Environment Canada
Water Science and Technology Directorate

Direction générale des sciences et de la technologie, eau
Environnement Canada

TITLE/TITRE: The Effects of Road Salts on Aquatic Ecosystems

AUTHORS/AUTEURS: M. Evans and C. Frick

REFERENCE/RÉFÉRENCE: WSTD Contribution No. 02-308

The Effects of Road Salts on Aquatic Ecosystems

Marlene Evans
National Water Research Institute
11 Innovation Blvd.
Saskatoon, SK, S7N 3H5

and

Cherie Frick
Department of Biology
University of Saskatchewan
Saskatoon, SK S7K 5E2

N.W.R.I. Contribution Series No. 02-308
August 2001

MANAGEMENT PERSPECTIVE

This report is based on an assessment of the known and potential effects of road salts on Canadian aquatic ecosystems. It is one of several supporting documents to the Priority Substance List 2 assessment of the toxicity of road salts. A literature review was conducted on chloride toxicity and the environmental impacts of elevated chloride concentrations, primarily from road salt applications. Provincial water quality monitoring data provided information on chloride levels in various water bodies. Based on these investigations, it is concluded that chloride salts from road salt are toxic in the environment although in limited geographic areas where large amounts of salt are applied near highways. Such areas occur primarily in southern Ontario (especially Metropolitan Toronto) and Quebec. Improperly designed and/or maintained road salt depots is another cause for concern as are improper snow disposal practices. At lower concentrations, road salt can have chronic impacts on aquatic ecosystems. Such impacts will be more widespread.

The designation of road salt as a toxic substance will necessitate various actions to reduce road salt inputs to aquatic, terrestrial, and groundwater ecosystems. This will include modifying application rates, improving the operation of road salt storage depots, and using safe waste-snow disposal methods. While chloride has been measured during provincial water quality monitoring programs, these data generally have been poorly synthesized to assess long-term trends and spatial variation in chloride concentrations. Lakes have been particularly poorly studied. There also is a poor understanding of chloride effects at chronic concentrations (ca. 100-1,000 mg/L), and the concomitant entry of nutrients and metals into aquatic ecosystems with road salt runoff. While often low in concentration, increased inputs of chloride, nutrients, and metals with highway runoff may stimulate the productivity of oligotrophic ecosystems. As more highways are constructed in relatively undeveloped and pristine regions, particularly on the Canadian Shield, and rural aquatic ecosystems become incorporated into the urban, aquatic ecosystems located near these roadways may be adversely impacted. In particular, species shifts may occur, aquatic ecosystems become may more productive, and some lakes may become meromictic (chemically stratified).

SOMMAIRE À L’INTENTION DE LA DIRECTION

Ce rapport est fondé sur une évaluation des effets connus et potentiels des sels de voirie sur les écosystèmes aquatiques canadiens. Il est l’un des documents qui a été soumis lors de l’évaluation de la toxicité des sels de voirie inscrits sur la deuxième liste des substances d'intérêt prioritaire. Une étude de la documentation a été réalisée sur la toxicité des chlorures et sur les effets environnementaux des hausses des concentrations de chlorure causées principalement par l’épandage des sels de voirie. Des données sur les concentrations des chlorures dans divers plans d’eau ont été obtenues des programmes provinciaux de surveillance de la qualité de l’eau. Les résultats de ces recherches ont démontré que les chlorures des sels de voirie sont toxiques pour l’environnement, mais seulement dans les quelques régions où de grandes quantités de sel sont appliquées à proximité des routes. Ces régions sont concentrées dans le sud de l’Ontario (en particulier la Communauté urbaine de Toronto) et du Québec. Les installations d’entreposage du sel de voirie mal conçues ou mal entretenues et les mauvaises pratiques d’élimination de la neige usée contribuent également au problème. À des concentrations plus faibles, les sels de voirie peuvent avoir des effets chroniques sur les écosystèmes aquatiques. Ces effets sont plus répandus.

La reconnaissance de la toxicité des sels de voirie nécessitera l’application de diverses mesures destinées à réduire les effets de ces sels sur les écosystèmes aquatiques, terrestres et souterrains. Il faudra notamment réduire les quantités appliquées, mieux gérer les installations d’entreposage des sels de voirie et utiliser des méthodes sûres pour éliminer la neige usée. Bien que les concentrations des chlorures soient mesurées dans le cadre des programmes provinciaux de surveillance de la qualité de l’eau, peu d’efforts ont été consacrés à la synthèse de ces données en vue d’évaluer les fluctuations à long terme et spatiales des concentrations de chlorure. L’état de la situation dans les lacs a été très peu étudié. Notre compréhension des effets des chlorures à des concentrations induisant des répercussions chroniques (environ 100 à 1,000 mg/L) est également fragmentaire. En outre, les effets des eaux de ruissellement contaminées par des sels de voirie sur l’entrée concomitante d’éléments nutritifs et de métaux dans les écosystèmes aquatiques demeurent à préciser. Bien que faibles, les concentrations accrues de chlorures, d’éléments nutritifs et de métaux charriés par les eaux de ruissellement des routes peuvent stimuler la productivité des écosystèmes oligotrophes. À mesure que de nouvelles routes sont construites dans des régions relativement peu développées ou sauvages, particulièrement sur le bouclier canadien, et que les écosystèmes aquatiques ruraux sont intégrés au paysage urbain, les écosystèmes aquatiques situés à proximité de ces routes risquent d’être affectés. Des espèces pourraient être remplacées, la productivité des écosystèmes aquatiques pourrait augmenter, et certains lacs pourraient devenir méromictiques (stratifiés chimiquement).

ABSTRACT

This study is based on the Priority Substance List 2 assessment of the toxicity of the application of road salt to the aquatic environment. The assessment begins with a review of the PSL2 assessment process, including the formation of an Environment Resource Group (ERG) to conduct this assessment. Road salt is then characterized including its composition, properties, and application rates across Canada. While road salt is a complex mixture of chloride salts, various anticaking compounds such as sodium ferrocyanide, and abrasives, this assessment focuses on the chloride salts. Sodium followed by calcium are the primary chloride salts used in the winter application of road salt; calcium chloride is used mainly as a summer dust suppressant. Small amounts of potassium and magnesium chloride salts also are used as a winter deicer. Winter road salt is applied most heavily in southern Ontario, Quebec, Nova Scotia, New Brunswick, British Columbia, and the central (Edmonton) area of Alberta.

An extensive review of the Canadian and United States literature determined that a relatively small number of studies have been conducted investigating road salt in the aquatic environment. Road salt impacts, defined as increased chloride concentrations, were most pronounced in urban areas and areas located near heavily salted highways. Furthermore, there is some evidence of more gradual increases in chloride concentrations in lakes in these regions, in part a result of road salt application. Fewer studies have investigated the impacts of road salt to aquatic ecosystem structure and function. However, some small, shallow lakes have been shown to become meromictic, benthic drift has increased in some streams with pulses of road salt being flushed through the system, and other alterations in benthic and phytoplankton communities noted. No studies were located which investigated the impacts of calcium chloride as a dust suppressant.

An extensive review was conducted of laboratory studies investigating the toxicity of sodium, calcium, potassium, and magnesium chlorides. By far the largest data sets were obtained for sodium chloride. Magnesium and potassium chloride appear to be more toxic than sodium chloride for all organisms investigated. Plankton and invertebrates appear to be more sensitive to calcium than sodium chloride while the reverse appears to occur for fish. Most laboratory studies were conducted over short time intervals (i.e., 4 days or less) and investigated mortality. Tolerances were high, with LC50 ranging from 1,400-13,085 mg/L chloride. However, this range also spans the range of chloride concentrations observed in road salt snow melt, storm sewers, and urban creeks and rivers. In these environments, short-term exposures to elevated chloride concentrations are likely to be toxic. Chloride concentrations in this range also have been observed in wetlands near leaking road salt depots. A few studies were conducted over a 7-day period with EC50 ranging from 874-3,330 mg/L chloride. Chloride concentrations in this range have been observed in some urban creeks, rivers, and ponds. Chronic toxicity (i.e., exposure to elevated chloride concentrations over extended time periods) was estimated as ranging from 150-1,402 mg/L chloride. Concentrations in this range have been observed for salt-impacted creeks, rivers, ponds, and lakes, again primarily in urban areas or areas near major highways.

It is concluded that chloride salts from road salt are toxic in the environment although in limited geographic areas where large amounts of salt are applied to highways or near leaking road salt depots. The documentation of these impacts is problematic with much of the water quality data (e.g., on chloride concentrations) in the gray literature or in data sets which have yet to be

synthesized. Chloride is a required element for the well being of organisms and, in some sense can be viewed like phosphorous.

RÉSUMÉ

Cette étude est fondée sur l'évaluation de la toxicité pour les écosystèmes aquatiques des sels de voirie figurant sur la deuxième liste des substances d'intérêt prioritaire. L'évaluation débute par un examen du processus d'évaluation prévu pour les substances figurant sur la LSIP2, y compris la mise sur pied d'un groupe ressource environnemental en vue de cette évaluation. Les sels de voirie sont ensuite caractérisés d'après leur composition, leurs propriétés et les quantités auxquelles ils sont appliqués au Canada. Bien que les sels de voirie soient un mélange complexe de chlorures, de divers agents antiagglomérants comme le ferrocyanure de sodium et d'abrasifs, la présente évaluation vise les chlorures. Le sodium, suivi du calcium, sont les principaux chlorures utilisés comme sels de voirie en hiver. Le chlorure de calcium est utilisé surtout en été comme agent dépoussiérant. Le chlorure de potassium et le chlorure de magnésium sont également utilisés en faibles quantités en hiver pour le déglaçage des routes. En hiver, les plus fortes quantités de sels de voirie sont appliquées dans le sud de l'Ontario, au Québec, en Nouvelle-Écosse, au Nouveau-Brunswick, en Colombie-Britannique et dans le centre de l'Alberta (Edmonton).

Une importante étude de la documentation publiée au Canada et aux États-Unis a révélé qu'un nombre relativement restreint d'études ont été consacrées aux effets des sels de voirie sur l'environnement aquatique. Les impacts des sels de voirie, assimilés à une hausse des concentrations des chlorures, sont plus prononcés dans les régions urbaines et dans les secteurs situés à proximité de routes recevant de fortes quantités de sels. En outre, les hausses graduelles de chlorures semblent plus importantes dans les lacs situés dans ces régions, en partie à cause des épandages de sels de voirie. Un nombre encore plus restreint de chercheurs se sont intéressés aux effets des sels de voirie sur la structure et les fonctions des écosystèmes aquatiques. Toutefois, certains petits lacs peu profonds sont devenus méromictiques, la dérive des organismes benthiques s'est intensifiée dans certains ruisseaux par suite de l'introduction subite de quantités massives de sels de voirie, et d'autres modifications des communautés benthiques et phytoplanctoniques ont été notées. Nous n'avons trouvé aucune évaluation des impacts du chlorure de calcium comme agent dépoussiérant.

Une attention particulière a été consacrée à l'examen des évaluations en laboratoire de la toxicité des chlorures de sodium, de calcium, de potassium et de magnésium. Les plus vastes ensembles de données se rapportent au chlorure de sodium. Le chlorure de magnésium et le chlorure de potassium semblent plus toxiques que le chlorure de sodium pour tous les organismes testés. Les organismes planctoniques et les invertébrés semblent plus sensibles au chlorure de calcium qu'au chlorure de sodium. La tendance s'inverse chez les poissons. La plupart des évaluations réalisées en laboratoire ont été réalisées sur de courtes périodes (4 jours ou moins) et ont porté essentiellement sur les effets létaux. Les seuils de tolérance étaient élevés, les CL_{50} variant entre 1 400 et 13 085 mg/L de chlorure. Toutefois, cet intervalle correspond à la gamme des concentrations de chlorure mesurées dans les eaux de fonte, les égouts pluviaux et les ruisseaux et rivières urbains. Dans ces environnements, l'exposition à court terme aux fortes concentrations de chlorure a probablement des effets toxiques. Des concentrations de chlorure de cet ordre ont également été observées dans des milieux humides situés près d'installations d'entreposage de sels de voirie présentant des fuites. Quelques études réalisées sur une période

de 7 jours ont produit des CE_{50} de 874 à 3 330 mg/L de chlorure. Des concentrations comparables de chlorure ont été observées dans quelques ruisseaux, rivières et étangs urbains. La toxicité chronique (exposition prolongée à de fortes concentrations de chlorure) a été estimée à des concentrations variant entre 150 et 1 402 mg/L de chlorure. Des concentrations de cet ordre ont été enregistrées dans des ruisseaux, rivières, étangs et lacs contaminés par des sels de voirie, encore une fois principalement en milieu urbain ou à proximité de routes importantes.

