8-3-2015

Winter Deicing Effects on Groundwater Quality Beneath Permeable Asphalt

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Winter Deicing Effects on Groundwater Quality Beneath Permeable Asphalt

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B.A., University of Vermont, 2009

A Thesis Submitted
in Partial Fulfillment of the
Requirements for the Degree of
Master of Science
at the
University of Connecticut 2015
Approval Page

Master of Science Thesis

Winter Deicing Effects on Groundwater Quality Beneath Permeable Asphalt

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Acknowledgements

This study was funded by the Connecticut Sea Grant through a National Oceanic and Atmospheric Agency grant. The Department of Natural Resources and the Environment and the Center for Integrative Geosciences at the University of Connecticut provided facilities, materials and supplies.

I would like to thank my advisor and friend, Dr. Michael Dietz for his patience, guidance and dedication to this project. I am lucky to have had the opportunity to work with him and learn from his example. I would also like to thank Dr. Gary Robbins for first introducing me to the world of groundwater hydrology and for giving me the opportunity to do something exciting with my life. Additionally, I would like to thank Dr. Glenn Warner and Dr. Jean Crespi for their advice and council during this project, as well as Dr. Jack Clausen for generously allowing me use of his water quality lab and equipment.
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ABSTRACT

This study examined the effects of winter deicing on groundwater quality beneath an 860-m² permeable asphalt parking lot at the University of Connecticut in Storrs, Connecticut. While the beneficial impacts of permeable asphalt on water quality and hydrology have been documented in the literature and include increases in groundwater recharge, reduced peak flows and reductions in metals and hydrocarbon contaminant concentrations in groundwater, chloride has not been observed to be filtered from percolate beneath permeable asphalt. The potential issues related to chloride contamination of groundwater include aquatic toxicity and heavy metals mobilization. Monitoring wells were installed up-gradient and down-gradient in which electrical conductivity was monitored at 10 minute intervals from September 26, 2014 to May 8, 2014. Biweekly samples were collected and analyzed for chloride during the period of study and used to develop a predictive relationship which allowed electrical conductivity to serve as a proxy for continuous chloride concentrations. Samples were also analyzed for sodium (Na), calcium (Ca) and magnesium (Mg) on two occasions before deicing activities began in September and October 2014 and two occasions while deicing activities were active in January and March 2015 and used to evaluate cation exchange potential.

Changes in electrical conductivity and chloride were found to be directly proportional to changes in water table elevations after deicing salt applications and snowmelt percolation and inversely proportional during non-winter months due to dilution by percolating water. Chloride concentrations in groundwater beneath permeable asphalt increased by 9.480 % over the period of study, though the mean concentration of 303.7 mg/L calculated down-gradient was lower than the up-gradient mean concentration of 1,280 mg/L. Ca and Mg in the down-gradient well increased by 4.632% and 1.946%, respectively, though metals concentrations in the up-gradient
well were higher than down-gradient for all sampling dates. Despite the frequent high peaks of chloride as well as the increases in metals observed in the down-gradient well, monitoring throughout the study period revealed lower contaminant concentrations down-gradient than up-gradient before and after deicing activities were active. These results suggest that the use of permeable asphalt in environments with high background contaminant concentrations is beneficial to shallow groundwater quality in the long term due to dilution.
LITERATURE REVIEW

INTRODUCTION

Throughout the past century land use across the world has changed significantly. Agricultural and natural landscapes have given way to urban and suburban development as societies changed from largely agrarian to industrial societies (Arnold and Gibbons 1996).

Between the years 1985 to 2010, developed areas and turf and grass in Connecticut have increased by 148.9 and 74.9 square miles, respectively, with agricultural lands and forested areas decreasing by 62.9 and 190.1 square miles, respectively (CLEAR 2010). This change from rural to urban has resulted in an increase in impervious cover (IC), such as roads, rooftops, sidewalks and other surfaces that prevent the infiltration of water into the ground. Increases in IC therefore become the dominant variable which dictates the hydrologic function of an urban watershed (Pappas et al. 2008). Relationships have been established that strongly correlate increases in IC with degraded water quality and ecological impacts (Schueler 2009; Fay and Shi 2012).

Low impact development practices (LID) have been implemented to mitigate the environmental impacts from surface runoff by decreasing water volume to stormwater systems, increasing groundwater recharge and increasing the rate of natural contaminant attenuation (Prince George’s County 1999). Permeable pavement is one LID technology which accommodates surface runoff through a permeable surface structure and subsurface reservoir, allowing for high surface infiltration rates, greatly reducing the potential for runoff and increasing groundwater recharge. Studies investigating the potential for groundwater contamination from permeable pavement have shown little impact to groundwater from road surface contaminants, except for chloride (Cl) (Pitt et al. 1999; Schueler 2009; Roseen et al. 2011).
The environmental impacts from road salting are widespread and result in degraded water quality and impaired habitats. As deicing salts are increasingly used during winter months to prevent ice buildup on road surfaces, the risk for Cl contamination of groundwater under permeable pavement becomes a greater concern. The aim of this literature review is to explore the impacts of urbanization on receiving waters. Low impact development practices will be introduced and the impacts of winter deicing will be explored, with a focus on the effects of deicing on groundwater beneath permeable pavements.

**Impervious Surface Impacts on Hydrology**

IC increases in urban areas result in water resource degradation and changes to the local hydrologic cycle (Arnold and Gibbons 1996). This change to the hydrologic cycle is dominated by a transformation from subsurface flow to a dominance of surface runoff (Shuster et al. 2005). As anthropomorphic changes in land use occur, with significant increases in IC, natural flow regimes are altered as infiltration decreases and runoff rates increase. This increase surface runoff results in decreased groundwater recharge, decreased baseflow and low flows, and increases in flood peaks during storm periods (Leopold 1968).

Albrecht et al. (1974) determined that an increase of 20 to 100% IC over a 14 year period resulted in a 50% increase in surface runoff and a 200% reduction in groundwater infiltration. Changes in hydrology including reduced baseflows and increased flood peaks have implications which impact contaminant migration (Arnold and Gibbons 1996), erosion and sediment transport (Pappas et al. 2008), and a loss of biological diversity and suitable biological habitat (Corsi et al. 2010).
Dietz and Clausen (2008) observed that flow rates increased exponentially with increased IC resulting in increases in pollutant loads at a development utilizing traditional stormwater management techniques. An increase from 1% to 32% IC at the Jordan Cove site resulted in an annual runoff increase of 49,000% (Dietz and Clausen 2008).

*Impervious Surface Impacts on Water Quality*

Surface water quality is impaired as a result of increases in IC, due to direct discharges from contaminated surface runoff and contaminated baseflow (Environment Canada 2001). Nonpoint pollution is created as an array of contaminants from many sources flow into surface waterbodies more effectively due to increases in IC and surface runoff rates (Arnold and Gibbons 1996). Water quality is degraded by IC by changing the local hydrologic cycle, serving as a physical area where anthropomorphic sources of contaminants are used, interrupt natural contaminant attenuation in the soil and serve as a direct pathway for contaminants to reach waterbodies (Arnold and Gibbons 1996). Under pervious conditions, contaminants would not be as readily transported by surface runoff and instead would have a greater ability to naturally attenuate as water percolates through soils (Pitt et al. 1999).

Potential nonpoint stormwater pollutants include a variety of inorganic (calcium, chloride, cadmium, copper, magnesium, lead, zinc, nitrogen and phosphorous), organic (polychlorinated biphenyl (PCBs), polycyclic hydrocarbons (PAHs), halogenated aliphatics, monocyclic aromatics, phenols and cresols, phthalate esters), pesticides and micro biological contaminants (Makepeace et al. 1995). The biggest source of these pollutants is road surfaces through vehicle emissions (Stechmann and Dannecker 1990), parts degradation (Pitcher et al. 2004), vehicular fluid leakage and atmospheric deposition (Sansalone and Buchberger 1997; Boving et al. 2008).
road deicing applications and homeowner and municipal pesticide and fertilizer applications (Pitt et al. 1995).

The United States Environment Protection Agency (US EPA) reported that 34,871 miles of river systems were contaminated due to urban runoff and storm sewer systems, accounting for 12% of the sources to impaired river miles (US EPA 2002). An increase in surface runoff leads to increases in the total dissolved content of surface water and decreases dissolved oxygen, resulting in a lesser ability for bacteria to flourish which aid in the natural attenuation of hydrocarbon fuel contamination (Leopold 1968; Pitt et al. 1999).

IC contributes to the contamination of groundwater through surface runoff infiltration. Urban roadway runoff carrying nitrogen and phosphorous sourced from roadside fertilizer application, vehicular exhaust and motor oil has been linked to groundwater contamination in Florida (Pitt, et al 1999). The US EPA (2002) observed that 12% of groundwater contamination is due to urban runoff alone, with fertilization and pesticide applications accounting for an additional 38% collectively. Additional contaminants of concern are deicing salts, including calcium chloride, magnesium chloride and sodium chloride (Environment Canada 2001).

**WINTER DEICING TRENDS**

Deicing agents are commonly used in cold climate regions during winter months to melt and prevent accumulation of ice and snow on road surfaces. The most commonly used deicing agents include sand, calcium chloride (CaCl₂), magnesium chloride (MgCl₂), sodium chloride (NaCl), potassium chloride (KCl) and salt brines (Environment Canada 2001; Sleeper 2013). In the United States, NaCl is the cheapest road salt available and is used most frequently (Sleeper 2013). The quantity of deicing salts used annually in Connecticut has increased in response to increases in urban and suburban development as well as from the Connecticut Department of
Transportation’s (CT DOT) increased reliance on NaCl. In the 2010-2011 winter season alone, the CT DOT reported to have applied 179,000 tons of NaCl and 2,300 tons of CaCl$_2$ across the state, resulting in a high influx of chloride into the environment (Cassanelli and Robbins 2013).

