup>	-7.89 <sup>A</sup>	-----
Chlorfenvinphos	470-90-6	-7.54 <sup>A</sup>	2.47 <sup>A</sup>	3.81 <sup>A</sup>	-8.01 <sup>A</sup>	-----
Chloroacetic acid	79-11-8	-8.03 <sup>A</sup>	1.49 <sup>A</sup>	0.22 <sup>A</sup>	-4.07 <sup>A</sup>	2.87 <sup>A</sup>
Chlorodifluoromethane	75-45-6	-1.39 <sup>A</sup>	0.93 <sup>A</sup>	1.08 <sup>A</sup>	0.98 <sup>A</sup>	-----
Chloromethane	74-87-3	-2.05 <sup>A</sup>	1.11 <sup>A</sup>	0.91 <sup>A</sup>	0.75 <sup>A</sup>	-----
Chlorpyrifos	2921-88-2	-4.45 <sup>A</sup>	2.99 - 4.49 <sup>A</sup>	4.96 <sup>A</sup>	-7.58 <sup>A</sup>	-----
Chlortetracycline	57-62-5	-26.46 (est) <sup>C</sup>	0.25 - 1.86 (est) <sup>C</sup>	-0.62 <sup>C</sup>	-24.12 (est) <sup>C</sup>	-----

**Table A4. Select properties of organic contaminants potentially found in stormwater.**

Compound	CAS #	Log K <sub>H</sub> <sup>4</sup>	Log K <sub>OC</sub> <sup>5</sup>	Log K <sub>OW</sub> <sup>6</sup>	Log P <sup>0,7</sup>	pKa <sup>8</sup>
Units		L m <sup>3</sup> /mol	L/kg	L/kg	atm	
Cholesterol	57-88-5	-3.78 <sup>A</sup>	4.2 <sup>A</sup>	8.74 (est) <sup>C</sup>	-11.99 <sup>A</sup>	-----
Choline Chloride	67-48-1	-18.69 (est) <sup>C</sup>	-2.43 - 0.16 (est) <sup>C</sup>	-5.16 <sup>A</sup>	-12.19 (est) <sup>C</sup>	-----
Chrysene	218-01-9	-6.03 <sup>A</sup>	5.12 - 7.79 <sup>A</sup>	5.73, 5.81 <sup>A</sup>	-11.09 <sup>A</sup>	-----
Cimetidine	51481-61-9	-15.02 <sup>A</sup>	1.59 <sup>A</sup>	0.4 <sup>A</sup>	-11.14 <sup>A</sup>	6.8 <sup>A</sup>
Ciprofloxacin	85721-33-1	-16.97 <sup>C</sup>	-0.004 - 1.00 (est) <sup>C</sup>	0.28 <sup>A</sup>	-15.44 <sup>A</sup>	6.09, 8.74 <sup>A</sup>
Cis-androsterone	53-41-8	-11.20 (est) <sup>C</sup>	2.75 - 3.34 (est) <sup>C</sup>	3.69 <sup>C</sup>	-10.80 (est) <sup>C</sup>	-----
Citalopram	59729-33-8	-13.57 (est) <sup>C</sup>	3.23 - 4.44 (est) <sup>C</sup>	3.74 (est) <sup>C</sup>	-9.83 (est) <sup>C</sup>	-----
Clarithromycin	81103-11-9	-31.76 (est) <sup>C</sup>	2.18 <sup>A</sup>	3.16 <sup>A</sup>	-27.52 <sup>A</sup>	8.99 <sup>A</sup>
Clofibric Acid	882-09-7	-10.66 (est) <sup>C</sup>	1.63 - 1.64 (est) <sup>C</sup>	2.57 <sup>C</sup>	-7.01 (est) <sup>C</sup>	-----
Cocaine	50-36-2	-10.38 <sup>A</sup>	2.78 <sup>A</sup>	2.3 <sup>A</sup>	-9.60 <sup>A</sup>	8.61 <sup>A</sup>
Codeine	76-57-3	-13.12 <sup>A</sup>	2.4 - 2.85 <sup>A</sup>	1.19 <sup>A</sup>	-11.14 <sup>A</sup>	8.21, 10.6 <sup>A</sup>
Coronene	191-07-1	-11.57 - -10.67 (est) <sup>C</sup>	6.63 - 6.80 (est) <sup>C</sup>	7.64 <sup>C</sup>	-14.55 <sup>C</sup>	-----
Cotinine	486-56-6	-11.48 <sup>A</sup>	2.11 <sup>A</sup>	0.07 <sup>A</sup>	-6.30 <sup>A</sup>	-----
Cyanazine	21725-46-2	-9.59 <sup>A</sup>	2.26 - 2.57 <sup>A</sup>	2.22 <sup>A</sup>	-9.74 <sup>A</sup>	0.87 <sup>A</sup>
Cyanazine amide	36576-42-8	-16.87 (est) <sup>C</sup>	1.37 - 1.97 (est) <sup>C</sup>	1.19 (est) <sup>C</sup>	-10.09 (est) <sup>C</sup>	-----
Cyclophosphamide	50-18-0	-10.85 <sup>A</sup>	1.72 <sup>A</sup>	0.63 <sup>A</sup>	-7.23 <sup>A</sup>	-----
Debromodiphenyl ether	1163-19-5	-7.92 <sup>A</sup>	5.45 <sup>A</sup>	9.97 <sup>A</sup>	-13.04 <sup>A</sup>	-----
Deethylcyanazine	36556-77-1	-17.33 (est) <sup>C</sup>	0.82 - 1.64 (est) <sup>C</sup>	0.15 (est) <sup>C</sup>	-10.02 (est) <sup>C</sup>	-----
Dehyronifedipine	67035-22-7	-16.06 - -15.97 (est) <sup>C</sup>	2.93 - 3.38 (est) <sup>C</sup>	3.15 (est) <sup>C</sup>	-10.81 (est) <sup>C</sup>	-----
Deisopropylatrazine	1007-28-9	-8.10 - -8.93 (est) <sup>C</sup>	1.37 - 1.84 (est) <sup>C</sup>	1.15 <sup>C</sup>	-6.56 (est) <sup>C</sup>	-----
Desethylatrazine	6190-65-4	-8.82 <sup>A</sup>	1.58 - 3.48 <sup>A</sup>	1.51 <sup>A</sup>	-6.91 <sup>A</sup>	-----
Diatrizoate	117-96-4	-17.74 <sup>A</sup>	1 <sup>A</sup>	1.37 <sup>A</sup>	-17.33 <sup>A</sup>	1.13, 7.95 <sup>A</sup>
Diazinon	333-41-5	-7.20 <sup>A</sup>	1.60 - 3.16 <sup>A</sup>	3.81 <sup>A</sup>	-6.93 <sup>A</sup>	2.6 <sup>A</sup>