Il est donc établi que les chlorures utilisés comme sels de voirie sont toxiques dans l'environnement, bien que le problème soit limité aux régions où de grandes quantités de sel sont appliquées sur les routes ou à proximité des installations d'entreposage de sels de voirie non étanches. La documentation de ces impacts est difficile, car la plupart des données de surveillance de la qualité de l'eau (p. ex. concentrations des chlorures) extraites de la littérature grise ou de base de données n'ont pas encore été synthétisées. Les chlorures jouent un rôle indispensable dans le bien-être des organismes, comme le phosphore à certains égards.

ACKNOWLEDGMENTS

This study was supported in large measure by the Commercial Chemical Evaluations Branch (CCEB) of Environment Canada who funded the initial literature review, additional laboaratory tests, and provided support for all Environmental Resource Group activities. Special appreciation is extended to Barbara Elliott and Rober Chenier. The National Water Research Institute, Burlington, also supported this research through technical and other support. We thank the members and participants in the ERG for their contributions to the PSL2 assessment of road salt, including the work reported here. Special appreciation is extended to Ken Howard (University of Toronto at Scarborough), Art Letts (Morton Salt), Tanya Mayer (NRWI), David Morin (CCEB), and Bill Snodgrass (Ontario Ministry of Transport).

TABLE OF CONTENTS

1.0 INTRODUCTION

1.1 General Background

Road salts are applied to roadways in northern-temperate climates to prevent traffic accidents. In winter, road salts are applied to lower the freezing point of road-ice or precipitation so that water returns to or remains in a liquid state, avoiding the problems associated with ice formation on roads (e.g., slipperiness) (Perchanok et al. 1991; Kostick 1993). Various studies indicate that traffic accidents can be reduced by 20 to 90% when icy and snowy roads are salted and restored to bare pavement (Vaa et al. 1996; Kuemmel and Hanbali 1992; Hanke and Levin 1990). In summer, road salts such as calcium chloride are applied to gravel roads for dust suppression (I. D. Systems Ltd. 1989).

One of the most commonly used road salts in North America is sodium chloride (Field and O'Shea 1992; Perchanok et al. 1991). Road salt can, however, be made up of different compounds and mixtures of compounds, including chloride salts/brines (e.g., ions of chloride, sodium, calcium, magnesium and potassium), organic compounds (e.g., urea and glycol), abrasives (e.g., sand), as well as additives that help prevent caking (e.g., sodium/ferric ferrocyanide) and corrosion.

Road salts first came into use in Canada and the northern United States in the 1940s. This usage has increased through the decades with accelerating population growth, greater urbanization and highway development, and improved highway safety programs. Levels of chloride in many North American lakes and streams have increased concurrently with the increased usage of road salts. In many instances, these increased chloride levels have been directly related to road salt (Boucher 1982; Crowther and Hynes 1977; Kerekes 1974; Ohno 1990; Williams et al. 1997). Due to concerns regarding the potential effects of road salts released into the environment, road salts were included as one of 25 substances recommended by the Minister's Expert Advisory Panel for assessment under the Priority Substances Assessment Program of the Canadian Environmental Protection Act (CEPA). This report is based on one aspect of that assessment, specifically the impacts of road salt on the aquatic environment. The general background for the overall assessment is as follows.

1.1.1 Canadian Environmental Protection Act

The Canadian Environmental Protection Act (CEPA) instructs the Federal Ministers of the Environment and of Health to develop a Priority Substances List (PSL). Substances on this List are then given priority assessment to determine whether they are "toxic" as defined under the Act. Section 11 of CEPA defines toxic substances as follows (Environment Canada 1997c):

> "For the purposes of this Part, a substance is toxic if it is entering or may enter the environment in a quantity or concentration or under conditions
>
> a) having or that may have an immediate or long-term harmful effect on the environment;
> b) constituting or that may constitute a danger to the environment on which human life depends; or
> c) constituting or that may constitute a danger in Canada to human life or health."

The first Priority List of Substances was published in the *Canadian Gazette* in 1989 and contained 44 substances which required evaluation within the legislated 5-year time frame for PSL1 assessments. Following the assessment 25 substances were determined to be CEPA toxic. The second Priority List of Substances was published in 1995 and consists of 25 substances which require assessment under PSL2. Road salts are one of those 25 substances.

1.1.2 The PSL2 CEPA Assessment Process

Assessment under PSL2 involves several processes. Health Canada is responsible for assessing the risk to human health from environmental exposure to the Priority Substance while Environment Canada assesses the risk to the environment. The Chemical Evaluations Branch of Environment Canada begins this process by problem formulation. During this formulation, the goals and focus of the assessment are established. Following this, an Environmental Resource Group is formed to conduct an environmental assessment during which knowledge and data gaps are identified and strategies are developed to obtain this knowledge, where essential. Information is obtained on the modes of entry of the substance into the environment, exposure levels in the environment, and the subsequent effects of this substance on the environment, including humans. With this information, a series of three levels (tiers) of risk assessments are performed. If adverse effects are unlikely under the Tier 1-3 assessments, the substance is not considered CEPA "toxic". However, if adverse effects are likely, these effects are estimated and/or the ecological consequences described. The substance is then declared CEPA "toxic". The draft risk assessment is subject to review by the Environmental Resource Group, other government departments, and other experts. This is followed by a public scrutiny. Finally, as required under Section 13 of CEPA, Environment Canada and Health Canada will jointly publish the Assessment Report including the conclusion with respect to CEPA "toxic". A notice is then published in the *Canadian Gazette* summarizing the report and announcing the Ministers' intentions (Environment Canada 1997c). Other documents, such as supporting documents to the Assessment, reports, fact sheets, and reprints from scientific journals will also be made available.

1.1.3 The Environmental Resource Group

The Environmental Resource Group (ERG) for the assessment of road salt consists of several experts, each of whom is responsible for a particular area of the assessment process. Areas of expertise include representatives from the highway department, the salt industry, terrestrial vegetation, terrestrial vertebrates, groundwater, aquatic chemistry, and aquatic ecology. During its early meetings, the ERG deemed that a literature review and assessment of the potential effects of road salt and its additives on stream, lake, and wetland ecosystems was an essential component of the Road Salt Assessment. Furthermore, they determined that this review would serve as one of the supporting documents for the Road Salt Assessment Report.

This report focuses on the assessment of the environmental impacts of road salt applications on the aquatic environment. The toxicity of sodium, potassium, magnesium, and chloride salts to aquatic organisms is reviewed and the impacts of elevated salinity, including that derived from road salt applications on aquatic environments is assessed. Information also was obtained on environmental concentrations of road salt in the Canadian and United States environments. Together, this information is used in the Tier 1-3 Assessments of the toxicity of road salt.

1.2 Purpose and Objectives of the Literature Review and Assessment

The objectives of the literature review are as follows:

1) Provide a context for salts in the aquatic environment. This information was required because salts are a natural and essential feature of the aquatic environment having important effects on physical, chemical and biological processes. Moreover, salt concentration and composition vary naturally with geographic setting, being substantially higher in saline Prairie lakes than in lakes located on the Canadian Shield.

2) Provide a brief synthesis of the entry of road salts into the aquatic environment and their ultimate pathways and concentrations. This synthesis was required in order to link laboratory toxicity studies with various components of the aquatic environment. Information for this synthesis was obtained from published studies and the recent efforts of other ERG members.

3) Review and summarize existing literature regarding lethal and sublethal effects of sodium, calcium, magnesium, and potassium chloride salts on aquatic organisms. This information was synthesized in order to characterize potential effects of road salt application on aquatic organisms.

4) Review existing literature regarding known environmental effects of road salt application on stream, lake, and wetland ecosystems, with an emphasis on aquatic ecosystems in Canada. This information was synthesized in order to identify the situations where effects have occurred or could be expected to occur.

5) On the basis of Objectives 2, 3, and 4, assess whether all regions of Canada are equally sensitive to perturbation of the aquatic environment from road salt application.

6) Identify approaches and points that may be useful in assessing the environmental consequences of road salt application as related to freshwater environments.

Following the literature review, Tier 1, 2 and 3 Assessments were conducted. This process is explained in Section 7 of the report.

1.2.1 Road Salt Compounds of Concern

Road salts are a formulation of inorganic and organic compounds which are applied to roads for deicing and dust suppression. Chloride salts are more commonly used than organic salts, such as calcium magnesium acetate (CMA), which is used on roads, and urea and potassium acetate, which are used on runways. Road salts also contain additives such as abrasives (e.g., sand and cinders), anti-corrosive compounds (e.g., PCI or lignosulfonate, and CG-90 products), and anti-caking compounds (e.g., sodium ferrocyanide and ferric ferrocyanide). Only chloride salts, sodium ferrocyanide, and ferric ferrocyanide are being considered under the Road Salt Assessment. The chloride assessments form the basis of this report while the sodium ferrocyanide and ferric ferrocyanide assessments are being conducted under the leadership of A. Letts (Morton Salt, Chicago, IL). Other potential components of road salt, such as urea, calcium magnesium acetate, and abrasives (e.g., sand), were not included because they are either monitored under other programs or are not used to any great degree in Canada.

1.2.2 Study Area, Species of Concern, and Time Period

The study area includes inland streams, lakes, and wetlands across Canada, although studies from the United States have also been included. “Stream” is a generic term referring to lotic or running-water environments, ranging from springs, creeks, and streams. “Rivers” are larger flowing waters. A “lake” is defined as a permanent body of water that has two distinct zones, the littoral and the pelagic. The littoral zone is located near the lake shore where rooted and floating aquatic plants (i.e., macrophytes) grow. The pelagic zone is further from shore and rooted plants do not occur in these deeper waters. Smaller water bodies such as ponds are included in the “lake” designation. A “wetland” is differentiated from a lake in this review by the fact that wetlands may be ephemeral (i.e., they may dry up during the summer). Wetlands, being shallow systems, generally have a widespread growth of rooted and/or floating aquatic macrophytes. Marshes, bogs, swamps and sloughs are included in the category of wetlands.

Species of concern include all aquatic species for which there is data (i.e., bacteria, protozoa, fungi, phytoplankton, zooplankton, macrophytes, benthic invertebrates, fish, amphibians, and aquatic birds). Information on salt toxicity was collected for individual species as well as effects on ecosystem structure and function.

The time period of road salt application considered includes winter applications for deicing of roads and summer applications for dust suppression. The review itself includes information gathered from the early 1900’s (toxicity studies) to the present time.

1.2.3 Methodology

Most of the information summarized in this literature review was obtained using two methods: 1) collection of published literature and 2) collection of information from personal correspondence. Books, journal articles, and conference proceedings containing information on road salts and their additives were collected from local libraries. As well, many references were received from a literature search on road salts conducted by the Chemical Evaluations Branch during the problem formulation stage (Environment Canada 1997a).