Alternatives to NaCl are available including potassium and calcium-magnesium acetates, calcium and magnesium chloride salts, brines and sand. Sand is no longer is used as a deicing agent on state roads in Connecticut as it requires additional spring cleanup costs and has been found to increase sediment flux to water bodies (The Connecticut Academy of Science and Engineering 2006). Though CaCl$_2$ and MgCl$_2$ salts melt snow and ice more effectively at lower temperatures and require 1/3 less application by mass than NaCl to melt the same amount of snow, these options are used less frequently as their use is cost prohibitive and some still act as potential sources for Cl contamination. While less corrosive than Cl salts, calcium magnesium acetate needs high application rates, is cost prohibitive and has the potential to decrease dissolved oxygen in receiving waters (Kelting and Corey 2010).

Pre-wetting brines are effective at reducing the total amount of deicing applications through a reduction in bouncing, blowing and splashing associated with dry deicing agents. In addition, they cover more area through liquid spreading and speed up the brine formation process. To accomplish this, brines require more advanced machinery and higher levels of training for DOT workers resulting in in higher operational costs (Kelting and Corey 2010).
Winter Deicing Impacts on Water Quality

Cl contamination is primarily the result of deicing salts which dissociate into the anion Cl- and cations (Na+, Mg+, Ca+) when mixed with water in the environment. The Cl- anion is transported by snow melt runoff to surface water bodies and infiltrating groundwater. Areas adjacent to roads, sidewalks, and other high-traffic areas are most susceptible to infiltration by Cl-contaminated water. Depending on soil conditions, runoff rates and Cl concentrations in infiltrating water, Cl can also amass in the adjacent soils acting as a future source for Cl releases (Bastviken et al. 2007).

Cl is not commonly found in natural environments other than from the weathering of certain bedrock and soil sources and atmospheric deposition. The Cl transport rate is not slowed in water or soil, and it moves at the same velocity as water and reacts conservatively in the environment (Environmental Canada 2001; Thunqvist 2003).

Cl levels are most elevated during periods in winter and early spring when deicing agents have been used and snow melt runoff is produced (Brown et al. 2011) as well as during low flow conditions when groundwater is the primary source of stream baseflow (Kaushal et al. 2005). While background concentrations of Cl in undeveloped areas are generally no more than a few milligrams per liter, the average Cl concentrations measured in a study conducted in Montreal found concentrations averaging between 3,000 to 5,000 mg/L for primary and secondary street runoff, with some levels reaching 18,000 mg/L (Environment Canada 2001). In comparison to these elevated levels measured in the thousands milligrams per liter, the Connecticut Department of Energy and Environmental Protection (CT DEEP) has established chronic and acute aquatic toxicity criteria of 230 mg/L and 860 mg/L respectively. The US EPA has established a secondary contaminant level for Cl concentrations above 250 mg/L (US EPA 2002).
Environment Canada (2001) reports that 10-60% of all Cl of anthropomorphic origin infiltrates into groundwater with all of it eventually transported to surface water through groundwater springs. Kelly et al. (2008) estimates that 91% of the salt input to the Wappinger watershed in Dutchess County, New York is from deicing activities, with a 3 fold increase in concentrations between 1986 and 2005, an increase rate of 1.5 mg/L of Cl per year.

Cassanelli and Robbins (2013) found an increase of an order of magnitude in groundwater Cl levels throughout Connecticut over the past century. Cl concentrations reported from 1894 and 1902 across the state ranged from 0 to 5 ppm, with levels of 2.5 to 5 ppm observed along the coastline. These concentrations have increased yearly with a mean observed concentration of 29.1 ppm and maximum concentration of 1500 ppm observed during the 2002 to 2007 sampling period. Cassanelli and Robbins (2013) correlated the mean Cl concentrations from each reporting year with national highway salt sales and calculated a correlation coefficient of 0.90. From these data Cassanelli and Robbins (2013) estimated that by the year 2030 mean Cl concentrations in Connecticut will be 38 ppm. Cassanelli and Robbins (2013) also found higher mean Cl concentrations along roads and developed areas, with concentrations from the 2002 to 2007 sampling period reported at 48.8 ppm and 44.2 ppm respectively.

Baseflow Implications

During low flow periods in summer months, baseflow from shallow groundwater is the primary source of contributing water for gaining streams. Under these flow regimes, streams are at an increased risk to Cl contamination as high levels of Cl in groundwater cannot be diluted out and can pose a greater threat to the ecological health of streams. Higher stream discharges and precipitation events may allow for dilution of Cl during wetter seasons. (Kaushal et al. 2005).
The Minnesota Pollution Control Agency found that under summer baseflow conditions, Eagle Creek consistently showed high concentrations of Cl, due to local Cl contamination of shallow groundwater. The detected Cl levels were reduced by dilution during precipitation events and were highest during low flow summer months (Asleson and Minnesota Pollution Control Agency 2013).

A strong correlation between urbanization and Cl contamination has been observed in both surface water and groundwater (Williams et al. 2000). Williams et al. (2000) found an increase between 21% and 34% in groundwater Cl concentrations in samples collected from groundwater springs over a one year period. Surface water concentrations reached as high as 1390 mg/L while groundwater springs were as high as 1345 mg/L. Seasonal variation was observed in both surface and groundwater, though the groundwater springs exhibited less seasonal variability in comparison to surface water concentrations. Williams et al. (2000) concluded that groundwater springs are more reliable and accurate representations of Cl impacts to aquatic environments, and established a Cl contamination index (CCI) which can be used to estimate Cl concentrations, based on the presence or absence of indicator macro invertebrates.

Corsi et al. (2015) observed that increases in river Cl concentrations occur during all seasons, suggesting that Cl stored in shallow groundwater and released as baseflow to streams is responsible for long term increases. Additionally Cl concentrations were inversely proportional to low flows, suggesting that rainfall and high flows from snow melt dilute baseflow concentrations (Corsi et al. 2015).
Ecological Impacts

As IC increases and contaminant levels in groundwater and surface water reach acute and chronic environmental thresholds, biodiversity, organism survival, development, reproduction and growth rates may be negatively impacted (Sanzo and Hecnar 2005; Siegel 2007; Corsi et al. 2010; Brady 2012).

In 2012, the CT DEEP reported that a total of 165 stream segments out of a total of 527 segments did not support aquatic life as defined by CT DEEP criteria (CT DEEP 2012). CT DEEP estimates that 22% of wadeable streams in Connecticut do not support aquatic life. Land use cover, specifically IC, was the most important factor in determining the aquatic health of wadeable streams. It was found that while 90% of the wadeable streams which do not support aquatic life were in watersheds with greater than 12% IC, none of the segments that fully supported aquatic life were in watersheds with greater than 12% IC (CT DEEP 2012).

Also in Connecticut, Bellucci (2007) found strong correlations between watersheds with 12% or greater IC upstream and decreased macro invertebrate richness (Bellucci 2007). Above 12% IC, biotic integrity, baseflow and maximum number of fish species were found to be low (Wang et al. 2001).

Cl influx from contaminated runoff has significant impacts to organisms at levels above 860 ppm and 230 ppm for acute and chronic levels, respectively (CT DEEP). Effects of deicing salts on ecology is varied (Table 1). Organisms at risk include plants (Siegel 2007), fish, invertebrates (Corsi et al. 2010), insects and amphibians (Brady 2012), with the threshold and nature of the impact dependent on the specific organism, stage of development and the duration and concentration of the Cl exposure (Siegel 2007).
<table>
<thead>
<tr>
<th>Species</th>
<th>Response Type</th>
<th>NaCl (mg/L)</th>
<th>Cl (mg/L)</th>
<th>Response</th>
</tr>
</thead>
<tbody>
<tr>
<td>Frog</td>
<td>7 to 10 day LC50</td>
<td>2540</td>
<td>1524</td>
<td>Sub-chronic survival</td>
</tr>
<tr>
<td>Rainbow Trout</td>
<td>4 day LC50</td>
<td>11,112</td>
<td>6,743</td>
<td>Acute survival</td>
</tr>
<tr>
<td></td>
<td>6 hour LC40</td>
<td>20,000</td>
<td>12,312</td>
<td>Acute survival</td>
</tr>
<tr>
<td>Brook Trout</td>
<td>15 minute LC50</td>
<td>50,000</td>
<td>30,330</td>
<td>Acute Survival</td>
</tr>
<tr>
<td>Snail</td>
<td>4 day LC50</td>
<td>4088</td>
<td>2480</td>
<td>Acute Survival</td>
</tr>
<tr>
<td>American Eel</td>
<td>4 day LC50</td>
<td>17,964-21,571</td>
<td>10,900-13,085</td>
<td>Acute Survival</td>
</tr>
<tr>
<td>Caddisfly</td>
<td>4 day LC50</td>
<td>5526-7014</td>
<td>4039-4255</td>
<td>Acute Survival</td>
</tr>
</tbody>
</table>

Brady (2012) investigated salamander embryonic development in roadside and woodland pools in Connecticut, in which variation in specific conductivity was the only major variable. Considerable differences in embryo development were observed between salamander pool populations (Brady 2012). Specific conductivity and Cl were 886 microsiemens/cm and 188 mg/L in roadside pools and 28 microsiemens/cm and 2.8 mg/L in woodland pools. The impaired water quality in the roadside pool as indicated by the high specific conductivity suggests impacts related to deicing activities and subsequent snowmelt runoff into the pools (Brady 2012). Results from the study showed embryos from roadside pools were 11.3% smaller on average than those from woodland pools. The survival rate of salamander was only 56% in roadside pools compared with 87% in woodland pools, and samples raised in roadside pools grew 5.5% slower on average. Brady found local adaptation in the roadside pools evidenced by the fact that when groups of embryos were placed in different pools from which they were born, the roadside population out-survived the woodland population in roadside pools by 25%, with no advantage.
observed in both populations in the woodland pool. These results suggest a relationship between increases in Cl concentrations with negative impacts to survival and development rates, as well as suggest natural selection as an important evolutionary driving force (Brady 2012).