**Table A4. Select properties of organic contaminants potentially found in stormwater.**

Compound	CAS #	Log K <sub>H</sub> <sup>4</sup>	Log K <sub>OC</sub> <sup>5</sup>	Log K <sub>OW</sub> <sup>6</sup>	Log P <sup>0,7</sup>	pKa <sup>8</sup>
Units		L m <sup>3</sup> /mol	L/kg	L/kg	atm	
Dibenz(a,h)anthracene	53-70-3	-6.85 <sup>A</sup>	5.76 - 7.68 <sup>A</sup>	6.5 <sup>A</sup>	-11.90 <sup>A</sup>	-----
Dibenzofuran	132-64-9	-3.68 <sup>A</sup>	3.62 <sup>A</sup>	4.12 <sup>A</sup>	-5.49 <sup>A</sup>	-----
Dibenzothiophene	132-65-0	-4.47 <sup>A</sup>	3.72 - 4.31 <sup>A</sup>	4.38 <sup>A</sup>	-6.57 <sup>A</sup>	-----
Dibromoacetic acid	631-64-1	-8.35 <sup>A</sup>	0.18 <sup>A</sup>	0.7 <sup>A</sup>	-4.52 <sup>A</sup>	1.48 <sup>A</sup>
Dibromochloroacetic acid	5278-95-5	-8.59 <sup>A</sup>	0.51 - 1.05 (est) <sup>C</sup>	1.62 <sup>A</sup>	-5.17 <sup>A</sup>	0.03 <sup>A</sup>
Dibromochloromethane	124-48-1	-3.11 <sup>A</sup>	1.92 <sup>A</sup>	2.16 <sup>A</sup>	-2.14 <sup>A</sup>	-----
Dibutyltin	1002-53-5	-3.42 (est) <sup>C</sup>	1.29 - 2.90 (est) <sup>C</sup>	1.49 <sup>C</sup>	-2.66 (est) <sup>C</sup>	-----
Dichloroacetic acid	79-43-6	-8.08 <sup>A</sup>	1.88 <sup>A</sup>	0.92 <sup>A</sup>	-3.63 <sup>A</sup>	1.26 <sup>A</sup>
Dichlorodiphenyltrichloroethane (DDT)	50-29-3	-5.08 <sup>A</sup>	5.05 - 5.54 <sup>A</sup>	6.91 <sup>A</sup>	-9.68 <sup>A</sup>	-----
Dichloromethane	75-09-2	0.41 <sup>A</sup>	0.9 - 1.68 <sup>A</sup>	1.25 <sup>A</sup>	-0.24 <sup>A</sup>	-----
Dichloroprop	120-36-5	-10.06 <sup>A</sup>	1.53 - 2.11 <sup>A</sup>	3.43 <sup>A</sup>	-10.01 <sup>A</sup>	3.1 <sup>A</sup>
Diclofenac	15307-86-5	-11.33 <sup>A</sup>	2.39 <sup>A</sup>	4.51 <sup>A</sup>	-10.1 <sup>A</sup>	4.15 <sup>A</sup>
Dieldrin	60-57-1	-1.95 <sup>A</sup>	3.29 - 4.37 <sup>A</sup>	5.40 <sup>A</sup>	-8.11 <sup>A</sup>	-----
Dienochlor	2227-17-0	-4.26 <sup>A</sup>	5.30 - 7.28 (est) <sup>C</sup>	3.23 <sup>A</sup>	-8.54 <sup>A</sup>	-----
Diethyl phthalate	84-66-2	-6.21 <sup>A</sup>	1.84 - 3.24 <sup>A</sup>	2.47 <sup>A</sup>	-5.56 <sup>A</sup>	-----
Dihydrotestosterone	521-18-6	-13.00 - -10.20 (est) <sup>C</sup>	2.67 - 3.34 (est) <sup>C</sup>	3.55 <sup>C</sup>	-10.95 (est) <sup>C</sup>	-----
Diisobutyl phthalate	84-69-5	-11.03 <sup>A</sup>	3.14, 5.9 <sup>A</sup>	4.11 <sup>A</sup>	-7.21 <sup>A</sup>	-----
Diisodecyl phthalate	26761-40-0	-5.96 <sup>A</sup>	5.46, 5.78, 6.04 <sup>A</sup>	10.36 <sup>A</sup>	-9.16 <sup>A</sup>	-----
Diisononyl phthalate	28553-12-0	-5.82 <sup>A</sup>	5.51 <sup>A</sup>	9.37 <sup>A</sup>	-9.15 <sup>A</sup>	-----
Diltiazem	42399-41-7	-19.06 (est) <sup>C</sup>	2.30 - 3.47 (est) <sup>C</sup>	2.7 <sup>A</sup>	-13.53 (est) <sup>C</sup>	8.06 <sup>A</sup>
Dimethenamid	87674-68-8	-12.22 (est) <sup>C</sup>	1.98 - 2.15 (est) <sup>C</sup>	2.15 <sup>C</sup>	-6.44 <sup>C</sup>	-----
Dimethyl phthalate	131-11-3	-6.71 <sup>A</sup>	1.74 - 2.58 <sup>A</sup>	1.6 <sup>A</sup>	-5.39 <sup>A</sup>	-----
Dimethyltin	23120-99-2	-1.15 (est) <sup>C</sup>	-2.69 - 1.34 (est) <sup>C</sup>	-3.10 <sup>C</sup>	-0.11 (est) <sup>C</sup>	-----
Dimetridazole	551-92-8	-9.46 (est) <sup>C</sup>	1.71 - 1.93 (est) <sup>C</sup>	0.31 <sup>C</sup>	-6.65 (est) <sup>C</sup>	-----