In order to broaden the background data, upon which the assessment is based, approximately eighty organisations and government agencies were contacted in summer 1998 in order to solicit unpublished information and opinions on the effects of road salts on the aquatic environment. Organisations that were contacted included provincial utilities, fisheries, academic institutes, and scientific societies. Two scientific societies, North American Lake Management Society and Society of Environmental Toxicology and Chemistry, posted notes on their web-sites and in their newsletters requesting information for this literature review. The federal government agencies contacted included the Department of Fisheries and Oceans and Parks Canada. Provincial government agencies contacted included Departments of Fisheries, Water Quality, Transportation, and Resource Management.

Approximately 50 replies were obtained following the initial correspondence. Some responders replied by simply stating that they were interested in the results of the project. Others provided personal experiences or unpublished literature. Many provided additional contacts and referrals to other individuals. Overall, the correspondence was effective for retrieving information that would not have been obtained via more traditional means, such as the library search. Information continued to be received as the activities of the Road Salt ERG became better known.

1.2.4 Report Organization

This report will focus on various aspects of the literature required to conduct the Tier 1-3 Assessments. In brief, these sections are as follows:

Section 2 (Road Salt Characterization) describes salinity in the environment and road salts.

Section 3 (Entry Characterization) describes the natural factors affecting water salinity and the entry of road salt into the environment. This section also synthesizes the results of other ERG members investigating natural levels of salinity in the Canadian aquatic environment and road salt application rates.

Section 4 (Exposure Characterization) is based on a literature review of the effects of road salt applications on the salinity of aquatic ecosystems. It also discusses the effects of road salt on lake meromixis. The results are presented in a more detailed and tabular form in Appendix A.

Section 5 (Effects Characterization, Part 1) discusses the lethal and sublethal effect levels of sodium chloride as determined from laboratory studies. Calcium, potassium, and magnesium toxicity data also are reported. Factors affecting toxicity and limitations of these studies are discussed. Appendix B presents these results in a more detailed and tabular form.

Section 6 (Effects Characterization, Part 2) begins with a brief synthesis of the effects of natural forms of salinity on aquatic environments. It then discusses field studies which have characterized the impacts of road salts on stream, lake, and wetland ecosystems. It also includes studies which indicate that other factors may have affected the elevated salinity levels observed in these studies. Included in this section are the results of studies investigating the impacts of natural brine seepages and increased salinification of streams and lakes on aquatic ecosystems. Appendix C presents these results in a more detailed and tabular form.

Section 7 (Tier 1-3 Assessments) contains the proposed assessment of “toxic” under CEPA. It identifies geographic regions and habitats which appear to be the most vulnerable to road salt impacts. It provides the final estimate and description of the ecological consequence of road salt applications on aquatic environments.

Section 8 (General Discussion) discusses the use of roadways in human society and develops scenarios in which road salts may pose a hazard to the environment. A final brief discussion is included at the end of this section.

2.0 CHARACTERIZATION OF SALINITY IN AQUATIC ECOSYSTEMS AND OF ROAD SALT

2.1 What is Salinity?

Salinity is the total concentration of salts in water. Inland waters usually are dominated by the cations Ca^{2+}, Mg^{2+}, Na^{+}, and K^{+} and the anions HCO_3^{2-}, CO_3^{2-}, SO_4^{2-}, and Cl^{-} (Wetzel 1983). The majority of freshwater lakes range in salinity from 100-500 mg/L and are calcium-carbonate dominated. By contrast, oceans have an average salinity of 35,000 mg/L and are sodium chloride dominated. When road salts (i.e., NaCl, KCl, $CaCl_2$, $MgCl_2$) are added to water, these salts return to an ionic state and directly increase the salinity of the receiving water body.

Inland aquatic ecosystems can be classified according to their salinity (Hammer 1986a). Waters are classified as fresh if their salinity is less than 500 mg/L, subsaline if their salinity ranges from 500 to 3,000 mg/L and saline when salinity is equal to or greater than 3,000 mg/L. Hammer (1986a) also defined three categories of saline waters: hyposaline (3,000 to 20,000 mg/L salinity), mesosaline (20,000 to 50,000 mg/L salinity) and hypersaline (greater than 50,000 mg/L).

2.1.1 The World's Waters

Most of the world's waters reside in the oceans (Table 2-1). The polar ice caps and groundwater are the next largest components of this water followed by freshwater lakes (Vallentyne 1972; Goldman and Horne 1983; Hammer 1986b). Saline lakes and inland seas, at 104,000 km^3 versus 125,000 km^3 for freshwater lakes, account for the next largest fraction in this classification. If the Aral (970 km^3) and Caspian (79,300 km^3) seas are removed from these estimates (Hutchinson 1957), saline lakes and inland seas account for 23,700 km^3 of the world's waters.

Table 2-1: Distribution of water in the world. Source: Goldman and Horne (1983)

Site	Volume (1000 km^3)
Freshwater lakes	125
Saline lakes and inland seas	104
Rivers and streams	1.3
Soil water	67
Groundwater	8,350
Polar ice caps and all glaciers	29,200
Total for land	37,800
Total for atmosphere	13
Total for oceans	1,320,000

It is quite apparent, then, that salinity, especially sodium chloride-dominated salinity, is a natural feature of the Earth's waters. Organisms have become adapted to a salt water life over the millions of years in which the Earth's oceans have existed. It also is apparent that reserves of easily-accessible surface freshwater are a relatively small component of the world's total water supply. Demand on these reserves is increasing with increased population and technological growth and the quality of these waters is being threatened. Road salt is one of a myriad of substances being investigated which potentially is harmful to the aquatic environment and human health.

2.1.2 The Ionic Composition of the World's Waters

The mean global salinity of seawater is 35,000 mg/L (Table 2-2). The global mean for river water is 104.7 mg/L. If nitrate (1 mg/L), ferric oxide (0.67 mg/L), and silica (13.1 mg/L) are included, the average becomes 120 mg/L. In North America, salinity is slightly higher at 132.4 mg/L, primarily due to higher concentrations of calcium and carbonate. Sodium and chloride, on average account for 21.1% and 16.6% of the cations and anions (on a milliequivalent basis) of North America's river waters.

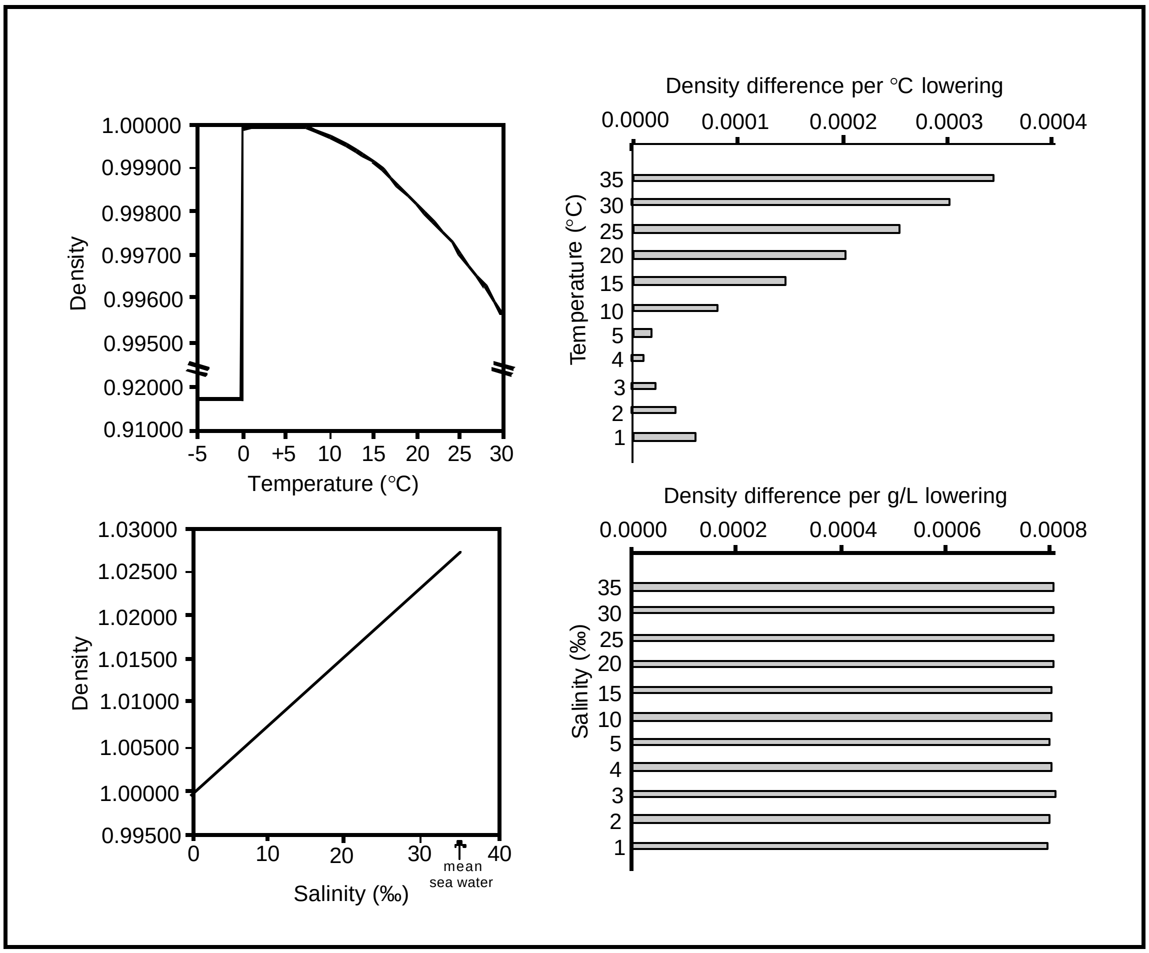
Table 2-2: Mean ionic composition of river water (North America and the world average) and the mean ionic composition of ocean water. Source: Wetzel (1983) and Raymont (1967)

	Ca^{2+}	Mg^{2+}	Na^{+}	K^{+}	CO_3^{2-} (HCO_3^-)	SO_4^{2-}	Cl^-	Sum
N. America (mg/L)	21	5	9	1.4	68	20	8	132.4
Cations (meq)	1.1	0.4	0.4	0.04				1.9
Anions (meq)					1.1	0.4	0.3	1.8
World (mg/L)	15	4.1	6.3	2.3	58	11.2	7.8	104.7
Cations (meq)	0.8	0.3	0.3	0.1				1.5
Anions (meq)					1.0	0.2	0.2	1.4
Oceans (mg/L)	409	1,300	10,770	388	140	2,710	19,370	35,087
Cations (meq)	20.5	108.4	468.4	10.0				607.3
Anions (meq)					2.3	56.4	546.3	605.0

2.1.3 Salinity and the Properties of Water

Salinity is an important factor affecting the density of water. The density of pure water at 4°C is 1.00000 (Figure 2-1). Density increases with increasing salt content to reach 1.02822 at sea water salinity (Wetzel 1983). The density increase between pure and 1 g/L salinity water is 0.00085, between 1 and 2 g/L salinity water is 0.00084, and between 2 and 3 g/L salinity water is 0.00082; density increases in approximately a linear manner with increasing salinity. Therefore, stream water with a salinity of 4 g/L, on entering a freshwater lake of the same temperature, would sink to the lake floor in a density plume. A great deal of energy is required to mix this dense water into the lighter and lower salinity overlying water column.

Figure 2-1: Density as a function of temperature and salinity. The density difference per change in temperature and salinity is also shown (adapted from Wetzel 1983).



Salinity lowers the freezing point of water by about 0.2 ^{o}C per 1 gm/L increase in salinity (Wetzel 1983). While pure water freezes at 0 ^{o}C, sea water freezes at -1.91 ^{o}C. Saline lakes located in cold regions such as Saskatchewan have subzero water temperatures during winter (Hammer and Haynes 1978; Hammer and Parker 1984). Theoretically, a saline lake with a salinity of 100 g/L could reach a winter temperature of -20 ^{o}C without freezing.