Additional studies investigating the effects of increasing Cl concentrations in aquatic environments suggest similar impacts to aquatic organisms. Karraker and Gibbs (2010) found that for Cl concentrations between 145 mg/L and 954 mg/L, salamander egg clutches in Rhode Island lost 4-41 % mass, indicating a loss of water intake which increases embryo risk to freezing, predation and disease.

The effects of Cl contamination on fish populations depend on the species, with thresholds for Cl concentrations ranging from 400 mg/L to 30,000 mg/L. At certain levels of Cl influx, salt tolerant fish species may become a dominant species (Siegel 2007). Corsi et al. (2010) found that at chronic Cl exposure levels of 2,920 mg/L or greater, fathead minnows exhibited reduced weight and survival rates in Milwaukee, Wisconsin.

Corsi et al. (2010) also investigated chronic levels of Cl exposure to invertebrates. Initial chronic toxic effects were observed in C. dubia (water flea) between Cl concentrations of 600 and 1100 mg/L. Reproduction was nonexistent at chronic Cl concentrations of 1770 mg/L, with 100% mortality observed at concentrations exceeding 2420 mg/L (Corsi et al. 2010).

Vegetation, including roadside and aquatic plants, is affected by disruptions in osmotic balance in response to increases in Cl concentrations. This imbalance can interfere with a plant’s ability to absorb water, and can stunt root growth, inhibit nutrient uptake and long-term growth, flowering, seed germination and growth of roots and stems (Siegel 2007).

Additional ecological effects from the influx of salt water resulting from deicing runoff include prevention of vertical mixing in lakes and ponds. Prevention of mixing results in anoxic
conditions in deep water and promotes the release of phosphorous from sediments. Eutrophication develops from the influx of phosphorous and hypoxic conditions can result (Environment Canada 2001).

*Cation Exchange and Metals Mobilization*

The relative ability of soils to store cations due to the presence of negatively charged soil surfaces is defined as cation exchange capacity. When cations are introduced into soil water, the cations dissolved in infiltrating water can exchange with the cations in the soil during a process called cation exchange (CE) (Mengel). The presence of additional cations in percolating soil water, mainly sodium from the NaCl used in deicing activities, can exchange with cations in the soil and mobilize trace metals. Major metals of concern for mobilization include cadmium, mercury, copper, lead and zinc (McLean et al. 1992 Environmental Canada 2001; Bäckström et al. 2003) which can further degrade water quality and alter ecological health (Fay and Shi 2012).

**LOW IMPACT DEVELOPMENT**

*Overview*

Low impact development (LID) techniques have been developed in response to the environmental degradation associated with traditional stormwater technologies such as catch basins and detention pools (Prince George’s County 1999). While traditional stormwater management practices primarily aim to reduce peak flows, LID practices (e.g. cluster layouts, grass swales, rain gardens, bioretention areas, and pervious pavements) all reduce total IC and allow for contaminant attenuation, reduced flow volumes and reduced peak flows (Prince George’s County 1999). LID management practices are designed to influence the hydrology of
an area to be as similar to predevelopment conditions as possible, including contaminant mobilization concentrations and surface runoff rates (Prince George’s County 1999).

Permeable Pavement Impacts on Hydrology

Permeable pavement systems are structural surfaces that are designed with high surface permeability and a subsurface water storage reservoir. Permeable pavement can include permeable cement, block plastic grids, permeable interlocking concrete pavers and permeable asphalt. While slightly different in material and design, all of these systems are implemented to reduce the total discharge volume and peak flow of storm events. A longer lag time between the precipitation event and the first instance of water discharged as shallow groundwater from subpavement reservoirs (typically 25-50% into the storm event) promotes more natural hydrologic conditions. Peak flows are reduced and infiltration periods extended beyond the extent of the storm (Pratt et al. 1995). Runoff was virtually non-existent on permeable pavement systems researched by Brattebo and Booth (2003), indicating higher levels of groundwater recharge in comparison to the high runoff volumes observed on traditional asphalt surfaces.

In a study comparing impacts to water quality in traditional and LID subdivisions in Connecticut, Dietz and Clausen (2008) found that flow regimes and total nitrogen and phosphorous export at the LID site were more similar to a forested watershed than an urban development. Despite an increase from 0% to 21% IC, the influence of LID techniques resulted in no increase in runoff or pollutant export for most pollutants measured (Dietz and Clausen 2008).
Permeable Pavement Impacts on Water Quality

Generally, hydrocarbon (Pitt et al. 1995), metals (Brattebo and Booth 2003; Welker et al. 2012) pesticides (Pitt et al. 1999) and phosphorous (Collins et al. 2007) concentrations in groundwater beneath permeable pavement have been low in comparison to surface runoff. Nitrate (NO$_3^-$) and nitrite (NO$_2^-$) concentrations increased depending on site geology (Boving et al. 2008) and permeable pavement design (Collins et al. 2007) though total nitrogen concentrations were equal to or less than background conditions (Kwiatkowski et al. 2007). Cl concentrations have been observed to fluctuate in shallow groundwater under permeable pavement as a function of season (Pratt et al. 1995). Cl peaks in groundwater have been observed in winter months (Welker et al. 2012) due to deicing activities and diluted below harmful levels during other seasons (Kwiatkowski et al. 2007).

Kwiatkowski et al. (2007) observed that Cl levels in groundwater beneath permeable concrete peaked during winter months where CaCl$_2$ was applied as a deicing agent. During winter months, Cl concentrations up-gradient of the permeable concrete measured 100 mg/L while concentrations beneath the permeable concrete reached as high as 1000 mg/L. In the months following the winter, rainwater infiltrating the permeable cement diluted Cl concentrations to under 10 mg/L (Kwiatkowski et al. 2007).

Brattebo and Booth (2003) compared percolate water quality beneath four different permeable pavement systems with surface runoff from traditional asphalt. Permeable pavement systems included two plastic interlocking grid systems and two concrete block systems. The infiltrating water beneath the permeable systems had lower copper and zinc levels, and no motor oil detected relative to runoff from traditional asphalt which had motor oil detections and metal concentrations above Washington state surface water quality standards.
Copper concentrations in all samples of surface runoff were above Washington state chronic and acute toxic levels, with zinc concentrations above both chronic and acute in all but one sample. Beneath the permeable pavement systems 72% of copper and 22% of zinc were below minimum detection limits. For copper, only one sample exceeded chronic criteria and for zinc one sample exceeded acute criteria and three at chronic criteria. Copper concentrations beneath two of the permeable pavement systems decreased over time. Increases in electrical conductivity (EC) and hardness were observed in the infiltrating water relative to the traditional asphalt surface runoff. Zinc levels beneath the permeable pavement were less than the surface asphalt runoff concentrations, though zinc concentrations did increase beneath the permeable pavement systems over the length of the six year study (Brattebo and Booth 2003).

Pratt (1995) found reductions in hardness, chloride, alkalinity, lead and suspended solids in effluent water beneath four permeable concrete block systems over the course of a six month monitoring study. Collins (2007) found that total nitrogen and ammonium (NH₄⁺) concentrations were lower beneath permeable pavement systems. Nitrogen as NO₃⁻ and NO₂⁻ were higher in percolate in three out of four pavement systems due to the nitrification of ammonium. Samples from the concrete grid pavers filled with sand had the lowest observed total nitrogen and NO₃⁻ concentrations due to nitrification and assimilation of NH₄⁺(Collins et al. 2007). Kwiatkoswsi et al. (2007) terminated testing for nitrogen as sampling sites up-gradient of the permeable pavement had higher concentrations than sampling sites beneath it. Welker (2012) agreed with observations by Kwiatkoswki et al., that the concentrations of nitrogen up-gradient of permeable pavement can be equal to or higher than concentrations beneath it.
PERMEABLE ASPHALT

Overview

Permeable asphalts (PA), also referred to as open graded asphalt, are surfaces consisting of coarse aggregates bonded with bituminous asphalt that has had fines removed to allow for greater porosity. Different porosities can be achieved through greater void space in the bituminous asphalt as well as the incorporation of different ratios of coarse to fine grain sizes. The minimum porosity of PA systems should be 16%, which allows for increases in groundwater recharge and decreased surface runoff and pollutant loading in comparison to traditional asphalt surfaces (http://water.epa.gov/polwaste/npdes/swbmp/Porous-Asphalt-Pavement.cfm).

Previous studies have investigated the retention of precipitation on PA. Retention values range from between 25% in PA installed above clay soil (Legret et al. 1996) to 100% in more permeable conditions (UNHSC 2012).