**Table A4. Select properties of organic contaminants potentially found in stormwater.**

Compound	CAS #	Log $K_H^4$	Log $K_{oc}^5$	Log $K_{ow}^6$	Log $P^{0,7}$	pKa <sup>8</sup>
Units		L m <sup>3</sup> /mol	L/kg	L/kg	atm	
Di-n-butyl phthalate	84-74-2	-5.74 <sup>A</sup>	3.05 <sup>A</sup>	4.5 <sup>A</sup>	-7.58 <sup>A</sup>	-----
Di-n-octyl phthalate	117-84-0	-5.59 <sup>A</sup>	6.3, 4.38, 5.22 <sup>A</sup>	8.1 <sup>A</sup>	-9.88 <sup>A</sup>	-----
Diocetyl tin	94410-05-6	-2.50 (est) <sup>C</sup>	3.91 - 4.99	4.50 (est) <sup>C</sup>	-6.11 (est) <sup>C</sup>	-----
Diphenhydramine	58-73-1	-11.43 (est) <sup>C</sup>	2.58 <sup>A</sup>	3.27 <sup>A</sup>	-8.12 <sup>A</sup>	8.98 <sup>A</sup>
Dipropyl phthalate	131-16-8	-9.39 <sup>C</sup>	2.54 - 2.60 (est) <sup>C</sup>	3.27 <sup>C</sup>	-6.76 <sup>C</sup>	-----
Diuron	330-54-1	-9.30 <sup>A</sup>	2.35 - 2.94 <sup>A</sup>	2.68 <sup>A</sup>	-10.04 <sup>A</sup>	-----
D-limonene	5989-27-5	-1.55 <sup>A</sup>	3.05 <sup>A</sup>	4.57 <sup>A</sup>	-2.59 <sup>A</sup>	-----
Endrin	72-20-8	-5.19 <sup>A</sup>	4.06 <sup>A</sup>	5.20 <sup>A</sup>	-8.40 <sup>A</sup>	-----
Epitestosterone	481-30-1	-11.45 (est) <sup>C</sup>	2.55 - 3.34 (est) <sup>C</sup>	3.32 <sup>C</sup>	-10.65 (est) <sup>C</sup>	-----
Erythromycin	114-07-8	-31.27 (est) <sup>C</sup>	2.76 <sup>A</sup>	3.06 <sup>A</sup>	-27.56 <sup>A</sup>	8.9 <sup>A</sup>
Estriol	50-27-1	-11.89 <sup>A</sup>	3.08 <sup>A</sup>	2.45 <sup>A</sup>	-13.89 <sup>A</sup>	10.54 <sup>A</sup>
Estrone	53-16-7	-9.42 <sup>A</sup>	2.66 - 4.26 <sup>A</sup>	3.13 <sup>A</sup>	-12.48 <sup>A</sup>	-----
Ethylbenzene	100-41-4	-2.1 <sup>A</sup>	2.35 - 2.41 <sup>A</sup>	3.15 <sup>A</sup>	-1.90 <sup>A</sup>	-----
Ethylene thiourea	96-45-7	-10.85 <sup>A</sup>	1.11 <sup>A</sup>	-0.66 <sup>A</sup>	-8.58 <sup>A</sup>	-----
Flumequine	42835-25-6	-12.57 <sup>A</sup>	3.44 - 4.39 <sup>A</sup>	1.6 <sup>A</sup>	-9.49 <sup>A</sup>	6.5 <sup>A</sup>
Fluoranthene	206-44-0	-1.98 <sup>A</sup>	4.47 - 5.47 <sup>A</sup>	5.16 <sup>A</sup>	-7.92 <sup>A</sup>	-----
Fluorene	86-73-7	-1.14 <sup>A</sup>	3.7 - 4.21, 6.45, 6.52 <sup>A</sup>	4.18 <sup>A</sup>	-6.10 <sup>A</sup>	-----
Fluoxetine	54910-89-3	-10.05 (est) <sup>C</sup>	3.05 - 4.97 (est) <sup>C</sup>	4.05 <sup>C</sup>	-7.48 (est) <sup>C</sup>	-----
Formaldehyde	50-00-0	-6.47 <sup>A</sup>	0.9 <sup>A</sup>	0.35 <sup>A</sup>	0.71 <sup>A</sup>	13.27 <sup>A</sup>
Galaxolide	1222-05-5	-3.89 <sup>A</sup>	4.59, 4.86 <sup>A</sup>	5.9 <sup>A</sup>	-6.15 <sup>A</sup>	-----
Gemfibrozil	25812-30-0	-7.92 <sup>A</sup>	2.63 <sup>A</sup>	4.77 <sup>A</sup>	-7.39 <sup>A</sup>	4.5 <sup>A</sup>
Gestodene	60282-87-3	-9.17 <sup>A</sup>	2.51 - 3.91 (est) <sup>C</sup>	3.26 <sup>A</sup>	-11.83 <sup>A</sup>	-----
Glyphosate	1071-83-6	-21.39 (est) <sup>C</sup>	3.41 - 3.69 <sup>A</sup>	-3.40 <sup>A</sup>	-9.89 <sup>A</sup>	2.34, 5.73, 10.2 <sup>A</sup>
Hydroxyatrazine	67-68-5	-13.33 - -14.87 (est) <sup>C</sup>	-0.17 - 2.77 (est) <sup>C</sup>	-1.74 (est) <sup>C</sup>	-9.62 (est) <sup>C</sup>	-----

