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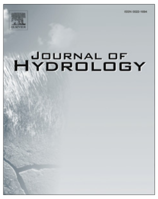

# Salt vulnerability assessment methodology for urban streams

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## SUMMARY

De-icing agents such as road salts while used for winter road maintenance can cause negative effects on urban stream water quality and drinking water supplies. A new methodology using readily available spatial data to identify Salt Vulnerable Areas (SVAs) for urban streams is used to prioritize implementation of best management practices. The methodology calculates the probable chloride concentration statistics at specified points in the urban stream network and compares the results with known aquatic species exposure tolerance limits to characterize the vulnerability scores. The approach prioritizes implementation of best management practices to areas identified as vulnerable to road salt. The vulnerability assessment is performed on seven sites in four watersheds in the Greater Toronto Area and validated using the Hanlon Creek watershed in Guelph. The mean annual in-stream chloride concentration equation uses readily available spatial data – with province-wide coverage – that can be easily used in any urban watershed.

## 1. Introduction

One of the most commonly employed management strategies implemented by winter road maintenance agencies is application of de-icing agents as anti-icing methods before an expected winter precipitation event or as de-icing methods after ice has formed on the road surface. In North America, the most commonly used de-icing agent by winter road maintenance agencies and for large industrial/commercial parking lots is rock salt comprised of sodium chloride with minor impurities (Perera et al., 2013). Road salts work by lowering the freezing point of water, thereby inhibiting the formation of ice.

In Canada, approximately 5 million tonnes of road salts are applied annually on roadways across the country (Environment Canada, 2004). In the US, approximately 18 million tonnes of road salts are applied each year (Jackson and Jobbagy, 2005). The majority of the road salts used in both Canada and the US are applied in major urban centers, mainly due to the high density of road networks and parking lots.

Various studies have documented that both aquatic and terrestrial ecosystems are adversely affected by exposure to high chloride concentrations associated with the typical use of road salts in urban streams (CCME, 2011; D'Itri, 1992; Adelman et al., 1976). Further, there is significant evidence of increasing chloride concentrations in both surface waters and groundwaters in urban watersheds due to the application of road salts (Mayer et al., 1999; Williams et al., 2000; Godwin et al., 2003; Thunqvist, 2004; Kaushal et al., 2005; Lundmark and Olofsson, 2006; Perera et al., 2009; Winter et al., 2011). This is a major concern for the ecological health of sensitive aquatic species as well as the quality of drinking water supplies.

Recent studies have shown that high concentrations of chloride ions associated with road salts have the potential for both immediate and long-term adverse effects on surface water systems (CCME, 2011; US EPA, 1988). High chloride concentrations in surface waters increase metal bioavailability, affect community food web structure, diversity and productivity of aquatic species (Environment Canada and Health Canada, 2001).

Chlorides do not biodegrade nor readily precipitate, volatilize, or bio-accumulate (CCME, 2011). The persistence of chlorides in the environment does not allow easy treatment, and, therefore, most of the effort in minimizing the impact of road salt is focused on optimizing salt application rates and implementing various best management practices (TAC, 2003). Currently, there are no federal regulations for the use of road salts in Canada or the United States. Several studies have developed thresholds and guidelines for road salts (US EPA, 1988; Environment Canada and Health Canada, 2001; TAC, 2003; Environment Canada, 2004; CCME, 2011). The US EPA developed toxicity thresholds for chlorides, which include

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## Nomenclature

| | | | |
|---|---|---|---|
| $A$ | Contributing area ($m^2$) | GIS | Geographical Information System (–) |
| CGVD28 | Canadian Geodetic Vertical Datum 1928 (–) | GPS | Global Positioning System (–) |
| BFC | Baseflow Chloride Concentration (mg/L) | SCC | Mean Annual Stream Chloride Concentration (mg/L) |
| BFI | Base Flow Index (–) | SVA | Salt Vulnerable Areas (–) |
| CAD | Chloride Application Density (–) | UAR | Unit Chloride Application Rate ($g/m^2$) |
| CCME | Canadian Council of Ministers of the Environment (–) | MAF | Normalized Mean Annual Flow (m) |
| $Cl^-$ | Chloride Ion (–) | MOE | Ontario Ministry of the Environment (–) |
| EC | Electrical Conductivity (–) | MTO | Ontario Ministry of Transportation (–) |
| EPA | Environmental Protection Agency (–) | PGMN | Provincial Groundwater Monitoring Network (–) |
| $LC_{50}$ | Lethal Concentration 50% (–) | US EPA | United States Environmental Protection Agency (–) |

chronic freshwater quality criterion of 230 mg/L and an acute freshwater quality criterion of 860 mg/L (US EPA, 1988). Canadian Council of Ministers of the Environment (CCME) developed Canadian drinking water standards that outlined aesthetic objectives for chloride and sodium at 250 mg/L and 200 mg/L, respectively.