Design

A typical PA installation consists of 2-4 inches of PA, depending on the expected vehicular traffic and use, followed by a 1-2 inch thick choker course (Figure 1). The choker course acts to stabilize and level the asphalt surface and is comprised of ¾ inch or smaller aggregate. Below the choker course is 18-36 inches of an open coarse aggregate base and sub-base. The base is typically 3-4 inches thick and made of crushed 3/4 inch to 3/16 inch stones. The sub-base is made of larger stones than the base, typically 3/4 inch to 2 1/2 inch stones. These two layers act as a water reservoir for the PA system. Often, a sub-drain, consisting of perforated pipes that feed into a nearby storm water system, is installed in the sub-base layer to accommodate flows that exceed the infiltration capacity of low permeability sub grade soils.
Beneath the sub-base, a geotextile is often used to prevent soil migration from the sub grade soils into the sub-base open course aggregate, although these fabrics have been found to clog, causing premature failures (Roseen et al. 2011).

![Cross section of a typical permeable asphalt installation.](http://water.epa.gov/polwaste/npdes/swbmp/Porous-Asphalt-Pavement.cfm)

The rate of vehicular usage, local climate and native soil type are most important in designing PA systems. To accommodate heavy traffic or total loads on the PA, extra thickness of PA must be included in the design as well as the addition of polymers which can be used in the PA mix to strengthen the surface. The sub-base reservoir has to be large enough to store average precipitation volumes for that climate and provide storage for soils with low infiltration. Extra sub-base thickness is needed with subgrade clay soils not only because of low infiltration rates, but also to accommodate the low structural support of clay soils. Impermeable liners may be used beneath sub-base reservoirs in locations with shrink swell clay subgrade soils or high water table (Collins et al. 2008).

**Maintenance Best Practices**

The National Asphalt Pavement Association (NAPA) recommends not applying sand as a winter deicing and traction agent to permeable asphalt surfaces, as the pores can easily clog,
reducing infiltration rates. Vacuuming is recommended at least twice a year, as well as pressure washing high traffic areas. Deicing agents are suggested for winter snow and ice removal, and Houle (2008) showed an average reduction in required salt application by as high as 75% compared to traditional asphalt. Due to the high infiltration rate of PA, fewer water pools on the surface result in less black ice formation, which allows for quicker snow and ice melting rates. Plowing permeable asphalt is permissible, and no special blade or other measures need to be taken (UNHSC 2012).

**Hydrology**

Roseen et al. (2011) reported that infiltration rates in New Hampshire were vary by location within the PA and potentially by season, with mean infiltration rates ranging from 1490 cm/h to 2690 cm/h. A net groundwater recharge area of 299 m$^3$ over an 18 month period of study was observed. During the wettest months of the study (May, June and October), percolating water from precipitation exceeded effluent from the underdrain of the PA indicating a net recharge of groundwater. No runoff was observed during the study, with event precipitation as high as 12.1 cm. Roseen et al. (2011) observed a 90% reduced peak flow, increased lag time and increased lag coefficient which are critical features of predevelopment conditions and reflective of successful LID strategies. The reductions in peak flow observed by Roseen et al. (2011) is supported by stormwater management models by Hogland et al. (1987), which suggest an 80% reduction in peak flow when PA is used.
Water Quality Benefits

Similar to other permeable pavement systems, PA has been shown to reduce concentrations of petroleum hydrocarbons, metals (zinc) and total suspended solids, yet has not been shown to mitigate NO$_3^-$ or Cl concentrations, with mixed results for phosphorous treatment (Legret and Colandini 1999; Pagotto et al. 2000; Boving et al. 2008; Roseen et al. 2011). At the University of Rhode Island, Boving et al. (2008) investigated the efficiency of a PA parking lot averaging 722 cars per day in mitigating infiltrating water. NO$_3^-$ concentrations in percolate were typical for urban runoff and typically below drinking water criteria of 10 mg/L. The highest nutrient concentrations were detected during spring and fall due to fertilizer dust generated by nearby agricultural activities. NO$_3^-$ and phosphate (PO$_4^{3-}$) concentrations reached as high as 1.5 mg/L for both nutrients. The PA system was shown through tracer testing to eliminate 27% of nutrients from percolating water, though mass flux of PO$_4^{3-}$ in percolate was observed to be 5 times higher than conventional asphalt runoff, potentially due to fertilizer dust deposition on the lot from an adjacent farm (Boving et al. 2008). This increase in PO$_4^{3-}$ is contrary to findings by Roseen et al. (2011), who observed a 43% reduction in phosphorous in percolate beneath PA.

Cl concentrations were highest from January through March, and decreased from a high concentration of 930 mg/L to below detection limit. Zinc and copper concentrations remained below US EPA limits of 5 mg/L and 1.3 mg/L, respectively, and tracer tests showed that the PA system was capable of removing greater than 90% of zinc and copper from infiltrating water. Metals concentrations were highest during winter and early spring, possibly due to an increase in automobile part corrosion from deicing salts (Boving et al. 2008). These results are similar to
those found by Legret et al. (1996) in which lead, zinc and cadmium were reduced by 79%, 72% and 67%, respectively.

Polycyclic Aromatic Hydrocarbon (PAH) concentrations were observed near the detection limit of 0.3 ug/L, while the highest concentration was 4.9 ug/L. Concentrations in nearby surface runoff from a traditional parking lot were measured at 1.4 ug/L. PAH flux analyses showed that the mass flux of PAH through the PA parking lot was an order of magnitude lower than fluxes from conventional surfaces, suggesting that PA systems slow or eliminate PAHs from infiltrating water (Boving et al. 2008).

Pagotto et al. (2000) found that replacing a traditional asphalt highway with 30 mm of porous asphalt underlain by an impervious surface resulted in a reduction of hydrocarbons by 92%, total dissolved solids by 81%, copper 35% and lead 78%, with hydrocarbon and heavy metal concentrations below France’s drinking water standard. The differences between Cl concentrations in surface runoff from traditional asphalt and from percolate beneath the PA were not significantly different (P<0.05), suggesting no attenuation of Cl (Pagotto et al. 2000).

Legret and Colandini (1999) observed that water quality beneath PA is significantly improved with respect to heavy metals (p=0.05) in comparison to stormwater runoff. The primary location of heavy metal reduction occurred in the surface porous asphalt layer where sediment is trapped and clogging occurs. Soils under the PA were not impacted at the end of the seven year study period, as concentrations were comparable to reference soils in close proximity to the site (Legret and Colandini 1999).
CONCLUSION

As LID practices continue to be implemented in urban areas to mitigate negative environmental impacts of urbanization to water resources, proper investigations must be completed to assure that LID does not bring with it environmental issues of its own. While studies have shown that metals and hydrocarbon contaminant concentrations in percolating water decline under PA and can mirror predevelopment conditions, broader impacts on groundwater remain unclear. Studies show that Cl and NO$_3^-$ concentrations are not filtered by percolation through PA. The presence of nitrogen should not be of concern for PA, as there is not a significant source of nitrogen applied to PA surfaces. Among the potential contaminants, only Cl has been shown to significantly impact groundwater quality beneath permeable pavement (Roseen et al. 2011). As deicing salts are increasingly used during winter months to prevent ice buildup on road surfaces, the risk for groundwater contamination from Cl under permeable pavement becomes a greater concern.

Elevated Cl concentrations beneath permeable pavement have been observed in many studies, though interestingly, concentrations have been observed to be diluted in wet non-winter months. Despite dilution, little data exist to affirm or negate the claim that Cl poses a long-term threat to groundwater quality beneath permeable surfaces, with very few studies evaluating long-term deicing impacts to Cl concentration beneath permeable asphalt.

The objective of this research is to evaluate the potential for Cl contamination of groundwater beneath permeable asphalt due to winter deicing salt applications.
LITERATURE CITED


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INTRODUCTION

As landscapes change from rural to urban, impervious cover (IC), such as roads, rooftops and sidewalks increase in area. From 1985 to 2010, developed areas in Connecticut increased by 386 km$^2$ (CLEAR 2010). These impervious surfaces disrupt the hydrology (Arnold and Gibbons 1996) as precipitation is prevented from infiltrating into the ground and instead runs off (Shuster et al. 2005; Pappas et al. 2008). Traditional stormwater management systems such as detention ponds and catch basins have been used to mitigate surface runoff, but studies have shown that these systems can increase sediment loads and contaminant migration from roads to surface water bodies, further degrading water quality (Prince George’s County 1999). Relationships have been established that strongly correlate IC of 12% and higher with degraded water quality and negative ecological impacts (Schueler 2009; Fay and Shi 2012). Of the wadeable streams that did not support aquatic life in Connecticut, 90% had watershed with IC above 12% (CT DEEP 2012).

Low impact development practices (LID) were developed as an alternative to traditional stormwater management methods, with the goal of preserving predevelopment hydrologic conditions (Prince George’s County 1999). Reductions in peak flows and contaminant migration are achieved by increasing the potential for natural attenuation of contaminants by means of groundwater recharge (Prince George’s County 1999).

Permeable asphalt (PA) is one LID technology which accommodates surface runoff through a permeable structure and subsurface reservoir. This allows for high surface infiltration rates, greatly reducing the potential for runoff, increasing groundwater recharge and the natural
attenuation of contaminants (Pitt et al. 1999). A significant amount of contaminant filtration occurs in the top layer of PA, as sediment clogs in the pores as infiltrating water passes through the voids (Legret and Colandini 1999). Studies investigating the potential for groundwater contamination from PA have shown little impact to groundwater from road surface contaminants, except for chloride (Cl) (Pitt et al. 1999; Schueler 2009; Roseen et al. 2011).