Temperature also is an important factor affecting water density (Figure 2-1). Freshwater reaches its maximum density at 4 ^{o}C. Density decreases with lower temperature, with the maximum density of

1.0000 occurring at 4 °C (Wetzel 1983). Above this temperature, the density decreases, but at a non-linear rate. The energy required to mix water is a function of the difference in density between the water masses. The amount of work required to mix waters between 29 °C and 30 °C (density difference = 0.0002979) is about 40 times that required to mix waters between 4 °C and 5 °C (density difference = 0.0000081). Thus, as lake waters warm through summer, they thermally stratify because of this increased resistance to mixing. Any residual salt-laden water at the bottom of the lake will be even more resistant to mixing in summer than winter.

Salinity tends to have a more powerful effect on lake water density than temperature, particularly during the colder months of the year. The density difference between 4 °C and 5 °C is only 0.0000081 and it requires 10 mg/L of salt to give the same resistance to mixing (Wetzel 1983). The amount of work required to mix waters between 24 °C and 25 °C is 30 times that required to mix waters between 4 °C and 5 °C or equivalent to a 300 mg/L increase in salinity. Therefore, a 300 mg/L increase in salinity will have a pronounced effect on water column mixing, particularly if that salinity forms a cold, deep water layer.

2.1.4 Salinity and Lake Meromixis

Most lakes undergo vertical mixing, resulting in an exchange of deep and surface waters. Such exchanges are important for transferring oxygen-rich water to the deeper regions of the lake. In the absence of such exchanges, deep-waters can become anoxic. Vertical exchanges also are important in transferring nutrients regenerated in the deeper regions of the lake to the surface where they become available to the phytoplankton community. Vertical mixing is driven by the winds, which mix surface waters down to depths dependent on lake fetch, surrounding topography, and water temperature. Vertical mixing also is driven by changes in water temperature. In regions where winter water temperatures decline to below 4 °C, spring warming is accompanied by an increase in water density, promoting the vertical exchange of warmer, nearshore waters with offshore, colder and less dense deep waters. In autumn, lake cooling is accompanied by an increase in water density and similar inshore-offshore and vertical exchanges of waters.

Meromixis occurs in lakes when conditions develop which prevent the vertical exchange of surface and deep waters (Wetzel 1983). This occurs when a sufficient gradient exists in salinity to override the effects of seasonal variations in water temperature on density and lake mixing. This chemically induced stratification can occur in three basic ways. Ectogenic meromixis occurs when salt water intrudes into a lake. This typically occurs in coastal regions. Another instance of ectogenic meromixis occurs when saline lakes become overlain with a layer of freshwater from an intense rainfall or from irrigation. Crenogenic meromixis occurs when submerged saline springs enter freshwater lakes. Biogenic meromixis occurs when salts are released in deep waters through the decomposition of organic matter. Nearly all deep tropical lakes are meromictic. Biogenic meromixis increases in frequency in lakes which are very deep and also is common among lakes that are small in surface area, of moderate depth, and are sheltered from prevailing winds. Such lakes are especially common in continental regions which experience long, severe winters where ice cover is prolonged.

Road salt has the potential to impair the normal circulation within lakes when it is introduced as a dense plume. Small, moderately deep lakes will be the most vulnerable to such meromixis, especially in areas of heavy road salt application (e.g., the Maritimes and southern Ontario and Quebec, especially in urban areas). Larger lakes are less vulnerable because the intruding salt-water plumes experience greater dilution as they flow along the lake floor towards the deeper regions of the lake. In addition, larger lakes have greater fetches and hence more powerful wind-driven currents and other water exchanges.

2.2 What is Road Salt?

Road salt consists of various chlorides: sodium, calcium, magnesium, and potassium. In addition, oil field brines containing calcium, sodium, and magnesium chlorides can be used in road salt formulations (Environment Canada 1997a).

Sodium chloride is by far the leading chloride used for road deicing with the total amount used in Canada estimated at 4,240 kt/year (Environment Canada 1997a). The general Canadian application rate for sodium chloride is 130 kg/2-lane km (two-lane kilometre) of highway although the rate varies from 50-300 kg/2-lane km of highway depending on the highway and road conditions. Sodium chloride is applied as a solid and is highly soluble in water with solubility decreasing with decreasing temperature. The cryohydric or eutectic point (the lowest freezing temperature of a solution) is -21.1 ^{o}C (Table 2-3). While sodium chloride is applied at temperatures as low as -28 ^{o}C, at this temperature it acts only as grit; its effective working range is -3.9 to -9.4 ^{o}C (Nova Scotia Department of the Environment 1989). Thus, sodium chloride is most effective as a deicer in those regions of Canada which experience relatively mild winters (e.g., the Maritimes, southern Ontario and Quebec, and British Columbia). Sodium chloride has a low hygroscopicity, meaning it does not readily absorb atmospheric water, and is endothermic, in that it requires external heat in order to dissolve. Most sodium chloride is obtained through rock salt mining or as a by-product of potash (potassium chloride) production.

Calcium chloride is the second most common chloride used with a total annual use rate of 267.8 kt/year (Environment Canada 1997a). It is primarily used as a dust suppressant in the summer on gravel and other unpaved roads, particularly during road construction. Only 26.8 kt/year is used in winter for road deicing at rates substantially lower than that of sodium chloride. As a dust suppressant, calcium chloride works most effectively on gravel roads that have less than 12-15% fine particles such as sand and silt (CH2M Hill 1993) and is applied at a rate of 2.4-5.4 kg/2-lane km of highway. For municipal gravel roads, calcium chloride works most effectively in regions where the relative humidity exceeds 50%. During periods of high rainfall, the road surface may become muddy, slippery and the calcium chloride is leached from the road surface.

Calcium chloride is sometimes mixed with sodium chloride in order to improve deicing and reduce road salt application rates. Calcium chloride can be applied as a flake, liquid, or brine, depending on its formulation. Brines are applied during dust suppression. Calcium chloride is even more soluble in water than sodium chloride with solubility decreasing with decreasing temperature (Table 2-3). The eutectic point is -51.6. While calcium chloride is applied at temperatures below -23 ^{o}C, its effective working range is -3.9 to -31.6 ^{o}C (Nova Scotia Department of the Environment 1989). Calcium chloride has a high hygroscopicity and exists as calcium dihydrate ($CaCl_2 \cdot 2H_2O$) and calcium hexahydrate ($CaCl_2 \cdot 6H_2O$). Calcium chloride is an exothermic salt, releasing heat when it dissolves. Most calcium chloride used in Canada is obtained as a by-product of brine well production. Calcium chloride requires more care in its application as a road salt than sodium chloride because, under certain conditions, its use can create slippery roads.

Potassium chloride is also used as a road deicer and has an estimated annual usage rate of 3.3 kt/year. It is applied as a solid, usually as potash mine tailings or in formulations such as Motech, a by-product of sugar beet processing. There is no data on potassium chloride application rates. Potassium chloride is similar to sodium chloride in its physical-chemical properties although it has substantially higher working (down to -3.9 ^{o}C) and eutectic (-10.5 ^{o}C) temperatures (Table 2-3).

Magnesium chloride is used as a road deicer, deicing additive, and for off-road dust suppression (i.e., on material piles, road shoulders, or transfer material ponds). There is no data on its annual usage rate. Magnesium chloride can be applied as a flake, liquid, or brine, depending on its formulation. Freezegard, a 70% water, 25% $MgCl_2 \cdot H_2O$ and 5% PCI or lignosulfonate formulation, has been tested as a deicer and used at an application rate of 595 L/2-lane km of highway. Brines are applied for dust suppression. Magnesium chloride is more soluble in water than sodium chloride (Table 2-3) and has a lower eutectic temperature (-33.3 ^{o}C). Its working temperature is down to -15 ^{o}C. It is more hygroscopic than calcium chloride and requires greater care when handling in a dry condition; it is mostly effectively used in a dissolved condition (OECD 1989). In Europe, magnesium chloride is obtained as a by-product of the production of potassium compounds. It is highly corrosive making its use controversial.

Oil field brines tend to be used as dust suppressants. There is no data for their annual usage or application rates. Oil field brine is a solution of approximately 30% calcium, sodium, magnesium and chloride, although magnesium also can occur in higher concentrations (CH2M Hill 1993).

Table 2-3: Physical-chemical properties of road salts. Source: Nova Scotia Department of the Environment (1989); Kirchner et al. (1992); Environment Canada (1997a); TAC Salt Management Guide (1999).

	$NaCl$	$CaCl_2$	$CaCl_2 \bullet 2H_2O$ (37%)	35 % $CaCl_2$	$MgCl_2$	KCl
Form	solid	Flake	liquid	brine	solid	solid
Molecular Weight	58.44	110.99	147.02	-	95.21	74.55
Density	2.17	2.15	1.85	1.35	2.32	1.98
Melting Point (^{o}C)	801	772	deh. 176	-7	714	770
Boiling Point (^{o}C)	1,413	1,935	deh. 176	116	1,412	sub. 1,500
Eutectic Temperature (^{o}C)	-21	-51.6	-	-	-33.3	-10.5
Practical Working Temperature (^{o}C)	to -9.4	To – 31.6	-	-	to -15	to -3.9
Ineffective Temperature (^{o}C)	-17	-34.4	-	-	-	- 15.0
Water solubility						
gm/L at 0 ^{o}C	357	371	977	very sol.	543	344
gm/L at 100 ^{o}C	391	425	326	very sol.	727	567

3. 0 ENTRY CHARACTERIZATION

3.1 What Affects the Salinity of Aquatic Ecosystems?

The salinity of aquatic ecosystems is affected by natural forces and anthropogenic activities (Wetzel 1983; Hammer 1986b). Natural forces, which have been occurring for hundreds of millions of years, have resulted in the world's oceans having an average salinity of 35,000 mg/L while the world's rivers have an average salinity of only 105 mg/L. Anthropogenic activities are more recent and include the addition of salts through various discharges, increased erosion, and the concentration of salts in a water body through the reduction of water inputs (e.g., through diversions).

3.1.1 Natural Forces

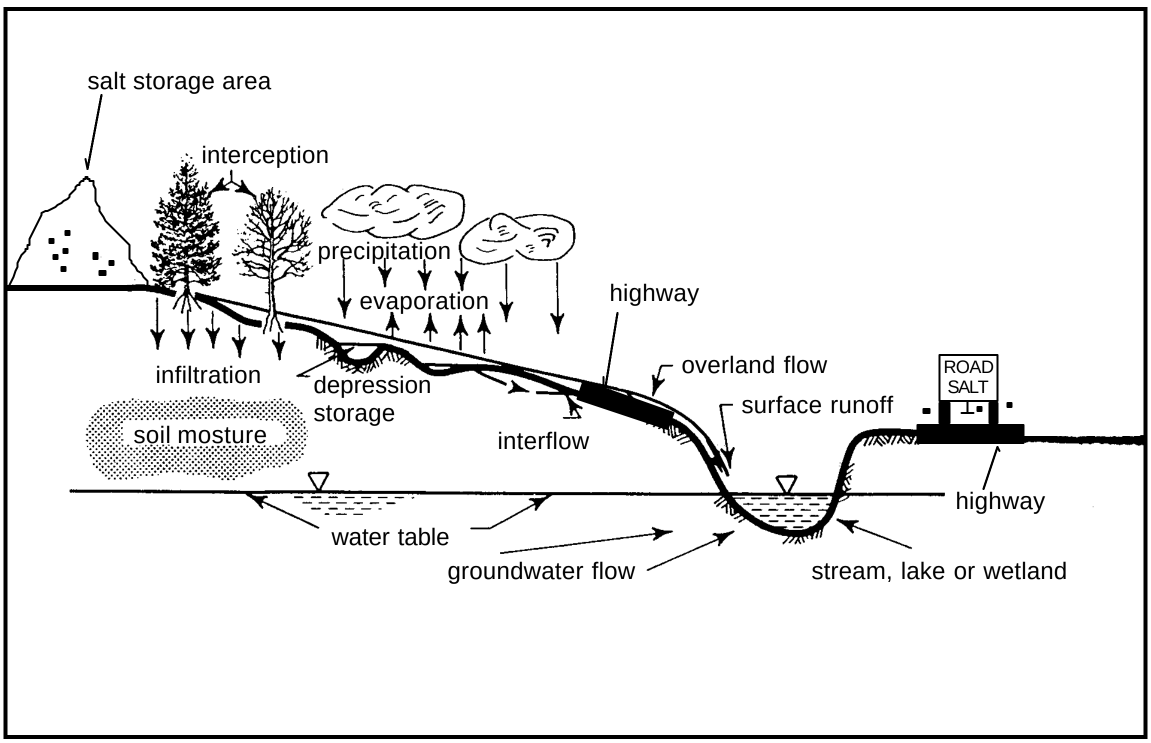
Salts naturally enter surface waters through many pathways of the water cycle, including direct precipitation, stream inflow, overland runoff, and groundwater inputs (Figure 3-1). Rivers and streams pick up salts as they erode rocks or travel through soils, subsequently transporting salts to lakes and wetlands. Different rock types yield different amounts of salts. Igneous rocks and their associated soils generally have lower chloride concentrations than shale and limestone (Pringle et al. 1981). Groundwater can enter rivers and lakes directly by seeping into surface waters or indirectly via springs that originate from groundwater. While rainwater tends to be low in dissolved salts, precipitation can carry salts inland from the oceans. Underwood et al. (1986) identified sea salt as the major source of total ions present in both the precipitation and inland lakes of Nova Scotia, Canada.