**Table A4. Select properties of organic contaminants potentially found in stormwater.**

Compound	CAS #	Log K <sub>H</sub> <sup>4</sup>	Log K <sub>OC</sub> <sup>5</sup>	Log K <sub>OW</sub> <sup>6</sup>	Log P <sup>0,7</sup>	pKa <sup>8</sup>
Units		L m <sup>3</sup> /mol	L/kg	L/kg	atm	
Hydroxydeethylatrazine	19988-24-0	-18.34 (est) <sup>C</sup>	-0.42 - 2.66 (est) <sup>C</sup>	-2.27 <sup>C</sup>	-9.90 (est) <sup>C</sup>	-----
Hydroxydeisopropylatrazine	7313-54-4	-18.46 (est) <sup>C</sup>	-0.65 - 2.47 (est) <sup>C</sup>	-2.69 (est) <sup>C</sup>	-9.77 (est) <sup>C</sup>	-----
Ibuprofen	15687-27-1	-6.82 <sup>A</sup>	3.53 <sup>A</sup>	3.97 <sup>A</sup>	-7.21 <sup>A</sup>	4.91, 5.2 <sup>A</sup>
Ifosfamide	3778-73-2	-10.85 <sup>A</sup>	1.85 <sup>A</sup>	0.86 <sup>A</sup>	-7.41 <sup>A</sup>	-----
Indeno pyrene	193-39-5	-6.46 <sup>A</sup>	5.78 - 8.82 <sup>A</sup>	6.7 <sup>A</sup>	-12.77 <sup>A</sup>	-----
Indole	120-72-9	-6.28 <sup>A</sup>	2.27 - 2.54 <sup>A</sup>	2.14 <sup>A</sup>	-4.80 <sup>A</sup>	-2.4 <sup>A</sup>
Indomethacin	53-86-1	-13.5 <sup>A</sup>	2.95 - 3.15 <sup>A</sup>	0.91 <sup>A</sup>	-12.89 <sup>A</sup>	4.5 <sup>A</sup>
Iohexol	66108-95-0	-31.58 (est) <sup>C</sup>	-2.13 - 1.00 (est) <sup>C</sup>	-3.05 <sup>C</sup>	-31.27 (est) <sup>C</sup>	-----
Iomeprol	78649-41-9	-26.75 (est) <sup>C</sup>	-1.99 - 1.00 (est) <sup>C</sup>	-2.79 <sup>C</sup>	-31.40 (est) <sup>C</sup>	-----
Iopamidol	60166-93-0	-24.96 <sup>A</sup>	1 <sup>A</sup>	-2.42 <sup>A</sup>	-32.77 <sup>A</sup>	10.7 <sup>A</sup>
Iopromide	73334-07-3	-31.00 (est) <sup>C</sup>	-1.67 - 1.00 (est) <sup>C</sup>	-2.05 <sup>C</sup>	-30.68 (est) <sup>C</sup>	-----
Isophorone	78-59-1	-5.18 <sup>A</sup>	2.3 <sup>A</sup>	1.7 <sup>A</sup>	-3.24 <sup>A</sup>	-----
Isopropyl chloride	75-29-6	-1.74 <sup>A</sup>	1.72 <sup>A</sup>	1.9 <sup>A</sup>	-0.17 <sup>A</sup>	-----
Isoproturon	34123-59-6	-11.72 (est) <sup>C</sup>	2.30 - 2.44 (est) <sup>C</sup>	2.87 <sup>C</sup>	-10.49 <sup>C</sup>	-----
Isoquinoline	119-65-3	-9.38 - -9.16 (est) <sup>C</sup>	1.81 - 3.19 (est) <sup>C</sup>	2.08 <sup>C</sup>	-4.32 <sup>C</sup>	-----
Ketoprofen	22071-15-4	-13.67 (est) <sup>C</sup>	2.08 - 2.59 (est) <sup>C</sup>	3.12 <sup>C</sup>	-8.72 (est) <sup>C</sup>	-----
Lincomycin	154-21-2	-22.52 <sup>A</sup>	1.84 <sup>A</sup>	0.2 <sup>A</sup>	-19.77 <sup>A</sup>	7.6, 7.8 <sup>A</sup>
Lindane	58-89-9	-5.29 <sup>A</sup>	2.60 - 3.30 <sup>A</sup>	3.72 <sup>A</sup>	-7.26 <sup>A</sup>	-----
Lomefloxacin	98079-51-7	-20.87 (est) <sup>C</sup>	0.01 - 1.70 (est) <sup>C</sup>	-0.30 <sup>C</sup>	-14.25 (est) <sup>C</sup>	-----
Malathion	121-75-5	-8.31 <sup>A</sup>	2.96 - 4.25 <sup>A</sup>	2.36 <sup>A</sup>	-7.28 <sup>A</sup>	-----
Mecoprop	93-65-2	-8.42 <sup>A</sup>	0.7 - 1.6 <sup>A</sup>	3.13 <sup>A</sup>	-7.80 <sup>A</sup>	3.10, 3.21, 3.78 <sup>A</sup>
Metalaxyl	57837-19-1	-8.52 <sup>A</sup>	1.48 - 2.45 <sup>A</sup>	1.65 <sup>A</sup>	-8.13 <sup>A</sup>	-----
Metaldehyde	108-62-3	-4.28 <sup>A</sup>	2.38 <sup>A</sup>	0.12 <sup>A</sup>	-4.18 <sup>A</sup>	-----
Metazachlor	67129-08-2	-13.24 (est) <sup>C</sup>	2.46 - 3.00 (est) <sup>C</sup>	2.13 <sup>C</sup>	-8.82 (est) <sup>C</sup>	-----
Metformin	1115-70-4	-15.12 <sup>A</sup>	1.08 - 1.28 <sup>A</sup>	-2.64 <sup>A</sup>	-7.00 <sup>A</sup>	-----

**Table A4. Select properties of organic contaminants potentially found in stormwater.**

Compound	CAS #	Log K <sub>H</sub> <sup>4</sup>	Log K <sub>OC</sub> <sup>5</sup>	Log K <sub>OW</sub> <sup>6</sup>	Log P <sup>0,7</sup>	pKa <sup>8</sup>
Units		L m <sup>3</sup> /mol	L/kg	L/kg	atm	
Methyl dihydrojasmonate	24851-98-7	-10.28 - -9.30 (est) <sup>C</sup>	2.18 - 2.70 (est) <sup>C</sup>	2.98 (est) <sup>C</sup>	-5.81 (est) <sup>C</sup>	-----
Methyl salicylate	119-36-8	-6.03 <sup>A</sup>	1.52 - 2.11 <sup>A</sup>	2.55 <sup>A</sup>	-4.35 <sup>A</sup>	9.8 <sup>A</sup>
Methyl tert-butyl ether (MTBE)	1634-04-4	-3.23 <sup>A</sup>	1.04 - 1.08 <sup>A</sup>	0.94 <sup>A</sup>	-0.48 <sup>A</sup>	-3.7 <sup>A</sup>
Methylparaben	99-76-3	-8.64 <sup>A</sup>	1.94 <sup>A</sup>	1.96 <sup>A</sup>	-6.51 <sup>A</sup>	8.4, 8.5 <sup>A</sup>
Methyltriclosan	4640-01-1	-8.55 - -6.85 (est) <sup>C</sup>	4.07 (est) <sup>C</sup>	5.22 (est) <sup>C</sup>	-8.04 (est) <sup>C</sup>	-----
Metolachlor	51218-45-2	-8.05 <sup>A</sup>	1.34 - 3.37 <sup>A</sup>	3.13 <sup>A</sup>	-7.39 <sup>A</sup>	-----
Metoprolol	37350-58-6	-10.68 <sup>A</sup>	1.79 <sup>A</sup>	1.88 <sup>A</sup>	-9.44 <sup>A</sup>	9.6 <sup>A</sup>
Metronidazole	443-48-1	-10.77 <sup>A</sup>	1.36 <sup>A</sup>	-0.02 <sup>A</sup>	-9.39 <sup>A</sup>	2.38 <sup>A</sup>
Miconazole	22916-47-8	-11.61 (est) <sup>C</sup>	4.83 - 5.74 (est) <sup>C</sup>	6.25 (est) <sup>C</sup>	-12.63 (est) <sup>C</sup>	-----
Monobromoacetic acid	79-08-3	-8.19 <sup>A</sup>	1.6 <sup>A</sup>	0.41 <sup>A</sup>	-3.81 <sup>A</sup>	2.89 <sup>A</sup>
Monobutyltin	78763-54-9	-4.01 (est) <sup>C</sup>	1.86 - 2.87 (est) <sup>C</sup>	3.31 (est) <sup>C</sup>	-1.00 (est) <sup>C</sup>	-----
Monochloroacetic acid	79-11-8	-8.03 <sup>A</sup>	1.49 <sup>A</sup>	0.22 <sup>A</sup>	-4.07 <sup>A</sup>	2.87 <sup>A</sup>
Musk ambrette	83-66-9	-6.15 <sup>A</sup>	3.72 <sup>A</sup>	4 <sup>A</sup>	-7.77 <sup>A</sup>	-----
Musk ketone	81-14-1	-8.72 <sup>A</sup>	3.94 <sup>A</sup>	4.3 <sup>A</sup>	-9.12 <sup>A</sup>	-----
Musk xylene	81-15-2	-8.11 <sup>A</sup>	3.83 - 4.53 (est) <sup>C</sup>	4.4 <sup>A</sup>	-9.08 <sup>A</sup>	-----
m-xylene	108-38-3	-2.14 <sup>A</sup>	2.22 - 2.44 <sup>A</sup>	3.2 <sup>A</sup>	-1.96 <sup>A</sup>	-----
N,N-Diethyl-3-methylbenzamide (DEET)	134-62-3	-7.68 <sup>A</sup>	2.06 <sup>A</sup>	2.02 <sup>A</sup>	-5.58 <sup>A</sup>	-----
N4-acetyl-Sulfamethazine	100-90-3	-12.20 (est) <sup>C</sup>	1.98 - 2.26 (est) <sup>C</sup>	1.48 (est) <sup>C</sup>	-13.43 (est) <sup>C</sup>	-----
Nalidixic Acid	389-08-2	-18.29 (est) <sup>C</sup>	1.00 - 1.34 (est) <sup>C</sup>	1.41 <sup>A</sup>	-10.09 (est) <sup>C</sup>	8.6 <sup>A</sup>
Naphthalene	91-20-3	-0.37 <sup>A</sup>	2.05 - 5.59 <sup>A</sup>	3.3 <sup>A</sup>	-3.95 <sup>A</sup>	-----
Naproxen	22204-53-1	-9.47 <sup>A</sup>	2.52 <sup>A</sup>	3.18 <sup>A</sup>	-8.60 <sup>A</sup>	4.15 <sup>A</sup>
Nitritotriacetic acid	139-13-9	-18.92 (est) <sup>C</sup>	<2.46 <sup>A</sup>	-3.81 (est) <sup>C</sup>	-11.04 <sup>A</sup>	3.03, 3.07, 10.7 <sup>A</sup>
Nonylphenol	25154-52-3	-5.25 <sup>A</sup>	4 - 4.7 <sup>A</sup>	5.71 <sup>A</sup>	-6.91 <sup>A</sup>	10.25 <sup>A</sup>
Nonylphenol ethoxylates	9016-45-9	-11.59 - -11.57	3.39 - 3.41 (est) <sup>C</sup>	5.30 (est) <sup>C</sup>	-10.92 (est) <sup>C</sup>	-----