Environment Canada (2012) concluded that attention to salt vulnerable areas was significantly lacking in provincial and municipal salt management plans (SMP). Less than 30% of the SMP inventoried salt vulnerable areas. Salt vulnerable areas are defined as any area susceptible to adverse impact to the health of the aquatic species or quality of drinking water sources as a result of the application of road salts during winter maintenance activities on roads and parking lots. The low rate of participation of road agencies in identifying salt vulnerable areas may be due to a lack of clear guidance of the methods and the concern that the process of identifying salt vulnerable areas may require expensive and advanced data collection and analysis (Environment Canada, 2012). Salt vulnerable areas are those which would benefit most from Best Management Practices (BMPs) outlined in salt management plans and hence it is prudent to identify these key areas in which to take action and reduce risk.

This paper describes a methodology, using readily available Geographical Information System (GIS) data, to identify areas vulnerable to road salts, through evaluation of the impact to aquatic species caused by the application of road salts. The methodology quantifies the vulnerability to identified areas in order to prioritize implementation of best management practices.

## 2. Study areas

Six different urban watersheds (Study Rivers) are considered within the City of Toronto boundary: Etobicoke Creek, Mimico Creek, Humber River, Don River, Highland Creek, and Rouge River. Estimation of the vulnerability to road salt application was performed on seven sites in four of the watersheds in the City of Toronto, Ontario (Fig. 1) and validated using Hanlon Creek Watershed in the City of Guelph, Ontario (Fig. 2).

Data were acquired from the City of Toronto Stream Chloride Monitoring Program. The Study Areas included two sites on the Humber River (at Steeles Ave. and Old Mill Rd.), the Don River (at Bloor St.), Highland Creek (at Morningside Ave.), and the Morningside Creek tributary of the Rouge River (at Finch Ave.). To supplement the City of Toronto monitoring data, Perera et al. (2009) added two additional stations in the Highland Creek watershed in 2007. These two locations were selected to represent the two main tributaries of Highland Creek, West Highland Creek (at Bellamy Rd.) and Malvern Branch (at Mammoth Hall Trail). Data for the Hanlon Creek watershed was collected as part of this research project for the water year of November 2010 to October 2011. This station was located just downstream of Highway 6 on Hanlon Creek.

### 2.1. Data collection

As a result of the Code of Practice for Road Salts, the City of Toronto Stream Chloride Monitoring Program collects hourly electrical conductivity readings, as a surrogate for chloride concentration, using a Hach conductivity sensor (model no. 5798A) with a Hach Sigma 900 Max autosampler attached (Perera et al., 2009). As part of this research project, electrical conductivity (as a surrogate for chloride concentration) was also monitored in the Hanlon Creek watershed using CTD-Diver (a Schlumberger product) with a depth range of 10 m and electrical conductivity range of 80 microSiemens per centimeter (μS/cm) on 10 min intervals.

The seven monitoring sites within Toronto cover a wide range of watershed areas and land use characteristics. The contributing watersheds to the seven monitoring sites range in size from 15 to 878 $km^2$. Land use characteristics also cover a wide range within the Study Areas with predominant land use changing from industrial, institutional, residential to open area.

The Hanlon Creek monitoring station was selected to provide an additional study area outside of Toronto for purposes of methodology validation. The contributing watershed area to the Hanlon Creek is small compared to the other seven Study Areas (10.7 $km^2$), but the land use characteristics include a wide range with almost an even split of industrial (27.1%), residential (28.1%) and open land use (29.3%), with the rest of the area consisting of city roads (11.9%), commercial (1.6%) institutional (1%) and MTO highway (1%). Table 1 presents the summary statistics of stream chloride concentrations for the eight case study watersheds, showing roughly sevenfold differences in the mean annual chloride concentrations ranging from 118 to 765 mg/L.

### 2.2. Electrical conductivity and chloride concentration

The monitoring program established by the City of Toronto was developed to collect continuous chloride concentration data. The monitoring program utilized measurements of specific conductance, also known as electrical conductivity (EC), as a surrogate for chloride concentrations ($Cl^-$). In order to establish the correlation between EC and $Cl^-$ readings, grab samples were collected and analyzed by the Toronto Water Laboratory for EC and major ions (sodium, calcium, magnesium, potassium, chloride, sulfate, and bromide). A number of studies (e.g. Howard and Haynes, 1993; Granato and Smith, 1999; Guan et al., 2010; Kilgour et al., in press, and Perera et al., 2013) identified that a strong linear relationship exists between EC and $Cl^-$. Perera et al. (2009) indicated that at low EC values a simple linear relationship resulted in poor accuracy and at