Evaporation from surface waters can increase salt concentrations by reducing the volume of water, especially in arid and semi-arid regions where evaporation exceeds precipitation (Wetzel 1983). If evaporation is high enough it can result in the crystallization and subsequent sedimentation of mineral salts. Salts can also accumulate in lakes when outflow water is restricted (Wetzel 1983). Conversely, seasonal rains provide rivers and lakes with low salinity water. While seasonal inputs of rains do not have a pronounced affect on the salinity of freshwater systems, they can significantly lower the salinity of shallow, saline lakes; this can cause stress in organisms adapted to these highly saline waters (Goldman and Horne 1983). This freshwater input can also result in the saline lake becoming meromictic.

Gibbs (1970) conducted a classic investigation of the factors affecting the salinity of the world's surface waters. He noted that salinity is controlled by rock dominance, atmospheric precipitation, and by evaporation and salt precipitation processes. Fresh waters where the salinities are most strongly determined by rock dominance tend to be calcium and bicarbonate dominated and their salinities are close to the world average. Waters draining sedimentary rocks are dominated by calcium, followed by magnesium with much lower proportions of chloride and potassium and by carbonates with lower proportions of sulfate and chloride (Wetzel 1983). Drainage from igneous rocks tends to have a lower salinity (i.e., < 50 mg/L) because of the lower dissolution potential for these hard and often well-weathered materials. In these waters, calcium again dominates, but is followed by sodium, and then magnesium and potassium. Moreover, chloride tends to be the dominant anion followed by sulfate and carbonate.

Rainwater, like sea water, is dominated by sodium and chloride ions (Gibbs 1970). As aquatic systems become dominated by precipitation, salinity decreases and the dominance of sodium and chloride ions increases (Gibbs 1970). For example, tropical regions such as the Amazon are low salinity, sodium-chloride dominated systems. Lakes and rivers located near marine systems also may receive enriched sodium and chloride concentrations from airborne sea spray that becomes incorporated into rainfall. For New England coastal lakes, the sea spray influence occurred primarily within 60 km of the coast (Sullivan et al. 1988). Such sodium-chloride dominated systems typically are low salinity systems.

Figure 3-1: Simplified representation of the major pathways of the runoff phase of the hydrological cycle (adapted from Wetzel 1983).



Another factor affecting lakes is evaporation, which causes increases in salinity, and, at the same time, mineral precipitation, causing changes in the ionic ratio (Gibbs 1970; Wetzel 1983). As calcium and magnesium carbonates precipitate with increasing salinity, the proportion of sodium and chloride increases. Calcium sulfate is less soluble than sodium chloride and tends to precipitate first (Birkeland and Larson 1978). Thus, oceans and seas are sodium chloride dominated. In addition, there are many naturally-occurring saline lakes in many regions of the world, including Canada, the United States, Australia, Ethiopia, and Mongolia (see Section 2 and 6).

Estuaries, where marine and fresh waters meet, form another category of sodium chloride-dominated saline systems. Sea waters may penetrate some distance inland under two driving forces – tides and entrainment. The Pacific Ocean may penetrate 25 km or more upstream of the Columbia River during high tide and low river flow (Pruter and Alveson 1972).

3.2 The Salinity of Canada's Surface Waters

The ionic concentration and composition of most of Canada's surface waters are rock dominated, with calcium and carbonate ions predominating and low total dissolved solids concentrations. In general, the lowest salinity waters tend to be found in lakes and rivers situated on the Canadian Shield, particularly in the maritime regions where precipitation rates are high. However, even within these regions, there can be small variations in ionic concentrations and composition of lake waters (Table 3-1).

Table 3-1: Mean ionic composition (mg/L) and conductivity (μS/cm) of rock-dominated lake waters in a series of lakes north of Great Slave Lake, NWT and in Wood Buffalo National Park, Alberta. Sources: [1]Pienitz et al. (1997) [2]Moser et al. (1998)

	Ca^{2+}	Mg^{2+}	Na^{+}	K^{+}	CO_3^{2-} (HCO_3^-)	SO_4^{2-}	Cl^-	Cond.
NWT[1]								
Boreal forest	7.5	-	2.2	1.1	6.1	1.6	2.2	52.5
Forest-tundra	2.0	-	0.8	0.6	1.3	1.0	0.7	17
Arctic tundra	0.7	-	0.4	0.4	1.2	0.9	0.6	9.5
Alberta[2]								
Muskeg	44.5	12.5	2.3	1.1	32.4	11.0	1.4	221
Sinkhole	51.6	22.1	4.4	2.5	37.3	47.6	2.4	326
Shield	4.9	1.7	2.1	0.4	3.6	2.9	3.8	38

The Laurentian Great Lakes present another example of regional geological influences in affecting salinity (Table 3-2). Lake Superior, located almost entirely in the Precambrian Shield, has a considerably lower salinity than lakes Ontario and Erie, located in sedimentary rocks. Anthropogenic activities also account for some of the regional differences in salinity (Beeton 1969).

Table 3.2: Total dissolved solids and ionic composition (mg/L) of the Great Lake waters. Source: Hough (1958)

Lake	Ca^{2+}	Mg^{2+}	Na^{+} + K^{+}	CO_3^{2-}	SO_4^{2-}	Cl^{-}	TDS [1]
Superior	13.5	3.2	3.3	28.5	2.2	1.1	60
Huron	24.2	7.0	4.4	51.0	6.2	2.6	108
Michigan	26.2	8.3	4.7	58.4	7.3	2.7	118
Erie	31.2	7.6	6.5	59.5	13.1	8.8	133
Ontario	31.7	7.4	6.4	61.2	12.3	7.9	134

[1]Other contributions to TDS are SiO_2 and Fe_2O_3

A more detailed presentation of information on ionic concentration and composition of Canada's rock-dominated surface waters may be found in Mayer et al. (1999) and various reports and publications cited therein.

3.3 Naturally Occurring Saline Lakes

Saline lakes in Canada occur in closed drainage basins (i.e., without an outlet) in regions where evaporation rates exceed precipitation rates. These saline lakes are primarily located in two regions: 1) the Canadian Prairies, specifically in the provinces of Saskatchewan and Alberta, and 2) the Southern Interior Plateau of British Columbia between the Coastal and Columbia Mountain ranges (Hammer 1984). These two regions are in the rain-shadows of the Rocky and Coastal Mountain Ranges, respectively.

Some saline lakes are found in Manitoba. For example, many saline marshes, seeps, and springs are located along Lake Winnipegosis, particularly along its western shore (McKillop et al. 1992). The sodium chloride originates from the dissolution of the Devonian Prairie Evaporite formation. Salinities at the 23 sites investigated in 1988 and 1989 ranged from 1,600 mg/L to 60,600 mg/L (McKillop et al. 1992).

Saskatchewan has the greatest number and volume of saline lakes in Canada including approximately 500 saline lakes with areas greater than one square kilometer (Hammer 1986a). Fifty-seven saline lakes with salinities from 3,000 to 342,000 mg/L have been investigated in the province (Hammer 1984). Many were dominated by sodium, magnesium and sulfate (Hammer 1984). Chloride, sodium, and calcium concentrations ranged from approximately 100 mg/L to 18,000 mg/L, 500 mg/L to 30,000 mg/L, and 50-700 mg/L respectively. Saline lakes are much less common in Alberta than in Saskatchewan, with most located in the Provost and Hanna regions. Hammer (1984, 1986a) classified approximately 30 lakes as saline and, like Saskatchewan, many were dominated by sodium-magnesium sulfate.

Saline lakes in British Columbia tend to be very small (maximum area = ~1 km^2) and shallow (maximum depth = ~20 m) (Hammer 1986a). The salinities of seventeen saline lakes in the Southern Interior Plateau of British Columbia ranged from 2,600 to 45,800 mg/L (Hammer and Forro 1992). However, within this region are sodium bicarbonate and magnesium sulfate brines, as well as, lakes with intermediate ionic composition (Cumming and Smol 1993). Saltspring Island, off the southwest coast of British Columbia, has many naturally saline springs. These springs originate from a source at least 1000 m deep and are distinct in chemical composition from the surrounding seawater (Ring 1991). The springs are hypersaline, with average salinities approximately 2.2-fold that of sea water.

Saline lakes can also be found in the Northwest Territories. Garrow Lake, located 3 km from the southern tip of Little Cornwallis Island, is a hypersaline lake in the Arctic (Ouellet et al. 1989). It has an area of 418 ha and a maximum water depth of 49 m. The salinity of the bottom water is extremely high (90,000 mg/L or nearly 3 times that of the ocean), which prevents complete vertical mixing, thus making Garrow Lake a meromictic lake. The deep, hypersaline water is believed to have been derived from isostatically-trapped marine waters.

3.4 Pathways for Road Salt Entry into Aquatic Ecosystems

Humans augment the quantity of salts in surface waters through the release of industrial effluents and domestic sewage as well as the application of road salts. The salts may be directly released to streams, lakes or wetlands or may enter surface waters indirectly via various pathways of the water cycle (e.g., precipitation containing atmospheric contaminants, leaching into groundwater, etc.). Other important processes include the physical disturbance of watersheds, which results in the increased mobilization of some salts, and the evaporative increase in salt concentration through water diversions. The increased salinification of waters is a growing problem in a number of semi-arid regions of the world where increasingly large volumes of water are being diverted for agricultural, industrial and urban needs (e.g. Hart et al. 1990; Leland and Fend 1998; Williams 1987).

Road salts may enter aquatic ecosystems along several pathways (Fig. 3-1), which are as follows:

1) Road salts may enter solution soon after application and are then carried by runoff into drainage ditches, streams, wetlands, etc. For example, Crowther and Hynes (1977) recorded a pulse in chloride concentrations of 1,770 mg/L Cl^- in Laurel Creek (southern Ontario) which was caused by road runoff. Similarly, chloride concentrations in a drainage ditch near Jamesville, New York (south of Lake Ontario) increased from 20 mg/L Cl^- to a maximum of 5,550 mg/L Cl^- following road-salt application and runoff (Champagne 1978). Several factors affect the concentration of road salt runoff into aquatic ecosystems. These include: a) the length of major road treated, b) the amount of salt applied prior to the thaw period, c) road drainage pattern and topography, d) discharge rate of the receiving stream, e) degree of urbanisation, f) rate of rise and duration of temperatures above freezing, and g) precipitation (Scott 1981).