**Table A4. Select properties of organic contaminants potentially found in stormwater.**

Compound	CAS #	Log K <sub>H</sub> <sup>4</sup>	Log K <sub>OC</sub> <sup>5</sup>	Log K <sub>OW</sub> <sup>6</sup>	Log P <sup>0,7</sup>	pKa <sup>8</sup>
Units		L m <sup>3</sup> /mol (est) <sup>C</sup>	L/kg	L/kg	atm	
Norfloxacin	70458-96-7	-21.06 (est) <sup>C</sup>	4.79 <sup>A</sup>	0.46 <sup>A</sup>	-15.05 <sup>A</sup>	6.32, 8.75 <sup>A</sup>
N-propyl chloride	540-54-5	-1.89 <sup>A</sup>	1.6 <sup>A</sup>	2.04 <sup>A</sup>	-0.35 <sup>A</sup>	-----
Octylethoxylate	9002-93-1	-9.91 - -9.15 (est) <sup>C</sup>	2.90 - 3.26 (est) <sup>C</sup>	4.86 (est) <sup>C</sup>	6.99	-----
Octylphenol	949-13-3	-8.35 - -8.06 (est) <sup>C</sup>	4.13 - 4.33 (est) <sup>C</sup>	5.50 (est) <sup>C</sup>	-6.89 (est) <sup>C</sup>	-----
Ofloxacin	82419-36-1	-22.30 (est) <sup>C</sup>	4.64 <sup>A</sup>	-0.39 <sup>A</sup>	-14.89 <sup>A</sup>	5.97, 9.28 <sup>A</sup>
OH-ibuprofen	51146-55-5	-14.26 (est) <sup>C</sup>	1.01 - 1.46 (est) <sup>C</sup>	2.29 (est) <sup>C</sup>	-9.26 (est) <sup>C</sup>	-----
op' DDA	34113-46-7	-7.49 - -8.71 (est) <sup>C</sup>	2.56 - 3.45 (est) <sup>C</sup>	4.35 (est) <sup>C</sup>	-9.06 (est) <sup>C</sup>	-----
Oxazepam	604-75-1	-9.26 <sup>A</sup>	2.59 <sup>A</sup>	2.24 <sup>A</sup>	-14.26 <sup>A</sup>	1.55, 10.9 <sup>A</sup>
o-xylene	95-47-6	-2.29 <sup>A</sup>	1.38 - 2.4 <sup>A</sup>	3.12 <sup>A</sup>	-2.06 <sup>A</sup>	-----
Oxytetracycline	79-57-2	-24.77 <sup>A</sup>	2.29 - 4.97 <sup>A</sup>	-0.9 <sup>A</sup>	-26.90 <sup>A</sup>	9.5 <sup>A</sup>
Para-cresol	106-44-5	-6 <sup>A</sup>	1.43 - 2.81 <sup>A</sup>	1.94 <sup>A</sup>	-3.84 <sup>A</sup>	10.26 <sup>A</sup>
Para-tert-octylphenol	140-66-9	-5.16 <sup>A</sup>	4.00 - 4.01 (est) <sup>C</sup>	5.28 <sup>A</sup>	-6.20 <sup>C</sup>	-----
Pentachlorophenol	87-86-5	-7.61 <sup>A</sup>	3.1- 5.09 <sup>A</sup>	5.12 <sup>A</sup>	-6.84 <sup>A</sup>	4.7 <sup>A</sup>
Pentoxifylline	6493-05-6	-16.25 (est) <sup>C</sup>	1.00 - 1.38 (est) <sup>C</sup>	0.29 <sup>C</sup>	-12.80 (est) <sup>C</sup>	-----
Perfluorobutanesulfonic Acid (PFBS)	375-73-5	-4.84 <sup>A</sup>	1.42 - 2.26 <sup>A</sup>	1.82 <sup>A</sup>	-4.45 <sup>A</sup>	-3.31 <sup>A</sup>
Perfluorooctanionic Acid (PFOA)	335-67-1	-1.04 <sup>A</sup>	1.92 - 2.59 <sup>A</sup>	4.81 <sup>A</sup>	-4.38 <sup>A</sup>	-0.5 - 4.2 <sup>A</sup>
Perfluorodecanoic Acid (PFDA)	335-76-2	-2.60 (est) <sup>C</sup>	3.56 - 5.72 (est) <sup>C</sup>	6.15 (est) <sup>C</sup>	-4.45 (est) <sup>C</sup>	-----
Perfluorododecanoic Acid (PFDoA)	307-55-1	-1.16 (est) <sup>C</sup>	4.30 - 7.03 (est) <sup>C</sup>	7.49 (est) <sup>C</sup>	-5.25 (est) <sup>C</sup>	-----
Perfluoroheptanoic Acid (PFHpA)	375-85-9	-1.77 <sup>A</sup>	1.52 - 4 <sup>A</sup>	4.15 <sup>A</sup>	-3.76 <sup>A</sup>	-2.29 <sup>A</sup>
Perfluorohexanesulfonic Acid(PFHxS)	355-46-4	-3.4 <sup>A</sup>	0.97 <sup>A</sup>	3.16 <sup>A</sup>	-5.22 <sup>A</sup>	0.14 <sup>A</sup>
Perfluorohexanoic Acid (PFHxA)	307-24-4	-5.48 (est) <sup>C</sup>	1.63 - 4.7 <sup>A</sup>	3.48 <sup>A</sup>	-2.59 <sup>A</sup>	-0.16 <sup>A</sup>
Perfluorononanoic Acid (PFNA)	375-95-1	-3.32 (est) <sup>C</sup>	2.4 - 5.08 <sup>A</sup>	5.48 <sup>A</sup>	-3.96 <sup>A</sup>	-0.21 <sup>A</sup>