Average annual road salt sales in the United States have increased from 0.28 million metric tons in the 1940’s to 16 million metric tons in 2008 (Corsi et al. 2010). As NaCl is increasingly applied to road surfaces during winter months to combat snow and ice, Cl is released into the environment at a large scale. Cassanelli and Robbins (2013) found an increase of an order of magnitude in groundwater Cl levels throughout Connecticut over the past century (Cassanelli and Robbins 2013). The environmental impacts include groundwater and surface water Cl and sodium (Na) contamination (Williams et al. 2000; Environment Canada 2001; Bastviken et al. 2007; Cassanelli and Robbins 2013), aquatic toxicity (Williams et al. 2000; Sanzo and Hecnar 2005; Bellucci 2007; Siegel 2007; Corsi et al. 2010) and heavy metal mobilization through cation exchange (McLean et al. 1992; Environment Canada 2001; Norrstrom and Jacks 1998; Bäckström et al. 2003; Norrstrom 2005).

Though studies have noted seasonal trends with groundwater Cl concentrations beneath PA increasing in winter months (Roseen et al. 2011) and decreasing below detectable levels in early summer (Boving et al. 2008), a quantitative analysis investigating the impact and magnitude of Cl contamination in groundwater beneath PA in relation to surrounding water quality conditions has not been thoroughly explored. The objective of this study was to evaluate the potential for Cl contamination of groundwater beneath permeable asphalt due to winter deicing salt applications.
METHODS

Study Site

Overview

Figure 1. Study site at the University of Connecticut in Storrs, CT.

The study site is located on the University of Connecticut campus in Storrs, CT. The site includes an 860-m² PA parking lot which can accommodate 27 vehicles (Figure 2). Groundwater under the PA flows in a north, north easterly direction with an average water table slope of 2.7 % depending on changes in water table elevations throughout the study. The depth to groundwater has been observed between 3-m to 4.6-m below grade, depending on location and season.

On average, the PA receives 125 cm of precipitation annually (NOAA-NCDC 2015). During winter months (November through March) the PA is plowed and salted with NaCl salt using traditional deicing and plowing methods. The soil under the PA parking lot is heavily disturbed and includes native glacial till and silty sand fill material with an average soil infiltration rate of 0.76 cm/hr (BVH Integrated Services, 2010) and an average hydraulic conductivity of 2.63 E-4 cm/s (Appendix B).
Permeable Asphalt

The PA was installed in October of 2010. It is composed of 4 layers averaging 71 cm thick throughout the site (Figure 3). The top layer is comprised of 10 cm of PG64-22 PA mix with 16 – 22 % air void content. Below the PA is a 10-cm thick choker course, a 36 cm filter course followed by an additional 15-cm thick reservoir course. A 15-cm perforated underdrain located a minimum of 5-cm from the bottom of the reservoir was installed to remove water that percolated more quickly than it could infiltrate into subgrade soils. The underdrain was tied into a nearby stormwater system. A geotextile was installed along the sides of the PA, extending to the top of the choker course, to prevent soil migration into the reservoir. Sub-base materials by percent finer classification are shown in Table 1. The PA surface gradient varies throughout the parking lot but averages 3.5% in a north-west direction. Single ring infiltrometer tests on the PA showed average (n=4) infiltration rates of 211 cm/hour in September 2013 and 129 cm/hr in May 2015.

Figure 2. Permeable asphalt parking lot and monitoring well locations on the University of Connecticut campus. The arrow indicates the average groundwater flow direction.
Figure 3. Cross section of permeable asphalt design.

Table 1. Grain size analyses of permeable asphalt sub-base (BVH Integrated Services, 2015).

<table>
<thead>
<tr>
<th>Percent Finer Than Grain Size</th>
<th>Pea Gravel 9.51 mm</th>
<th>38.1 mm</th>
<th>25.4 mm</th>
<th>12.7 mm</th>
<th>No. 4</th>
<th>No.8</th>
</tr>
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<tr>
<td>Choker Course</td>
<td>-</td>
<td>100</td>
<td>.</td>
<td>25-60</td>
<td>0-10</td>
<td>0-5</td>
</tr>
<tr>
<td>Filter Course</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>70-100</td>
<td>0-6</td>
</tr>
<tr>
<td>Reservoir</td>
<td>100</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td></td>
</tr>
</tbody>
</table>
**Monitoring Wells**

Three 2.54 cm overburden monitoring wells, Down-gradient 1, Down-gradient 2 and Up-gradient, have been installed around the perimeter of the site (Figure 2). All wells are designed with screen sections installed from 3.66-m to 5.18-m below grade, with well depth terminating at the bottom of the screen. Down-gradient 2 and up-gradient have a 1.5-m Geoprobe® prepack screen while down-gradient 1 has a 1.5-m PVC slotted screen surrounded by #2 New Jersey sand. All wells are installed in a silty sand fill material, with Down-gradient 1 exhibiting a large 1.29-m section of pea gravel from 2.44-m to 3.73-m below grade.

**Hydrology**

Hydraulic permeability was calculated for up-gradient and down-gradient 2 using a single well steady-state constant head pumping test. An INW-CT2X EC probe and pump inlet tubing were lowered to the bottom of the well. The pump was turned on to a specific discharge rate and the observed drawdown in the well was measured. This protocol was conducted 3 times at 3 different discharge rates for Down-gradient 2 and Up-gradient. Hydraulic permeability was calculated using the Hvorslev Half Ellipsoid steady state equation (Robbins et al. 2009).

\[
K = \left( \frac{slope \times 2.303 \log \left[ \frac{L}{D} \sqrt{T} + \left( \frac{L}{D} \right)^2 \right]}{2\pi L} \right)
\]

\(slope=\)Discharge/Drawdown, \(L=\)saturated length and \(D=\)diameter of influence.

(Eq. 1)

To assess hydraulic connectivity of the wells to the PA, a Cl tracer test was conducted prior to the start of deicing activities. Sixty-eight kg of NaCl salt were applied to the PA surface before a rain event on November 6, 2014. EC probes in the wells collected data every 10
minutes to monitor for increases in EC related to the dissolved Cl from the surface salt application.

**Water Quality Sampling and Analysis**

*Chloride*

Data for this study were collected between September 26, 2014 and May 8, 2015. Each well was equipped with an INW-CT2X electrical conductivity (EC) probe, located within the screen section of each well. The EC probes were programmed to collect readings at 10 minute intervals, including EC, temperature and pressure. Biweekly low flow sampling was conducted using a Geoprobe Geo Pump Two ®. Each well was pumped for 10 minutes at rates under 120 ml/minute. Samples were collected and kept refrigerated at 4 °C with a maximum holding time of 28 days from collection date. Cl was measured using an ion-specific electrode (ISE) using EPA Method 9212 (US EPA 1996).

A regression between EC and Cl was developed, allowing EC to act as a proxy to estimate continuous Cl concentrations. Cl concentrations, temperature and water table elevation were plotted against time and analyzed for changes in magnitude seasonally and spatially through the site.

*Metals*

Na, magnesium (Mg) and calcium (Ca) metals testing was completed on 4 occasions, 2 prior to deicing activities began and 2 after. The metals samples were collected using the same low flow sampling protocol as the Cl samples. The samples were then sent to the Center for Science and Engineering (CESE) at the University of Connecticut and prepared using EPA

Statistical Analyses

Laboratory Cl data were analyzed using Analysis of Variance (ANOVA) to determine if there were significant differences in Cl concentrations among up-gradient and down-gradient wells. Mean separation was performed using Tukeys’s test, and p<0.05 was used as the criterion for significance.

NaCl Application Rates

Total Cl applied to the PA parking lot was estimated by placing two 37 m² collection tarps on the PA before a salting truck applied NaCl using typical salting methods. The salt collected on each tarp was weighed and multiplied by a correction factor (area of PA divide by area of tarps) to obtain an estimated application rate per salting event. The total NaCl load to the PA was calculated based on this average application rate and an estimate of 3 NaCl applications per storm. Cl mass was found based on the ratio of Na to Cl masses (equation 2).

\[
\text{Total Kg Cl} = \text{PATotal Kg NaCl} \times 0.6067 \text{ Kg Cl/Kg NaCl}
\]

(Eq. 2)
RESULTS AND DISCUSSION

Site Hydrology

The salt tracer test conducted on November 6, 2014 during a 0.76-cm precipitation event (NOAA-NCDC 2015) established hydraulic connectivity between Down-gradient well 1 and the PA reservoir. Under non-salting conditions, changes in water table elevations (WTE) are inversely proportional to changes in EC due to dilution of groundwater by percolating water. This relationship is observed during a precipitation event on November 2nd. The increase in WTE observed in response to percolating water correlates to a decrease in EC. This relationship becomes directly proportional when percolating water with higher EC from dissolved deicing salt reaches the water table. During the tracer test, increases in water table elevation in Down-gradient 1 were directly proportional to increases in EC (Figure 4a), showing that dissolved Cl from the surface salt application had reached groundwater in the well. Increases in WTE in the Up-gradient well are consistently inversely proportional to changes in EC throughout the storm, showing no hydraulic relationship from the PA reservoir to Up-gradient (Figure 4b). Down-gradient 2 did not have an EC probe installed at the time of the test. Based on water flow direction calculations, Down-gradient well 2 is not hydraulically connected to the PA reservoir.