2) Rain can wash salts out of salt-contaminated soils and into surface waters long after salts are originally applied to the highway. Road salt applied to highways within the watersheds of the Black Creek and Don River (near the northern boundary of Metropolitan Toronto) entered the soil and increased both chloride and sodium levels for distances of approximately 15 m on either side of the highway pavement (Scott 1980a). During the summer months, some of the salt stored in the roadside soils leached into Black Creek and caused elevated salt levels of up to 405 mg/L (Scott 1980b). Concentrations were not as high as those observed in winter (up to 6,000 mg/L), but they persisted for most of the summer.

3) Ecosystems close to roads can be contaminated by salt spray created by moving vehicles. Such aquatic systems include ditches, streams, and wetlands. The maximum distance salt migrates away from the road due to moving traffic is directly proportional to the velocity of the vehicle. McBean and Al-Nassri (1987) found that salt from roads could be sprayed up to 37 m away from the highway by vehicles traveling 100 km/hour. Wind, road gradient and geometrical features of the road also affect salt migration (McBean and Al-Nassri 1987).

4) Urban salt-contaminated snow is often removed from city streets and deposited in other areas where contamination of aquatic ecosystems may occur. In Montreal, Quebec, snowstorms of more than 10 cm require snow-loading and removal operations (Leduc and Delisle 1990). The snow is dumped either at a quarry, down a sewer chute, or at a surface site. Snow also was dumped into the St. Lawrence River until 1986, when this method of snow disposal was prohibited by new regulations (Delisle et al. 1995). However, between 1997 and 1998, Montreal dumped 19.3% of its snow into the St. Lawrence River with the remainder being deposited into quarries (37.7%), surface storage (25%) and into sewer lines (18%) (Delisle and Deriger 1999). Similarly, snow dumping into coastal lakes in British Columbia has been identified as altering the water chemistry of these lakes (Warrington and Phelan 1998).

5) Runoff from salt-storage areas can result in significant quantities of salt entering both ground and surface waters. Ohno (1990) found that runoff from unprotected sand-salt piles in east-central Maine resulted in chloride concentrations up to 13,500 mg/L in surrounding surface waters. In Canada, Morin and Perchanok (2000) estimate that there are 1,300 patrol yards at the provincial level and an unknown number of municipal, county, and privately operated yards. The average amount of salt stored per patrol yard ranges from 1,275 tonnes to 3,753 tonnes within each province. These salts are stored inside igloos, domes, silos, and sheds, and outside on gravel sites, gravel pits, highway maintenance yards, on concrete, tarp, or firm ground (Environment Canada 1997b). Best storage management practices (indoor storage) results in an estimated salt loss of only 0.2% while outdoor storage can result in loss rates of 22% and higher (Snodgrass and Morin 2000). Poor management practices can therefore result in contaminated ground water and adjacent terrestrial and aquatic ecosystems.

6) Road salts can leach into groundwater and then enter lakes via the salt-contaminated groundwater. The chloride concentration in Sparkling Lake, Wisconsin, increased from 2.61 mg/L in 1982 to 3.68 mg/L in 1991 due to chloride inputs from contaminated groundwater (Bowser 1992).

3.5 Retention Time of Salts in Aquatic Ecosystems

Retention time is the length of time that a chemical remains in an aquatic ecosystem before it is removed by natural processes, such as being carried away in outflowing water. Retention time can affect the amount of salt accumulated in an ecosystem and the impact the salt has on the ecosystem.

When a pulse of salt is added to a stream, the pulse will travel down and out of the stream in a relatively short amount of time (i.e., days to weeks, depending on the width, gradient and length of the stream) because the water is constantly flowing through the stream. Therefore, streams, in general, have a relatively short retention time, limiting the window of opportunity the pulse of salt has to impact the stream ecosystem. However, if salt is continually added, the window of opportunity for salt to impact the stream is greatly increased. This may occur as salts continue to be leached from road salt contaminated soils through spring and summer and/or if streams are fed by salt-contaminated groundwater.

Drainage ditches may have short or long retention times, depending on their gradient and drainage pattern. If a ditch is steep and water within it drains rapidly to some other watercourse, the retention time is relatively short. However, if the ditch is level, salt-laden water will remain in the ditch and contribute to a long retention time. Evaporation through the summer may contribute to an increase in salt concentration.

Lakes tend to have relatively longer retention times (i.e., years to decades). As a result, even if salt is added in small quantities over a few months every year, some of the salt may be retained each year, leading to a gradual increase in salinity over several years. Longer retention times mean more time is required for salt concentrations to decrease after additions have stopped, therefore, increasing the length of time aquatic biota could potentially be affected.

Like lakes, wetlands may have long retention times. However, concentrations of salts in wetlands may be more greatly affected by evaporation of water than lakes are. Evaporation of water from shallow wetlands will reduce the volume of water, increasing the concentration of salts in the remaining water.

3.6 Road Salt Application Rates

Morin and Perchanok (2000), with the ERG, have provided estimates of road salt application rates on a Canada-wide basis. They estimated that some 2,950,728 tonnes of chloride are used on Canadian roads, the majority as sodium chloride (Table 3-3). Ontario is the largest user (1,148,570 tonnes) followed by Quebec (950,444 tonnes), Nova Scotia (230,182 tonnes) and Newfoundland (135,384 tonnes). Road salt application rates in British Columbia average 93,900 tonnes and 114,641 in Alberta.

Provinces such as Saskatchewan (33,642 tonnes) and Manitoba (46,880 tonnes) and the Yukon (2,139 tonnes) and the Northwest Territories (2,989 tonnes), have lower application rates due to a combination of the very low winter temperatures, which are below the working range for road salt, and the lower density of roads.

Table 3-3: Total chloride use (tonnes) applied in annually on roadways in Canada. Source: Morin and Perchanok (2000)

Province	Total chlorides
British Columbia	93,900
Alberta	114,641
Saskatchewan	33,632
Manitoba	46,880
Ontario	1,148,570
Quebec	950,444
New Brunswick	173,896
Nova Scotia	230,182
Prince Edward Island	18,061
Newfoundland	135,384
Yukon Territory	2,139
Northwest Territories	2,989
Total	2,950,728

Morin and Perchanok (2000) also determined that application rates vary between provinces, as well as within each province. Highest rates (3–11 kg/m^2 provincial saltable road) are in southern Ontario, Quebec, and in Alberta. The next highest rate, 1.5-3.0 kg/m^2 saltable road, are in most of the Maritime provinces, western Ontario, northwest Alberta, and the south interior of British Columbia. Rates of 1.0–1.5 kg/m^2 saltable road are in parts of Ontario, Alberta, British Columbia, and New Brunswick. This study provides useful information in identifying which regions of Canada are potentially the most vulnerable to adverse impacts of road salt on aquatic ecosystems.

4.0 REGIONAL EFFECTS OF ROAD SALT APPLICATION ON STREAM, LAKE, AND WETLAND ECOSYSTEMS IN CANADA

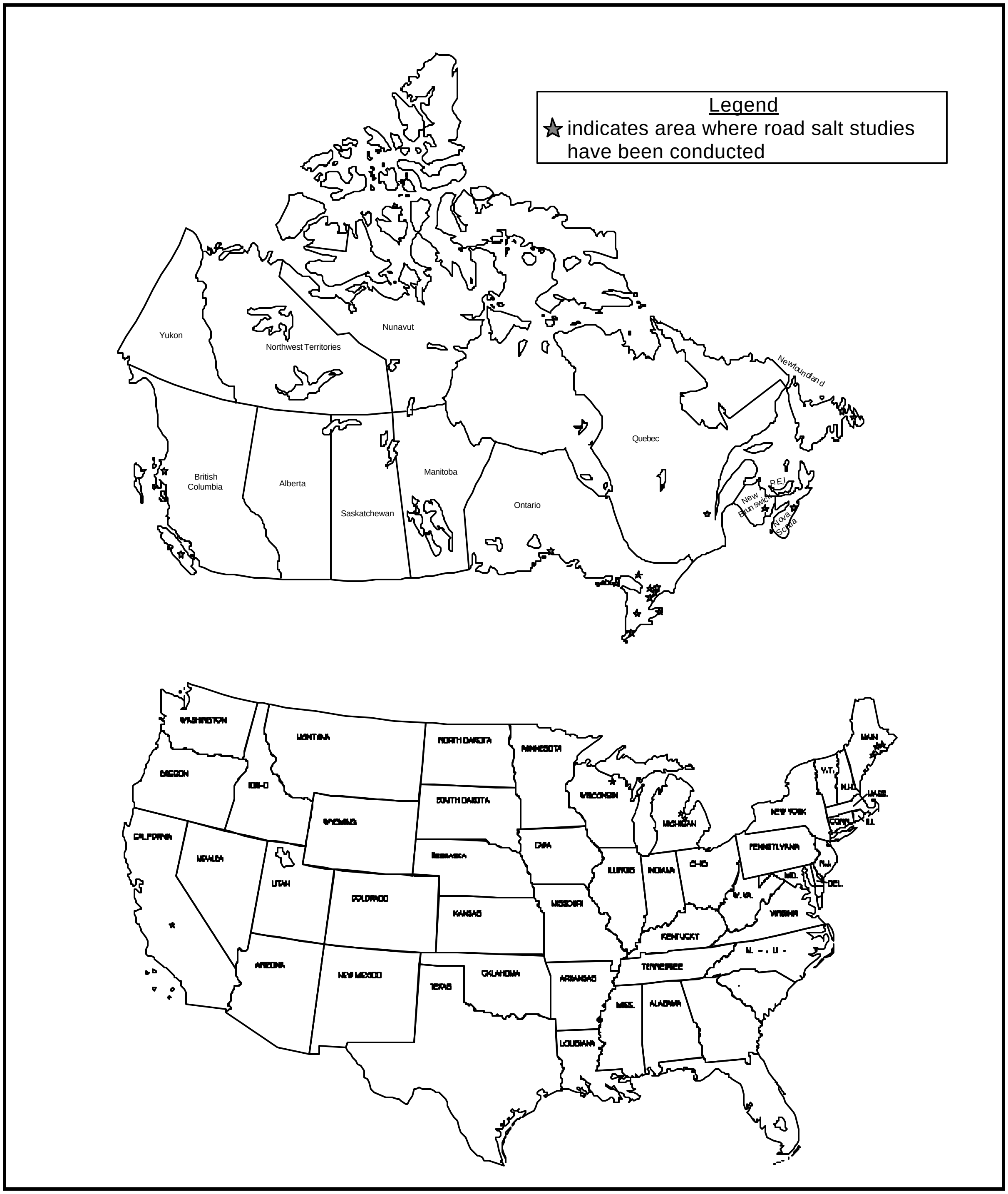
4.1 Introduction

This section reports the findings from studies investigating the effects of road salt on aquatic ecosystems in various regions of Canada as well as studies conducted in the United States. The properties considered in this section are increased salinity and the induction of meromixis in lakes. Meromixis occurs when vertical mixing of lake waters is prevented by strong density gradients resulting from pronounced chemical gradients. The purpose of this section is to generate the real-world chloride data used in the Tier 1-3 Assessments.

The results of the various studies are summarized in the text below. Studies investigating biological effects of road salts are further discussed in Section 6. Data is summarized in Appendix A. This appendix is designed to provide, in tabular form, a description of the ecosystem response to the salt loading, the duration of the response, the baseline or upstream concentrations, the new concentration after loading, and the duration of the new concentration. It also is designed to describe the ecological characteristic or species affected and the location of the effect. Unfortunately, values seldom were provided for the main loadings into the ecosystem for the various studies cited in this report. Nor was information provided on the composition of the applied road salt although most studies were based on road salt applied as a deicer.

The results of this literature review are presented by type of habitat affected – streams, rivers, wetlands, small lakes, meromictic lakes, and large lakes. The data is organized by geographic region, proceeding from the Maritimes and moving west to British Columbia. A surprisingly low number of studies were located during this literature review. Studies conducted in the United States are included to broaden the data sets. Highlights from e-mail and other correspondence also are included in this section. Locations where studies were conducted are located on Figure 4-1.