**Table A4. Select properties of organic contaminants potentially found in stormwater.**

Compound	CAS #	Log $K_H^4$	Log $K_{oc}^5$	Log $K_{ow}^6$	Log $P^{0,7}$	pKa <sup>8</sup>
Units		L m <sup>3</sup> /mol	L/kg	L/kg	atm	
Perfluorooctanesulfonic Acid (PFOS)	1763-23-1	-3.39 <sup>A</sup>	2.1 - 4.7 <sup>A</sup>	4.49 <sup>A</sup>	-5.58 <sup>A</sup>	<1.0 <sup>A</sup>
Perfluoroundecanoic Acid (PFUnA)	4234-23-5	-33.30 (est) <sup>C</sup>	0.86 - 3.98 (est) <sup>C</sup>	2.75 (est) <sup>C</sup>	-31.87 (est) <sup>C</sup>	-----
Perylene	198-55-0	-6.38 <sup>A</sup>	6.84 <sup>A</sup>	6.3 <sup>A</sup>	-12.18 <sup>A</sup>	-----
Phenanthrene	85-01-8	-1.59 <sup>A</sup>	3.96 - 7 <sup>A</sup>	4.46 <sup>A</sup>	-6.8 <sup>A</sup>	-----
Phenol	108-95-2	-3.39 <sup>A</sup>	3.46 - 3.49 <sup>A</sup>	1.46 <sup>A</sup>	-3.34 <sup>A</sup>	9.99 <sup>A</sup>
Pipemidic Acid	51940-44-4	-20.84 (est) <sup>C</sup>	-0.61 - 1.00 (est) <sup>C</sup>	-2.15 <sup>C</sup>	-14.63 (est) <sup>C</sup>	-----
pp' DDA	83-05-6	-7.62 - 8.77 (est) <sup>C</sup>	2.63 - 3.45 (est) <sup>C</sup>	4.48 - 4.64 <sup>C</sup>	-9.31 (est) <sup>C</sup>	-----
Primidone	125-33-7	-12.71 (est) <sup>C</sup>	1.38 - 2.14 (est) <sup>C</sup>	0.91 <sup>A</sup>	-12.36 (est) <sup>C</sup>	-----
Progesterone	57-83-0	-7.19 <sup>A</sup>	3.45 <sup>A</sup>	3.87 <sup>A</sup>	-6.33 <sup>A</sup>	-----
Propazine	139-40-2	-8.34 <sup>A</sup>	1.92 - 2.7 <sup>A</sup>	2.93 <sup>A</sup>	-9.77 <sup>A</sup>	1.7 <sup>A</sup>
Propranolol	525-66-6	-15.10 (est) <sup>C</sup>	2.45 - 2.96 (est) <sup>C</sup>	3.48 <sup>C</sup>	-9.91 (est) <sup>C</sup>	-----
Propylparaben	94-13-3	-8.37 <sup>A</sup>	2.46 <sup>A</sup>	3.04 <sup>A</sup>	-6.40 <sup>A</sup>	8.5 <sup>A</sup>
Propyphenazone	479-92-5	-11.74 (est) <sup>C</sup>	1.93 - 2.81 (est) <sup>C</sup>	1.94 <sup>C</sup>	-7.87 (est) <sup>C</sup>	-----
p-xylene	106-42-3	-2.16 <sup>A</sup>	2.39 - 2.73 <sup>A</sup>	3.15 <sup>A</sup>	-1.94 <sup>A</sup>	-----
Pyrene	129-00-0	-2.05 <sup>A</sup>	3.54 - 6.8 <sup>A</sup>	4.88 <sup>A</sup>	-8.23 <sup>A</sup>	-----
Roxithromycin	80214-83-1	-33.30 (est) <sup>C</sup>	0.86 - 3.98 (est) <sup>C</sup>	2.75 (est) <sup>C</sup>	-31.87 (est) <sup>C</sup>	-----
Salicylic acid	69-72-7	-8.14 <sup>A</sup>	2.6 <sup>A</sup>	2.26 <sup>A</sup>	-6.97 <sup>A</sup>	2.98 <sup>A</sup>
Sertraline	79617-96-2	-10.29 (est) <sup>C</sup>	3.81 - 5.23 (est) <sup>C</sup>	5.29 (est) <sup>C</sup>	-8.81 (est) <sup>C</sup>	-----
Simazine	122-34-9	-9.03 <sup>A</sup>	1.89 - 3.55 <sup>A</sup>	2.18 <sup>A</sup>	-10.54 <sup>A</sup>	1.62 <sup>A</sup>
Skatole	83-34-1	-5.68 <sup>A</sup>	2.78 <sup>A</sup>	2.6 <sup>A</sup>	-5.14 <sup>A</sup>	-----
Sodium decylbenzenesulfonate	1322-98-1	-10.45 (est) <sup>C</sup>	2.04 - 3.57 (est) <sup>C</sup>	2.02 (est) <sup>C</sup>	-16.77 (est) <sup>C</sup>	-----
Sodium dodecylbenzenesulfonate	25155-30-0	-10.20 <sup>C</sup>	2.58 - 4.10 (est) <sup>C</sup>	0.045 <sup>A</sup>	-17.52 <sup>C</sup>	-----
Sodium N-tridecylbenzenesulfonate	26248-24-8	-10.08 (est) <sup>C</sup>	2.86 - 4.35 (est) <sup>C</sup>	2.52 <sup>A</sup>	-17.90 (est) <sup>C</sup>	-----
Sodium Undecylbenzenesulfonate	27636-75-5	-10.32 (est) <sup>C</sup>	2.13 - 3.83 (est) <sup>C</sup>	2.51 (est) <sup>C</sup>	-17.15 (est) <sup>C</sup>	-----