A constant head permeability test was conducted on May 8, 2015 in Down-gradient 2 and Up-gradient, and an average hydraulic conductivity of 2.63 E-4 cm/s was calculated for the site (Appendix B).
Figure 4. Salt tracer experiment. A (Down-gradient 1), B (Up-gradient). The black star indicates the rain event on 11/6 after NaCl application on 11/5.

**Water Table Elevations**

WTE in all wells fluctuated throughout the study and was dependent on type and rates of precipitation from storms as well as surface and soil temperatures (Figure 5). In September through mid-December when the ground had not yet frozen, WTE increased in response to precipitation events. High peak WTE in Down-gradient 1 at observed during October and November suggest that the groundwater flow direction may reverse in direction during large
precipitation events. Due to a very dry beginning to the 2015 water year, major increases in WTEs were not observed until the first precipitation events in November. WTE increased until the ground began to freeze and then generally decreased or fluctuated around a fixed elevation until late winter. In early March, as air temperature rose and the ground thawed, the snow pack began to melt and large increases in WTE were observed. Between March 4 and March 27, 43 cm of snow melted in addition to precipitation events, resulting in large increases in WTE in all wells. As the spring melt concluded and precipitation diminished in late April and May, WTE decreased.

Figure 5. Water table elevations for Down-gradient 1, Down-gradient 2 and Up-gradient and precipitation during the monitoring period of 9/26/2014 through 5/8/2015.

Temperature

Temperature in all wells generally decreased over the period of the study (Figure 6). Down-gradient 1 and Down-gradient 2 wells are near university steam lines, and groundwater in close proximity to the wells is much warmer than up-gradient. Changes in temperature were found to be inversely proportional to changes in WTE in Down-gradient 1 throughout much of
the study period (Figure 7) and during large spring storms in Down-gradient 2 (March 2015 in Figure 6). The inverse relationship between WTE and temperature is dependent on the relative difference in temperature between the percolating water and the heated groundwater.

Fluctuations in temperature are more frequent in Down-gradient 1 due to the high surface infiltration rate of the PA and hydraulic permeability of the subsurface reservoir and surrounding soil, which allows percolating water to reach the water table faster than in the other wells.

![Graph showing groundwater temperature](image)

Figure 6. Groundwater temperature for Down-gradient 1, Down-gradient 2 and Up-gradient during the monitoring period of 9/26/2014 through 5/8/2015.
Regression Analysis

Biweekly Cl concentrations (n=32) were compared to EC readings from the same period of time (Figure 8). This relationship was then used to create continuous (10 minute average) Cl concentrations from EC measurements.

Figure 8. Regression of Cl and electrical conductivity based on 32 pairs of measured chloride and electrical conductivity concentrations. EC = electrical conductivity.
Salt Application Rates

Representative salt application sampling by tarp collection method results are shown in Appendix C. An average rate of 0.077 kg/m² was calculated. The University of Connecticut averaged 3 NaCl salt applications per storm and salting loads were based on 14 major storms for which the lot was salted. An estimated 2,623 kg of NaCl was applied to the PA during the period of study. Based on the amount of surface runoff, non-uniform spreading of salt during deicing applications, the total number of salt applications per storm and potential anisotropies in groundwater Cl concentrations in the aquifer would contribute to a varied response in the changes in Cl concentrations detected per storm beneath the PA.

Water Quality

EC and Chloride

EC (Figure 9) and Cl (Figure 10) concentrations generally increased in all wells over the period of study in response to changes in WTE and Cl concentration of percolating water. Cl vs. WTE for each well is presented in Figure 11. As observed during the salt tracer test (Figure 4), changes in EC and Cl are directly proportional to changes in WTE during non-deicing periods, and become inversely proportional to WTE during periods of infiltration of dissolved deicing salts. Roseen et al. (2011) observed this trend in effluent from a PA reservoir, which exhibited increases in specific conductance during storms events and decreases following the storm period.
Figure 9. EC concentrations in Down-gradient 1, Down-gradient 2 and Up-gradient during the monitoring period 9/26/2014 through 5/8/2015. Missing chloride data for Down-gradient 2 from 10/3/2014 through 11/6/2014 is due to no EC probe installed in the well during that time period.

Figure 10. Chloride concentrations in Down-gradient 1, Down-gradient 2 and Up-gradient during the monitoring period 9/26/2014 through 5/8/2015. Missing chloride data for Down-gradient 2 from 10/3/2014 through 11/6/2014 is due to no EC probe installed in the well during that time period.

Cl concentrations in Down-gradient 2 rose above the CT DEEP chronic aquatic toxicity limit of 230 mg/L and Up-gradient concentrations were observed above both the CT DEEP chronic and acute aquatic toxicity of 230 mg/L and 860 mg/L, respectively (CT DEEP 2011). Corsi et al. (2010) observed increases in Cl concentrations of spring fed rivers even after deicing.
applications ceased, suggesting that the impact from contaminated baseflow can affect water quality well beyond the time of deicing application. Concentrations in Down-gradient 1 are below both chronic and acute CT DEEP criteria, suggesting dilution from infiltration of rain water with low concentrations of Cl.

Cl Concentrations observed from September through early October are representative of Cl concentrations that exist during summer months when baseflow is often the major source of water for spring-fed surface water bodies. At these times, spring-fed water bodies are at the greatest risk of contamination by contaminated groundwater as there is less water available to dilute high concentrations (Kaushal et al. 2005; Asleson and Minnesota Pollution Control Agency 2013).

Precipitation increased in October and November 2014, resulting in WTE increases in all wells and decreases in Cl concentrations, presumably by dilution. On November 26, the first significant application of deicing salts occurred and is visible by increases in Cl concentration and WTE in each well (Figures 11). Cl concentrations increased through December 2014 in all wells as deicing salts were applied and transported as dissolved ions in snow melt to the water table. Down-gradient 2 and Up-gradient increased in concentration until late December when the ground began to freeze and infiltration slowed. Increases in Cl concentrations occur slightly further into December in Down-gradient 2 than Up-gradient as the ground is heated by steam pipes which allows the ground to stay unfrozen longer. Up-gradient and Down-gradient 2 experienced peak winter Cl concentrations of 3,991 mg/L and 467 mg/L respectively. Increases in Cl concentrations Up-gradient are most likely due to salting of sidewalks in close proximity to the well.
Down-gradient 1 concentrations varied through December and January, with large fluctuations in Cl concentrations occurring further into the winter than the other wells due to the higher surface infiltration rates of the PA in cold temperatures relative to native soils. A peak winter Cl concentration of 5,060 mg/L was observed on January 25, 2015 before the Cl concentrations in Down-gradient 1 began to decline. During this plume migration Cl concentrations increased by 3,773 mg/L in 24 hours, suggesting high rates of groundwater flow. In all wells, Cl concentrations decreased with time through much of the winter, despite occasional peaks associated with percolating water that contained high concentrations of deicing salts.

As surface and ground temperatures increased in early spring, the snow pack began to melt and infiltration rates increased; Large increases in Cl concentrations and WTE in all wells were observed. The fluctuations in Cl concentrations with respect to timing of NaCl applications and snowmelt events is consistent with Roseen et al. (2011) and Pagotto et al. (2000) who observed that Cl showed no removal by filtration through the PA reservoir. Cl concentrations in Down-Gradient 1 and Up-gradient reached maximum spring levels on March 21, 2015 at 5,808 mg/L and 2,925 mg/L, respectively. For Down-gradient 1 this concentration represented a 20,208 % increase from concentrations in September. Concentrations after this peak decline through mid-April as deicing applications stop and percolating water dilutes groundwater.

Increases in Cl Concentrations in all wells from late April 2015 to the end of the study correlate with decreases in WTE. Up-gradient Cl concentrations rose to 5,472 mg/L, an increase of 536% from the beginning of the study. Down-gradient 1 Cl concentrations rose to 2,740 mg/L, an increase of 9,480 %. Cl concentrations in Down-gradient 2 increase to 831 mg/L, an increase of 68%. Hogland et al. (1987) observed similar trends in Cl increases from snowmelt
under PA, with a 650% increase in Cl observed. The increase in Cl concentrations in all wells at the end of the study contradicts findings from Boving et al. (2011), who observed a reduction of Cl concentrations by 97.5% (930 mg/L in late February to 23 mg/L in late May).
Figure 11. Water table elevations and chloride concentrations from 9/26/2014 through 5/8/2015 
Up-gradient (A), Down-gradient 2 (B) and Down-gradient 1 (C). 
Missing chloride data for Down-gradient 2 from 10/3/2014 through 11/6/2014 is due to no EC 
probe installed in the well during that time period.
Laboratory Cl data were found to be log-normally distributed and were transformed prior to ANOVA analysis (Figure 12). Cl concentrations in down-gradient wells were significantly (p<0.01 and 0.001 for Down-gradient wells 1 and 2, respectively) less than Cl concentrations in the up-gradient well (Table 2). Down-gradient 2 had the lowest mean Cl concentration (212 mg/L) and the smallest standard deviation of all wells. This relatively low degree of variability further confirms that Down-gradient 2 is not hydraulically connected to the PA reservoir, as peaks of high Cl concentrations and periods of low concentrations from dilution would have been observed.