Figure 4-1: Locations of road salt toxicity studies in Canada and the United States.



4.2 Streams

4.2.1 Maritime Regions of Canada (Newfoundland, Nova Scotia, New Brunswick, and Prince Edward Island) and the United States.

No studies were found which documented the impacts of road salt application on Canadian maritime streams during the initial literature search. More recently, Arp (2001) reported on the impacts of leakage from a road salt storage depot on the chloride concentration and conductivity of nearby ditch, stream, and pond waters. This study was conducted in New Brunswick. Chloride concentrations near the depot were high, ranging from 3,000 to more than 10,000 mg/L. Chloride concentrations, while elevated, were lower (100 to 300 mg/L) in a nearby stream. Highest chloride concentrations were observed in summer due to reduced dilution because of lower precipitation and the higher evaporative losses of water.

Boucher (1982) investigated the effects of road salt runoff on Penjajawoc Stream in Bangor, Maine which passes through a commercially developed area. Seven sites which received road runoff from the development were investigated. Upstream of the development, chloride concentrations averaged 5-15 mg/L while sodium concentrations averaged 15-30 mg/L. Chloride concentrations at sites receiving road runoff averaged 10-50 mg/L and sodium concentrations averaged 5-12 mg/L over 1979-1982. Peak concentrations (e.g., 621 mg/L Cl^- and 406 mg/L Na^+ in 1979) occurred following runoff events and were short in duration (the precise time was not given, but would appear to have been in the order of hours to days). No estimate was provided of the sodium chloride loading to the stream, nor was the length of the stream affected noted.

Mattson and Godfrey (1994) investigated road salt contamination of 162 streams in Massachusetts. A multiple regression approach was used to predict sodium concentrations on the basis of four classes of roads and atmospheric deposition; this model explained 68% of the variance in sodium concentration. Highest salt loadings were associated with interstate (22,500 kg of salt per km, four lanes salted) and major state (17,700 kg salt per km, 2 lanes salted) highways and the lowest with small roads (4,380 kg per km, one lane equivalent salted) and streets (7,530 kg salt per km, one lane equivalent salted). Sodium concentrations reached or exceeded 50 mg/L in the most heavily impacted streams.

Herlihy et al. (1998) investigated the relationship between stream chemistry and watershed cover in the mid-Atlantic region. He noted that land cover was a good predictor of chloride concentration. Forested areas tended to have lower chloride concentrations than agricultural and urban areas. Moreover, chloride concentration was a good predictor of human disturbance in the watershed. While road salts may have been responsible for some of the elevation in chloride concentrations in disturbed watersheds, other anthropogenic sources may have been important including industry, fertilizer use, animal waste, and sewage (Pionke and Urban 1985; Schnabel et al. 1993; Prowse 1987).

Prowse (1987) investigated the major ion flux through an urban area in the Monks' Brook experimental catchment in Hampshire, southern England. Urban areas accounted for an average of 33.2% of catchment area. Prowse estimated that 258 tonnes of chloride were exported from the urban area versus 4 tonnes from the rural load. Road salt was one factor affecting the high chloride and sodium exports from the urban catchment. Chloride levels were estimated to be 63 times higher in the urban than rural area and high stormflow chloride concentrations were closely related to road salting events. However, other important sources of chloride included sewage and other wastes.

August and Graupensperger (1989) investigated the impacts of deicing programs on groundwater and surface water quality in Maryland. Twenty sample sites were investigated in the Washington, D.C., and Baltimore metropolitan areas. In the worst case studied, maximum observed sodium and chloride concentrations were 54.5 mg/L and 128 mg/L respectively. Although relatively large amounts of road salt are used during winter, the Maryland climate has an important role in diluting these salts in receiving waters. There are numerous thaw events that prevent the large accumulation of road salt in snowdrifts and frozen soils. Therefore, there are frequent releases of dilute pulses of salt-laden water rather than a few pulses of highly concentrated water.

Harned (1988) investigated the effects of highway runoff on stream flow and water quality at several sites in the Sevenmile Creek Basin, North Carolina. Chloride concentrations in two creeks near highway I-85 ranged from 9.3-320 mg/L and 5.0-2,500 mg/L; mean concentrations were 70 and 1,100 mg/L respectively. High chloride concentrations were related to road salt usage. Mean chloride concentrations at sites further away from the highway ranged from 2.9-10 mg/L.

4.2.2 Central Canada (Quebec and Ontario) and the United States

Crowther and Hynes (1977) investigated the effects of road salt on Laurel Creek that passes through the town of Waterloo in southern Ontario. Peak chloride concentrations of 680 mg/L were observed in February 1974 at one station located near a major road and a maximum of 1,770 mg/L in February 1995.

Black Creek, located in the northern boundary of Metropolitan Toronto, Ontario, exhibited elevated salt concentrations throughout the year resulting from applications of deicing salts to roads in winter (Scott 1980b). In general, high concentrations were associated with thaw periods in winter and early spring; the maximum chloride concentration was ca. 250 mg/L. Chloride concentrations then declined with the high flow rates associated with spring melting. However, once flow rates declined following the spring freshet, summer chloride concentrations remained at levels that were greater than background levels of 50-100 mg/L, but less than winter/early spring maximum concentrations. Scott estimated that 580 tonnes of chloride entered Black Creek between November 1974 and April 1975. Black Creek, as approximated from the map and text provided in Scott (1980b), was affected for at least 3 km of its length by these road salt applications.

The Ontario Ministry of the Environment has monitored water quality in a number of streams in the province, including the Toronto Watershed. Long-term monitoring has occurred on the Don River, Etobicoke Creek, Little Rouge Creek, the Rouge River, Highland Creek, Mimico Creek, and Black Creek and has demonstrated highly elevated chloride levels, particularly in winter. For example, three stations were monitored on Etobicoke Creek between 1990-1996 (Table 4.1). All these stations were located in developed areas, often near roads or highways and thus clearly impacted by road salt. The minimum chloride concentration ranged from 0-43 mg/L, the maximum from 2,140-3,780 mg/L, and the mean from 278-392 mg/L. Highest concentrations were observed in February followed by January, December, and March (Figure 4-2). At the one monitoring station on Mimico Creek, minimum, maximum and mean chloride concentrations were 51, 3,470, and 553 mg/L respectively. Highest concentrations were observed in December followed by February, January, and March. Similarly, minimum, maximum, and mean chloride concentrations at Highland Creek were 22, 1,390, and 310 mg/L and at Black Creek were 20, 4,310, and 495 mg/L respectively. Highest concentrations were observed in December followed by February, January, and March. Highest concentrations during the winter months are strongly suggestive of road salt inputs. Water quality frequently (18-62% of the measurements) exceeded the 250 mg/L guideline and often (8-22%) exceeded the 500 mg/L water quality guideline.

Elevated chloride concentrations from road salt runoff may result in contaminated groundwater. Williams et al. (1997) investigated chloride concentrations in 20 springs in southeastern Ontario and found chloride concentrations ranging from 8.1 mg/L to 1,149 mg/L. The higher chloride concentrations found in some of the springs were suspected to be caused by road salt contaminated groundwater. Williams et al. (1999) continued this research, noting that the mean chloride concentration in the Glen Majors Conservation Area was 2.1 mg/L and ~100 mg/L in rural areas. Concentrations in the rural areas increased 21-34% over the November 1996 to November 1997 period. Chloride concentrations in springs near rural areas were higher (>200 mg/L) with a maximum concentration of 1,345 mg/L and a mean concentration of 1,092 mg/L. This spring was adjacent to a highway and bridge. Chloride concentrations at these urban sites also increased between November 1996 and November 1997. Salinity contamination was related to road salt. Williams et al. (1999) noted that spatial patterns in road salt contamination were more readily detected by sampling springs than creeks. This is because chloride concentrations in spring waters exhibited relatively little seasonal variability. In contrast, chloride concentrations in creeks were highly variable seasonally.

Table 4-1: Chloride concentrations (mg/L) in various streams in the Toronto Remedial Action Plan Watershed for 1990-1996. Source: Provincial Water Quality Monitoring Network.

Station Location	# of Obs.	Min.	Max.	Mean	Median	% Frequency Exceedence of Guidelines	
						250 mg/L	500 mg/L
Etobicoke Creek							
at Derry Road	38	10	3,780	278	135	18	8
at Burnhamthorpe Road	40	0	2,670	392	206	40	20
at Highway #2	74	43	2,140	351	208	38	19
Mimico Creek							
at Highway #2	37	51	3,470	553	276	62	22
Black Creek							
at Scarlett Road	38	20	4,310	495	248	47	21
Highland Creek							
at Highway Creek Park	55	22	1,390	310	220	42	13

Overall these three Ontario studies suggest that road salt applications can have significant impacts on the salinity of streams and springs in urban areas of southern Ontario. Studies conducted in similar geographic regions in the United States confirm this.

Figure 4.2: Seasonal (boxplot) chloride concentration in Etobicoke Creek (A), Mimico Creek (B) and Highland Creek (C) watersheds in the greater Toronto area over 1989-1995 (from Provincial Water Quality Monitoring Network).