**Table A4. Select properties of organic contaminants potentially found in stormwater.**

Compound	CAS #	Log K <sub>H</sub> <sup>4</sup>	Log K <sub>OC</sub> <sup>5</sup>	Log K <sub>OW</sub> <sup>6</sup>	Log P <sup>0,7</sup>	pKa <sup>8</sup>
Units		L m <sup>3</sup> /mol	L/kg	L/kg	atm	
Sotalol	3930-20-9	-16.60 (est) <sup>C</sup>	0.74 - 1.35 (est) <sup>C</sup>	0.24 <sup>C</sup>	-14.75 (est) <sup>C</sup>	-----
Sucralose	56038-13-2	-18.4 <sup>A</sup>	1 <sup>A</sup>	-1 <sup>A</sup>	-16.37 <sup>A</sup>	-----
Sulfadiazine	68-35-9	-12.80 (est) <sup>C</sup>	1.39 - 1.87 (est) <sup>C</sup>	-0.09 <sup>C</sup>	-11.16 (est) <sup>C</sup>	-----
Sulfadimethoxine	122-11-2	-16.89 (est) <sup>C</sup>	1.48 - 2.45 (est) <sup>C</sup>	1.63 <sup>C</sup>	-14.71 (est) <sup>C</sup>	-----
Sulfamethazine	57-68-1	-9.71 <sup>A</sup>	1.65 - 2.39 <sup>A</sup>	0.14 <sup>A</sup>	-11.05 <sup>A</sup>	2.07, 2.65, 7.49, 7.65 <sup>A</sup>
Sulfamethoxazole	723-46-6	-12.19 <sup>A</sup>	1.86 <sup>A</sup>	0.89 <sup>A</sup>	-10.04 <sup>A</sup>	1.6, 5.7 <sup>A</sup>
Sulfamonomethoxine	1220-83-3	-16.62 (est) <sup>C</sup>	1.68 - 1.89 (est) <sup>C</sup>	0.70 <sup>C</sup>	-10.96 (est) <sup>C</sup>	-----
Sulfanilamide	63-74-1	-9.82 <sup>A</sup>	1.04 <sup>A</sup>	-0.62 <sup>A</sup>	-8.02 <sup>A</sup>	10.43, 10.58, 11.63 <sup>A</sup>
Sulfathiazole	72-14-0	-13.24 <sup>A</sup>	1.99 - 2.3 <sup>A</sup>	0.05 <sup>A</sup>	-10.26 <sup>A</sup>	2.2, 7.24 <sup>A</sup>
Terbutylazine	5915-41-3	-7.64 <sup>A</sup>	2.18 - 2.71 <sup>A</sup>	3.4 <sup>A</sup>	-9.05 <sup>A</sup>	2 <sup>A</sup>
Terbutryne	886-50-0	-7.68 <sup>A</sup>	2.56 - 4.62 <sup>A</sup>	3.74 <sup>A</sup>	-8.65 <sup>A</sup>	4.3 <sup>A</sup>
Tert-Amyl chloride	594-36-5	-4.59 (est) <sup>C</sup>	1.94 - 2.19 (est) <sup>C</sup>	2.52 <sup>C</sup>	-1.04 (est) <sup>C</sup>	-----
Tertrabutyltin	1461-25-2	-2.20 (est) <sup>C</sup>	4.90 - 8.13 (est) <sup>C</sup>	9.37 (est) <sup>C</sup>	-5.20 <sup>A</sup>	-----
Testosterone	58-22-0	-8.46 <sup>A</sup>	3.25 - 3.52 <sup>A</sup>	3.32 <sup>A</sup>	-10.65 <sup>A</sup>	-----
Tetrachloroethylene	127-18-4	-1.75 <sup>A</sup>	2.33 <sup>A</sup>	3.4 <sup>A</sup>	-1.62 <sup>A</sup>	-----
Tetracycline	60-54-8	-26.33 (est) <sup>C</sup>	-0.12 - 1.64 (est) <sup>C</sup>	-1.37 <sup>A</sup>	-23.56 (est) <sup>C</sup>	3.3 <sup>A</sup>
Thiabendazole	148-79-8	-10.68 <sup>A</sup>	3.4 - 3.67 <sup>A</sup>	2.47 <sup>A</sup>	11.28 <sup>A</sup>	4.64 <sup>A</sup>
Toluene	108-88-3	-2.18 <sup>A</sup>	1.57 - 2.25 <sup>A</sup>	2.73 <sup>A</sup>	-1.43 <sup>A</sup>	-----
Tolylfluanid	731-27-1	-6.12 <sup>A</sup>	3.51 <sup>A</sup>	3.9 <sup>A</sup>	-8.71 <sup>A</sup>	-----
Tonalide	21145-77-7	-3.85 <sup>A</sup>	3.8 - 4.8 <sup>A</sup>	5.7 <sup>A</sup>	-6.17 <sup>A</sup>	-----
Tramadol	27203-92-5	-10.81 <sup>A</sup>	2.79 <sup>A</sup>	3.01 <sup>A</sup>	-9.22 <sup>A</sup>	9.23, 13.08 <sup>A</sup>
Tribromoacetic acid	75-96-7	-8.48 <sup>A</sup>	0.72 <sup>A</sup>	1.71 <sup>A</sup>	-6.44 <sup>A</sup>	0.72 <sup>A</sup>
Tribromomethane	75-25-2	-3.27 <sup>A</sup>	2.06 - 2.1 <sup>A</sup>	2.4 <sup>A</sup>	-2.15 <sup>A</sup>	-----
Tributyl phosphate	126-73-8	-5.85 <sup>A</sup>	3.38 <sup>A</sup>	4 <sup>A</sup>	-5.83 <sup>A</sup>	-----