Mean concentrations in Down-gradient 2 are slightly lower than the DEEP chronic aquatic toxicity limit (Table 2). Although there are sidewalks that are salted in close proximity to Down-gradient 2 that may act as a potential source of Cl, a much greater area of non salted IC in the immediate area of the well, including a large building and impermeable parking lot, may decrease the potential for infiltration by Cl contaminated snowmelt which could otherwise impact water quality.

The Up-gradient well had the highest mean Cl concentration (Table 2) which was above both the DEEP chronic and acute Cl aquatic toxicity limits. Despite being up-gradient from the PA parking lot, deicing activities on sidewalks in the immediate area around Up-gradient result in this high level of Cl contamination. Down-gradient 1 mean Cl concentration lies between Down-gradient 2 and Up-gradient at 303.7 mg/L. The standard deviation is the highest at Down-gradient 1 at 1736.6 due to the high variability of Cl content in percolating water through the PA reservoir and high infiltration rates. While the mean Cl concentration in Down-gradient 1 was above both the DEEP chronic aquatic toxicity level and the mean concentration in Down-gradient 2, concentrations would be expected to be much higher in Down-gradient 1 had there
not been the high volume of dilution from the PA parking lot. Dilution of groundwater from the increased infiltration by the PA can therefore be credited with mitigating the effects of Cl contamination in an urban environment with high background Cl concentrations.

Figure 12. Box plots of chloride concentrations from 9/26/2014 through 5/8/2015. Bold horizontal lines represent the median chloride concentration.

Table 2. ANOVA analyses for Cl in Down-gradient 1, Down-gradient 2 and Up-gradient wells. Means followed by the same letter are not significantly different from each other.

<table>
<thead>
<tr>
<th>Location</th>
<th>Mean (antilog)</th>
<th>Standard Deviation</th>
</tr>
</thead>
<tbody>
<tr>
<td>Up-gradient</td>
<td>1584.2b</td>
<td>987.4</td>
</tr>
<tr>
<td>Down-gradient 2</td>
<td>212.0a</td>
<td>155.2</td>
</tr>
<tr>
<td>Down-gradient 1</td>
<td>303.7 a</td>
<td>1736.6</td>
</tr>
</tbody>
</table>
Metals

Groundwater samples were collected both before deicing activities had started in fall 2014 and after deicing activities had begun in winter 2015. Concentrations of metals in all wells decreased between pre-deicing sampling dates, indicating dilution from precipitation (Table 3, Figures 13). Down-gradient 1 and Down gradient 2 Mg and Ca concentrations increased between October and January as well as from January to March. Up-gradient increased in Mg and Ca concentration between October and January. Mg and Ca concentrations decreased Up-gradient between January and March sampling dates. Increases in Ca and Mg concentrations observed between at least 1 pair of sampling dates in winter and early spring when Cl concentrations are elevated suggest cation exchange has occurred in all wells. The decrease in Na associated with increases in Mg and Ca in Down-gradient 2 between October and January further supports the evidence for cation exchange as there are no significant sources of Ca or Mg in the immediate vicinity of the site other than soil adhered ions.

Environment Canada (2001) and Bäckström et al. (2003) have documented increases in metal concentrations in groundwater from cation exchange between dissolved Na cations from NaCl deicing salts and soil metals. While many studies have investigated the fate of metal concentrations in groundwater beneath PA in relation to surface runoff (Legret and Colandini 1999; Pagotto et al. 2000; Boving et al. 2008; Roseen et al. 2011), studies have not investigated the extent of cation exchange in groundwater beneath PA due to surface applications of deicing salts. Increases in metal concentrations (zinc, cadmium and copper) in soil water in winter and early spring by means of cation exchange and the formation of Cl complexes associated with deicing salt have been documented in the literature (Bauske and Goetz 1993; Bäckström et al. 2003). Brattebo and Booth (2003) found increases in zinc in groundwater beneath permeable pavement over a six year period; it is not clear if those elevated concentrations are due to
contaminated percolating water or from cation exchange with soil ions. Additionally, concentrations of zinc and copper were observed by Boving et al. (2008) to be highest in groundwater beneath PA in late winter and early spring when Cl concentrations were highest, though these increases may be due to corrosion of automobile parts from deicing salts (Boving et al. 2008).

Despite the relatively large increase in Ca (4,632 %) from January to March and Mg (1,946 %) from October to March in Down-gradient 1, the concentrations for both Ca and Mg in Down-gradient 1 are lower than up-gradient for all sampling dates and all but one sampling date for Ca in Down-gradient 2. Additionally concentrations of Ca and Mg in Up-gradient and Down-gradient 2 are higher than average shallow groundwater concentrations found by Flanagan et al. (2001) for Ca (12.4 mg/L) and Mg (2.7 mg/L). Down-gradient 1 exceeds the average concentration for Ca only in March and for Mg in January and March. This suggests that the potential effect from cation exchange is greater in areas surrounding the PA than beneath it, as metals concentrations are generally much lower down-gradient. More research needs to be conducted to investigate the potential risk for metal mobilization from deicing salt applications on PA in areas with low Ca and Mg background concentrations.
Table 3. Ca, Mg, and Na concentrations from sampling events before salt applications (September and October) and during (January and March) in down-gradient 1, down-gradient 2 and up-gradient.

<table>
<thead>
<tr>
<th></th>
<th>Ca (mg/L)</th>
<th>% ∆</th>
<th>Mg (mg/L)</th>
<th>% ∆</th>
<th>Na (mg/L)</th>
<th>% ∆</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Up-gradient</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>26-Sep</td>
<td>157.4</td>
<td>N/A</td>
<td>30.3</td>
<td>N/A</td>
<td>300.1</td>
<td>N/A</td>
</tr>
<tr>
<td>3-Oct</td>
<td>142.8</td>
<td>-9</td>
<td>27.7</td>
<td>-9</td>
<td>283.1</td>
<td>-6</td>
</tr>
<tr>
<td>16-Jan</td>
<td>748.1</td>
<td>424</td>
<td>146.0</td>
<td>427</td>
<td>575.5</td>
<td>103</td>
</tr>
<tr>
<td>20-Mar</td>
<td>501.2</td>
<td>-33</td>
<td>93.6</td>
<td>-36</td>
<td>642.5</td>
<td>12</td>
</tr>
<tr>
<td><strong>Down-gradient 2</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>26-Sep</td>
<td>102.8</td>
<td>N/A</td>
<td>18.7</td>
<td>N/A</td>
<td>230.1</td>
<td>N/A</td>
</tr>
<tr>
<td>3-Oct</td>
<td>86.8</td>
<td>-16</td>
<td>16.0</td>
<td>-15</td>
<td>193.1</td>
<td>-69</td>
</tr>
<tr>
<td>16-Jan</td>
<td>102.5</td>
<td>18</td>
<td>22.8</td>
<td>42</td>
<td>60.6</td>
<td>-67</td>
</tr>
<tr>
<td>20-Mar</td>
<td>123.4</td>
<td>20</td>
<td>29.0</td>
<td>27</td>
<td>61.9</td>
<td>2</td>
</tr>
<tr>
<td><strong>Down-gradient 1</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>26-Sep</td>
<td>2.6</td>
<td>N/A</td>
<td>0.5</td>
<td>N/A</td>
<td>44.6</td>
<td>N/A</td>
</tr>
<tr>
<td>3-Oct</td>
<td>1.7</td>
<td>-36</td>
<td>0.3</td>
<td>-40</td>
<td>18.5</td>
<td>-59</td>
</tr>
<tr>
<td>16-Jan</td>
<td>2.7</td>
<td>62</td>
<td>5.5</td>
<td>1,946</td>
<td>712.0</td>
<td>3,746</td>
</tr>
<tr>
<td>20-Mar</td>
<td>127.5</td>
<td>4,632</td>
<td>22.9</td>
<td>314</td>
<td>2,219.0</td>
<td>212</td>
</tr>
</tbody>
</table>

Figure 13. A (Up-gradient), B (Down-gradient 2), C (Down-gradient 1). Na, Ca and Mg concentrations from 4 sampling dates 9/26/2014, 10/3/2014, 1/16/2015, and 3/20/2015.
Limitations

The average hydraulic permeability rate calculated for this site does not seem to be representative of actual groundwater conditions beneath the PA. The rate of change in WTE and Cl (Figure 10), temperature (Figure 6) and EC (Figure 8) suggests faster average linear velocities than can be achieved through such low permeability. Higher permeability zones may exist that were not accounted for in boring logs. Additionally, as water table elevations change through time, layers of different permeability may become included or excluded from the aquifer and result in changes in average groundwater velocities.

The boring log shows a 1.29 m pea gravel layer extending from 2.44 to 3.73 m below grade and is most likely responsible for the rapid rate of change observed throughout the study (Appendix A). Down-gradient 2 and Up-gradient boring logs do not provide geologic evidence to explain the relatively fast rate of Cl and water table elevation response in the wells in relation to calculated hydraulic permeability.

The limited number of wells installed on site may result in a skewed average hydraulic permeability for the site and layers of preferential flow may not be identified by available boring logs. Additionally, the termination depth of the 3 wells only allows for the top 1.5 m of groundwater to be monitored and therefore does not allow for a comprehensive analysis of vertical anisotropies in aquifer water quality and hydraulics.
CONCLUSION

Changes in EC and Cl were found to be directly proportional to changes in WTE after deicing salt applications and snowmelt percolation. An inversely proportional relationship was observed during non-winter months due to dilution by percolating water. A relationship between EC and Cl was established that allowed EC to be used as a proxy for Cl concentrations. Over the course of the study from September 26, 2014 through May 8, 2015 Cl concentrations in groundwater beneath PA increased by 9,480%. Despite such a large increase, the mean concentration of 303.7 mg/L calculated for Down-gradient 1 was lower than the Up-gradient mean concentration of 1,280 mg/L. Despite the frequent high peaks of Cl observed in the Down-gradient well throughout the study, increased groundwater recharge from the PA resulted in a lower mean concentration down gradient. This suggests that PA acts to reduce high Cl concentrations in groundwater through increased groundwater recharge and subsequent dilution. The long-term effect of winter deicing of PA on groundwater with low concentrations of Cl is unknown.