**Table A4. Select properties of organic contaminants potentially found in stormwater.**

Compound	CAS #	Log K <sub>H</sub> <sup>4</sup>	Log K <sub>oc</sub> <sup>5</sup>	Log K <sub>ow</sub> <sup>6</sup>	Log P <sup>0,7</sup>	pKa <sup>8</sup>
Units		L m <sup>3</sup> /mol	L/kg	L/kg	atm	
Tributyltin	688-73-3	-2.81 (est) <sup>C</sup>	3.56 - 3.91 (est) <sup>C</sup>	4.10 <sup>C</sup>	-4.28 (est) <sup>C</sup>	-----
Trichloroacetic acid	76-03-9	-7.87 <sup>A</sup>	2.11 <sup>A</sup>	1.33 <sup>A</sup>	-4.10 <sup>A</sup>	0.51 <sup>A</sup>
Trichloroethylene	79-01-6	-2.04 <sup>A</sup>	1.69 - 3.43 <sup>A</sup>	2.61 <sup>A</sup>	-1.04 <sup>A</sup>	-----
Trichloromethane	67-66-3	-2.44 <sup>A</sup>	1.53 - 2.29 <sup>A</sup>	1.97 <sup>A</sup>	-0.59 <sup>A</sup>	-----
Triclosan	3380-34-5	-7.68 <sup>A</sup>	3.38 - 4.20 <sup>A</sup>	4.76 <sup>A</sup>	-8.22 <sup>A</sup>	7.9 <sup>A</sup>
Tricyclohexyltin chloride	3091-32-5	-5.76 (est) <sup>C</sup>	5.72 - 6.15 (est) <sup>C</sup>	7.09 (est) <sup>C</sup>	-6.91 - -7.95 (est) <sup>C</sup>	-----
Tricyclohexyltin Hydroxide	13121-70-5	-7.58 (est) <sup>C</sup>	4.42 - 5.05 (est) <sup>C</sup>	8.20 (est) <sup>C</sup>	-9.31 (est) <sup>C</sup>	-----
Triethylphosphate	78-40-0	-7.44 <sup>A</sup>	1.81 <sup>A</sup>	0.8 <sup>A</sup>	-3.29 <sup>A</sup>	-----
Trifluoromethane	75-46-7	-1.02 <sup>A</sup>	1.51 <sup>A</sup>	0.64 <sup>A</sup>	1.67 <sup>A</sup>	-----
Triiodomethane	75-47-8	-4.51 <sup>A</sup>	1.54 <sup>A</sup>	3.03 (est) <sup>C</sup>	-4.28 <sup>A</sup>	-----
Triisobutyl phosphate	126-71-6	-8.50 (est) <sup>C</sup>	3.02 - 3.15 (est) <sup>C</sup>	3.60 (est) <sup>C</sup>	-4.77 (est) <sup>C</sup>	-----
Trimethoprim	738-70-5	-13.62 <sup>A</sup>	1.88 <sup>A</sup>	0.91 <sup>A</sup>	-10.89 <sup>A</sup>	7.12 <sup>A</sup>
Trimethyltin Chloride	1066-45-1	-4.40 (est) <sup>C</sup>	0.243 - 1.64 (est) <sup>C</sup>	0.24 - 1.64 (est) <sup>C</sup>	-2.91 (est) <sup>C</sup>	-----
Tri-n-Butyltin Hydride	688-73-3	-2.82 (est) <sup>C</sup>	3.56 - 3.91 (est) <sup>C</sup>	4.10 <sup>C</sup>	-4.28 <sup>A</sup>	-----
Trioctyltin chloride	2587-76-0	-3.63 (est) <sup>C</sup>	7.21 - 9.19 (est) <sup>C</sup>	10.59 (est) <sup>C</sup>	-7.48 (est) <sup>C</sup>	-----
Triphenyl phosphate	115-86-6	-5.48 <sup>A</sup>	3.4-3.55 <sup>A</sup>	4.59 <sup>A</sup>	-8.08 <sup>A</sup>	-----
Triphenylphosphine oxide	791-28-6	-12.28 (est) <sup>C</sup>	2.26 - 3.29 (est) <sup>C</sup>	2.83 <sup>C</sup>	-11.47 <sup>C</sup>	-----
Triphenyltin Acetate	900-95-8	-8.53 (est) <sup>C</sup>	1.32 - 4.84 (est) <sup>C</sup>	3.43 <sup>A</sup>	-7.73 <sup>A</sup>	-----
Triphenyltin chloride	639-58-7	-8.53 (est) <sup>C</sup>	3.64 - 5.72 (est) <sup>C</sup>	4.19 <sup>A</sup>	-2.12 <sup>A</sup>	-----
Triphenyltin hydride	892-20-6	-7.11 (est) <sup>C</sup>	3.04 - 5.53 (est) <sup>C</sup>	3.50 (est) <sup>C</sup>	-6.99 (est) <sup>C</sup>	-----
Triphenyltin hydroxide	76-87-9	-11.93 (est) <sup>C</sup>	3.30 <sup>A</sup>	3.53 <sup>A</sup>	-9.33 <sup>A</sup>	5.20 <sup>A</sup>
Tris(1,3-dichloroisopropyl)phosphate	13674-87-8	-8.59 <sup>A</sup>	3.04 <sup>A</sup>	3.65 <sup>A</sup>	-9.43 <sup>A</sup>	-----
Tris(2-butoxyethyl)phosphate	78-51-3	-10.92 <sup>A</sup>	3.1 <sup>A</sup>	3.75 <sup>A</sup>	-8.80 <sup>A</sup>	-----
Tris(2-chloroethyl)phosphate	115-96-8	-5.48 <sup>A</sup>	2.59 <sup>A</sup>	1.43, 1.78 <sup>A</sup>	-4.10 <sup>A</sup>	-----

**Table A4. Select properties of organic contaminants potentially found in stormwater.**

Compound	CAS #	Log K <sub>H</sub> <sup>4</sup>	Log K <sub>OC</sub> <sup>5</sup>	Log K <sub>OW</sub> <sup>6</sup>	Log P <sup>0,7</sup>	pKa <sup>8</sup>
Units		L m <sup>3</sup> /mol	L/kg	L/kg	atm	
Tylosin	1401-69-0	-37.24 <sup>A</sup>	2.74 - 4.98 <sup>A</sup>	1.63 <sup>A</sup>	-36.59 <sup>A</sup>	7.73 <sup>A</sup>
Valsartan	137862-53-4	-17.51 <sup>A</sup>	2.4 - 4.36 <sup>A</sup>	4 <sup>A</sup>	-17.97 <sup>A</sup>	3.6, 4, 4.61 <sup>A</sup>
Venlafaxine	93413-69-5	-10.69 <sup>A</sup>	2.28 <sup>A</sup>	3.2 <sup>A</sup>	-9.48 <sup>A</sup>	10.09 <sup>A</sup>
Xylene	1330-20-7	-2.29 - -2.14 <sup>A</sup>	2.11 - 2.52 <sup>A</sup>		-2.06 - -1.94 <sup>A</sup>	-----
α-BHC	319-84-6	-5.18 <sup>A</sup>	3.25 - 3.30 <sup>A</sup>	3.8 <sup>A</sup>	-7.23 <sup>A</sup>	-----
α-Endosulfan	959-98-8	-10.04 (est) <sup>C</sup>	3.21 - 3.83 (est) <sup>C</sup>	3.83 <sup>C</sup>	-9.10 <sup>C</sup>	-----
α-Hexachloro-cyclohexane	319-84-6	-5.18 <sup>A</sup>	3.25 - 3.30 <sup>A</sup>	3.8 <sup>A</sup>	-7.23 <sup>A</sup>	-----

<sup>1</sup>Melting Point  
<sup>2</sup>Boiling Point  
<sup>3</sup>C<sub>w</sub> is the solubility in water  
<sup>4</sup>K<sub>H</sub> is the Henry's Law constant  
<sup>5</sup>K<sub>OC</sub> is the organic carbon – water partitioning coefficient  
<sup>6</sup>K<sub>ow</sub> is the octanol – water partitioning coefficient  
<sup>7</sup>P<sup>0</sup> is the vapor pressure  
<sup>8</sup>pKa is the acid dissociation constant  
<sup>A</sup>Toxnet: <https://toxnet.nlm.nih.gov/newtoxnet/hsdb.htm> - Only values that were peer reviewed were used unless denoted by est for estimated value  
<sup>C</sup>EPI Suite version 4.1: U.S. EPA. Office of Pollution Prevention and Toxics and Syracuse Research Corporation. Copyright 2000 – 2012. Peer reviewed values were used when available, otherwise the values obtained are delineated using est  
Decomp= decomposes  
est= estimated value

# DISCONTINUED



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