Evaluation of metal concentrations spatially revealed at least one instance of Mg and Ca increases in all wells while deicing activities were active and Cl concentrations were at their highest. Despite the large increase of Ca (4,632%) and Mg (1,946%) in Down-gradient 1, metal concentrations in the Up-gradient well were higher than Down-gradient 1 for all sampling dates and all but one sampling date for Ca in Down-gradient 2. This suggests that the effect from cation exchange is less beneath PA than in surrounding areas. More research must be conducted to gain a more thorough understanding of cation exchange potential for metals beneath PA in areas with low Ca and Mg concentrations.
RECOMMENDATIONS

The results of this study suggest that the use of permeable asphalt in urban environments with high background Cl and metals concentrations is beneficial to shallow groundwater quality in the long term through dilution. The high surface infiltration rate of the PA allows for dilution of background Cl and metals concentrations during non-salting events, despite increases in Cl and metals concentrations beneath the PA during the winter from deicing salt applications. As observed during the period of study, Cl concentrations beneath the PA were lower than concentrations up-gradient during late summer when baseflow is the dominant source of available water for spring-fed streams. This suggests that the Cl concentration in baseflow down-gradient from PA is lower than the Cl concentration that would exist down-gradient without the PA. The limited extent of metals analysis during the study period suggests cation exchange has occurred, though a more comprehensive sampling regime is needed to explore this in greater detail. The installation of additional monitoring wells both up-gradient and down-gradient as well as at different terminating depths will build on the results of this study and help to understand vertical anisotropies in Cl concentrations, cation exchange potential and establish a relationship between up-gradient Cl concentrations and the effect of dilution on down-gradient concentrations.

Important factors which contribute or limit the ability for PA to properly function in diluting high groundwater Cl and metals concentrations include both the geology of the sub-grade soils as well as the PA surface infiltration rate. Over time, the pores in the PA can clog with sediment and infiltration rates will decline, reducing the volume of water available for dilution. Proper maintenance, including biannual vacuuming and restrictions on sanding on PA can help to maintain the surface porosity and sub-grade dilution.
Under conditions with relatively permeable soils, percolate from the PA reservoir can reach the water table and dilute concentrations of contaminants. In environments with dense clay and glacial till sub-grade soils permeability will be low and water in the PA reservoir will not be able to percolate through to the sub-grade soil efficiently. Under these circumstances, water may exit the system through the underdrain, which reduces the available water for dilution. Additionally, cation exchange potential will vary depending on soil composition as well as the quantity and species of anthropomorphic inputs of cations. Future studies will need to be conducted which examine cation potential and the potential for dilution under permeable asphalt with varying land use and sub-grade soil type.
LITERATURE CITED


Asleson, B. and Minnesota Pollution Control Agency (2013). Sharing the Road with the Environment. EPA's Stormwater Pollution Prevention Webinar Series: Road Salt Pollution Prevention Strategies.


Prince George’s County (1999). Low-impact development design strategies: An integrated design approach. Prince George's County, MD Department of Environmental Resources.


APPENDIX A: Well Boring Logs

Figure A-1. Boring log of Down-gradient 1 well.
Figure A-2. Boring log of Down-gradient 2 well.
Figure A-3. Boring log of Up-gradient well.
APPENDIX B: Constant Head Pumping Test

Table B-1. Constant Head Single Well Pumping Test Analyzed using Hvorslev Steady State Equation.

<table>
<thead>
<tr>
<th></th>
<th>Down-gradient 2</th>
<th>Up-gradient</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Pumping Rate (ml/m)</strong></td>
<td>82, 151, 300</td>
<td>78, 134, 264</td>
</tr>
<tr>
<td><strong>Drawdown (cm)</strong></td>
<td>17.68, 48.46, 105.77</td>
<td>14.14, 24.99, 51.63</td>
</tr>
<tr>
<td><strong>Hydraulic Permeability (cm/s)</strong></td>
<td>2.08 E-4</td>
<td>3.18 E-4</td>
</tr>
</tbody>
</table>
APPENDIX C: Estimation of Salt Application Rates

Table C-1. Estimated 2015 Winter deicing salt application rates. Two 37m² tarps collected salt during winter deicing salt application to determine average NaCl application rate.

<table>
<thead>
<tr>
<th>Salt Collected (kg)</th>
<th>Application Rate (kg/m²)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Collection Tarp # 1</td>
<td>2.60</td>
</tr>
<tr>
<td>Collection Tarp # 2</td>
<td>3.05</td>
</tr>
<tr>
<td><strong>Average</strong></td>
<td><strong>2.83</strong></td>
</tr>
</tbody>
</table>

Table C-2. 2015 Winter NaCl Loads to permeable asphalt. Estimates of the total number of deicing applications is based on an average of 3 applications per storm event.

<table>
<thead>
<tr>
<th>Total # of Storms</th>
<th>14</th>
</tr>
</thead>
<tbody>
<tr>
<td>Total # of Applications</td>
<td>42</td>
</tr>
<tr>
<td>Total NaCl Salt Per Application (kg)</td>
<td>62.45</td>
</tr>
<tr>
<td>Total NaCl Salt Load to PA (kg)</td>
<td>2,623</td>
</tr>
<tr>
<td>Total Cl Load to PA (kg)</td>
<td>1,591</td>
</tr>
</tbody>
</table>
**APPENDIX D: Laboratory Chloride Data**

Table D-1. Laboratory chloride measurements from ion specific electrode, analyzed using EPA Method 9212 (US EPA 1996).

**Chloride Concentrations (mg/L)**

<table>
<thead>
<tr>
<th>Date</th>
<th>Down-Gradient 1</th>
<th>Down-gradient 2</th>
<th>Up-gradient</th>
</tr>
</thead>
<tbody>
<tr>
<td>26-Sep-14</td>
<td>28.6</td>
<td>494.1</td>
<td>860.4</td>
</tr>
<tr>
<td>3-Oct-14</td>
<td>10.3</td>
<td>462.0</td>
<td>901.8</td>
</tr>
<tr>
<td>13-Oct-14</td>
<td>18.0</td>
<td>404.8</td>
<td>913.4</td>
</tr>
<tr>
<td>27-Oct-14</td>
<td>27.4</td>
<td>306.9</td>
<td>778.9</td>
</tr>
<tr>
<td>20-Nov-14</td>
<td>44.5</td>
<td>73.2</td>
<td>539.3</td>
</tr>
<tr>
<td>1-Dec-14</td>
<td>271.0</td>
<td>81.3</td>
<td>905.3</td>
</tr>
<tr>
<td>17-Dec-14</td>
<td>881.0</td>
<td>142.0</td>
<td>1935.3</td>
</tr>
<tr>
<td>30-Dec-14</td>
<td>247.8</td>
<td>229.8</td>
<td>2705.1</td>
</tr>
<tr>
<td>16-Jan-15</td>
<td>1226.3</td>
<td>146.9</td>
<td>3041.5</td>
</tr>
<tr>
<td>30-Jan-15</td>
<td>2626.0</td>
<td>127.3</td>
<td>2945.5</td>
</tr>
<tr>
<td>25-Feb-15</td>
<td>1323.3</td>
<td>80.1</td>
<td>2669.3</td>
</tr>
<tr>
<td>20-Mar-15</td>
<td>4339.6</td>
<td>298.3</td>
<td>2327.0</td>
</tr>
<tr>
<td>25-Mar-15</td>
<td>5507.0</td>
<td>318.9</td>
<td>2856.0</td>
</tr>
<tr>
<td>10-Apr-15</td>
<td>1038.0</td>
<td>472.5</td>
<td>2797.0</td>
</tr>
</tbody>
</table>
APPENDIX E: Precipitation Data

Table E-1. Precipitation data during period of study.

<table>
<thead>
<tr>
<th>Month</th>
<th>Study Period</th>
<th>29 Year Average</th>
<th>% Deviation from Normal</th>
</tr>
</thead>
<tbody>
<tr>
<td>September</td>
<td>3.89</td>
<td>10.386</td>
<td>-62.59</td>
</tr>
<tr>
<td>October</td>
<td>14.99</td>
<td>11.684</td>
<td>28.26</td>
</tr>
<tr>
<td>November</td>
<td>10.72</td>
<td>11.6078</td>
<td>-7.66</td>
</tr>
<tr>
<td>December</td>
<td>10.01</td>
<td>10.6934</td>
<td>-6.41</td>
</tr>
<tr>
<td>January</td>
<td>8.31</td>
<td>9.6266</td>
<td>-13.72</td>
</tr>
<tr>
<td>February</td>
<td>5.99</td>
<td>8.4836</td>
<td>-29.34</td>
</tr>
<tr>
<td>March</td>
<td>7.80</td>
<td>11.2776</td>
<td>-30.86</td>
</tr>
<tr>
<td>April</td>
<td>10.19</td>
<td>11.5062</td>
<td>-11.48</td>
</tr>
</tbody>
</table>