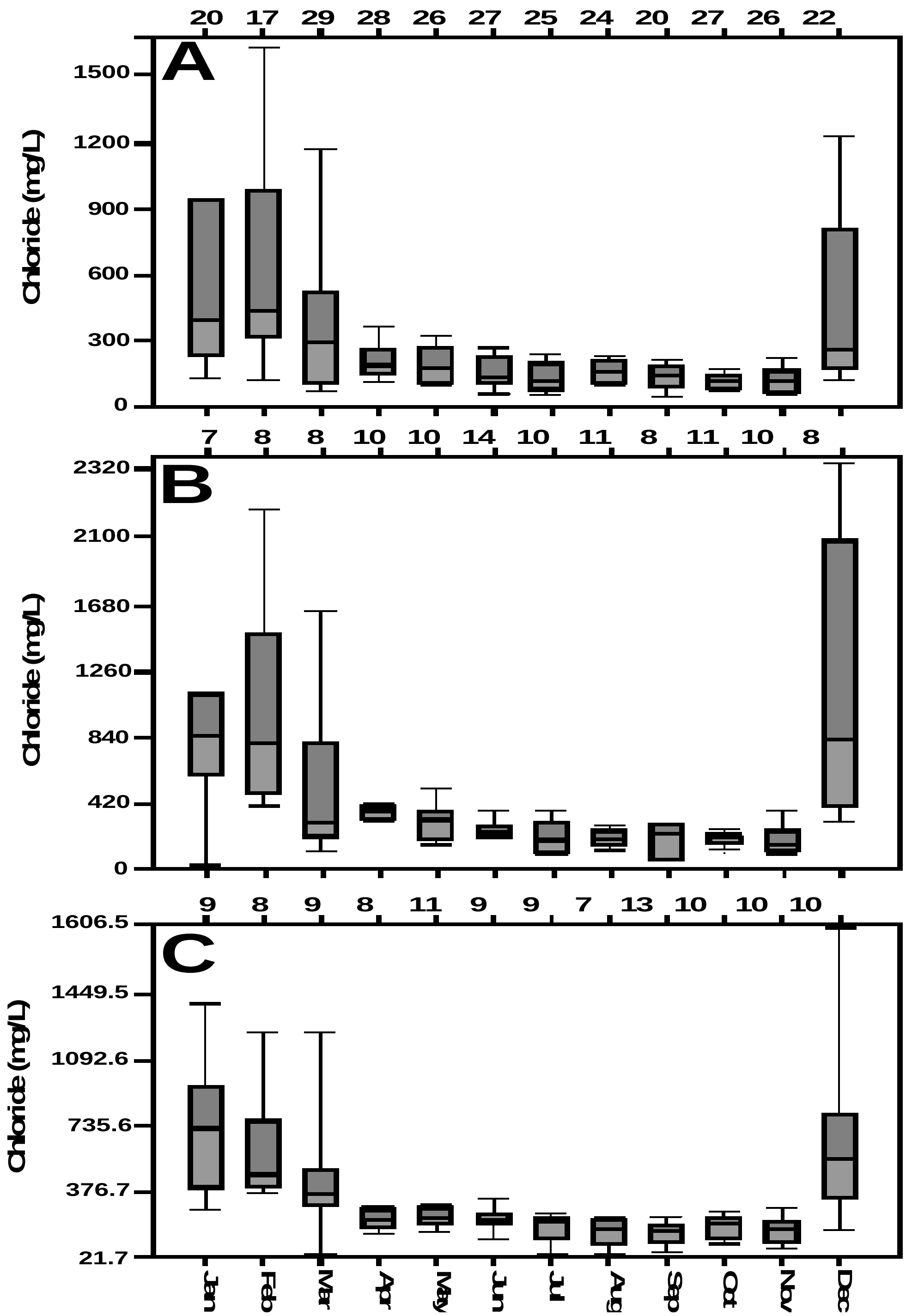
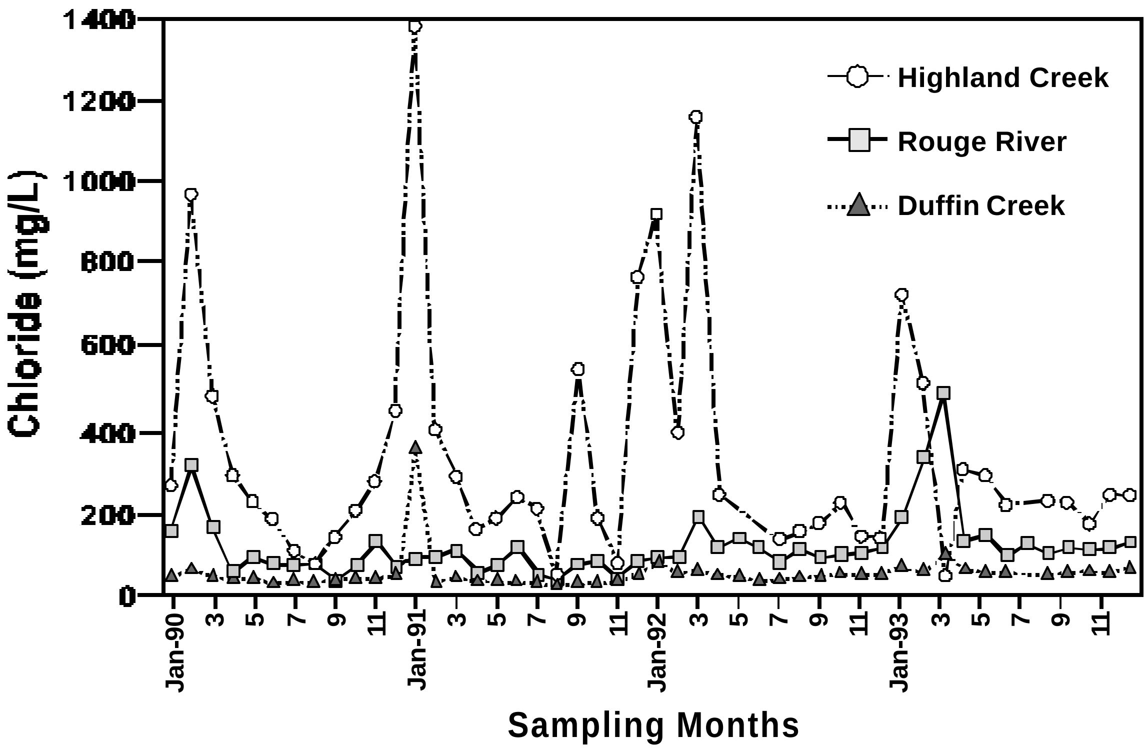


Figure 4-3: Changes in chloride concentrations in surface waters in the greater Toronto area over 1990-1993 (from Williams et al. 1999).



Champagne (1978) investigated chloride concentrations in a drainage ditch in Jamesville, New York. Concentrations varied significantly over the course of a day with temperature change and with rainfall events. The baseline chloride concentration was approximately 20 mg/L. The maximum recorded chloride concentration of 5,550 mg/L was observed in February 1975, following a period of heavy salting and a warming trend during a relatively cold period. This warming resulted in snow and ice melt and the subsequent movement of saline water into the ditch. In March 1975, following road salt application and daytime heating, chloride concentrations in the ditch rose to 340 mg/L, but then declined to 65 mg/L after night-time cooling. Four days later, the chloride concentration rose to 950 mg/L following a rain event. This rain event probably mobilized road salt residues temporarily stored in the vicinity of the roadway into the ditch, resulting in a short-term, but substantial elevation in chloride concentration. The length of ditch affected by the road salt application was not reported.

Demers and Sage (1990) investigated chloride concentrations in four streams near the town of Newcomb in the Adirondack region of northern New York. Four streams were sampled in a forested area along a 2 km stretch of New York State Highway 28N. The streams had a flow rate ranging from 0.01-0.20 m^3/sec and were 0.6-3.2 km long. The overall mean chloride concentration in upstream areas was 0.61 mg/L versus 5.23 mg/L in downstream areas. On occasion, chloride concentrations were up to 66 times greater in downstream than upstream segments of the streams. The stream with the highest downstream chloride concentration (mean 17.1mg/L chloride) flowed parallel to the highway within 12-46 m of the edge and over a distance of 257 m. The source of the chloride was road salt applied during the winter. The duration of the elevated concentrations was not reported, but elevated concentrations were observed 100 m downstream of the bridge. Demers (1992) reported that the Department of Transport estimated that between 8 and 10 MT of salt per lane km and 8.5 and 14 MT of abrasive per lane km are applied annually each year.

Bubeck et al. (1971) investigated chloride concentrations in Irondequoit Creek, near Rochester, New York, as part of a larger investigation of the effects of road salt on Irondequoit Bay. Chloride concentrations in this creek were high, averaging ca. 100 mg/L between March 1969 and November 1970. The average concentration increased to 320 mg/L from December to March. Chloride concentrations in 10 small streams and storm sewers sampled on a single occasion (February 18, 1970) ranged from 700-4,000 mg/L.

A study by Cherkauer (1975) illustrates how heavy road salt residues can continue to effect the chemistry of surface waters months after the last salt application. Sodium and chloride loading to Brown Deer Creek, an urban creek in Milwaukee, Wisconsin, were elevated compared to Trinity Creek, a nearby creek in a rural area with very low road salt application. Both creeks are small with watersheds of 7.5 km^2 for Brown Deer Creek and 9.7 km^2 for Trinity Creek. In October 1974 and 7 months after the last snowfall, there was a rainfall event of 2.2 cm that resulted in increased sodium and chloride loadings to both streams. Sodium loadings to Brown Deer Creek, the urban creek, were 25.7 kg/km^2 and chloride loadings were 37.5 kg/km^2. In comparison, chloride and sodium loadings to Trinity Creek, the rural creek, were only 0.6 kg/km^2 and 0.7 kg/km^2, respectively. The length of stream affected by the road salt application was not reported. Interestingly, the sodium and chloride concentrations were higher in the rural creek. Chloride concentrations at base flow averaged 170 mg/L for Trinity Creek versus 166 mg/L for Brown Deer Creek. At peak flow, chloride concentrations averaged 80 mg/L and 27 mg/L, respectively. Lower chloride concentrations in Brown Deer Creek, despite higher road salt application, were related to greater runoff from urban surfaces; Trinity Creek chloride concentrations were elevated as a result of greater infiltration of rural surfaces. In addition, street cleaning, lawn watering, and storm sewer inputs were important in the urban area. Sodium concentrations averaged 109 mg/L at base flow and 39 mg/L at peak flow for Trinity Creek and 102 mg/L and 19.5 mg/L respectively for Brown Deer Creek.

Smith and Kaster (1983) investigated chloride concentrations in Sugar Creek, Wisconsin (56 km southwest of Milwaukee) as part of a larger investigation of the effects of highway runoff on a rural stream ecosystem. Approximately 7,000-8,000 vehicles travelled on the highway each day. Peak salt concentrations of 53 mg/L chloride and 28 mg/L sodium were recorded during the March and April 1981 snow-melt events. Nickel (0.35 mg/L), iron (1.12 mg/L), and lead (0.15 mg/L) concentrations also were elevated, but baseline concentrations were not reported. Monthly discharge rates averaged 0.7-0.8 m^3/sec at three stations investigated along the creek.

4.2.3 Prairie Provinces of Canada (Manitoba, Saskatchewan, and Alberta), British Columbia and the United States.

No studies were found on the effects of road salts and their additives on the stream ecosystems of the Prairie Provinces and British Columbia. However, a number of studies were found for similar geographic regions in the United States.

Hoffman et al. (1981) investigated chloride concentrations in various streams in the central Sierra Nevada Mountains, California. Road salt was applied from December to April. Streams crossing heavily salted interstate highways were found to have sharp increases in winter chloride concentrations while creeks crossing smaller highways did not display such a pattern. For example, chloride concentrations upstream of a highway crossing over Billy Mack Creek were less than 1 mg/L while concentrations downstream ranged from approximately 20-70 mg/L chloride over a distance of at least 1.5 km. After road applications stopped in spring, the chloride concentration returned to normal in one month or less. Rivers and streams that ran parallel to salted roadways had chloride concentration patterns similar to those of creeks crossing heavily salted roadway. Chloride concentrations in runoff from salted highways ranged from 0-2,051 mg/L and averaged 170 mg/L.

Gosz (1977) reported increased sodium and chloride concentrations in Rio en Medio, a mountain stream in the Sante Fe Ski Basin, Sante Fe, New Mexico. Between 1971 and 1975, the average yearly concentrations of upstream road salt application areas were 0.3 mg/L for chloride and ranged from 1.77-2.05 mg/L for sodium. However, between 1972 and 1975, in an area downstream of road salt applications, the concentrations ranged from 11.9-17.5 mg/L chloride and 5.02-6.30 mg/L sodium. Highest concentrations occurred from February to April each year. Chloride and sodium loading into Rio en Medio were estimated at 3,675 kg/year and 2,100 kg/year, respectively. The length of stream affected was not reported.

4.3 Rivers

Rivers are large flow systems and, as such, may be expected to be less vulnerable to road salt impacts. Few studies were located specifically investigating the impacts of road salt applications on the salinity of Canada’s rivers.

Arsenault et al. (1985) investigated surface water quality in the Waterford River Basin, Newfoundland. Chloride concentrations at the Donovans Station, located in an industrial park, averaged 70 mg/L in comparison to 17 mg/L at Ruby Line, in the South Brook sub-basin, which had no known industrial sources. High chloride concentrations at the Donovans Station were related to industrial activities, road salting, contamination from two road salt depots, and to sea spray.

Oliver et al. (1974) reported that road salt was the primary factor affecting seasonal increases in chloride concentrations in the Rideau River. For example, in March 1992, chloride concentrations increased from a winter average of 9 mg/L to 57 mg/L following a severe ice storm. Chloride concentrations in the Ottawa River, some 32 km downstream of Ottawa, increased from 9 to 21 mg/L following this same storm.

The Don River, with a watershed of 335 km^2, is strongly impacted by Metropolitan Toronto (Paine 1979). In the late 1970s, 56% of the drainage area was in the Metropolitan Toronto watershed and 44% in the Region of York. Between November 1978 and April 1979, some 54,760 tonnes of chloride were added to the watershed. Paine estimated that 93.8% of the chloride was due to road salt applied to roads, 1.7% from salting by private citizens, and 4.5% due to sewage treatment plants. Paine further estimated that while 46% of the salt was removed from the watershed (the Don River discharges into Lake Ontario), 54% remained, possibly in groundwater. This was viewed as a growing environmental concern. Howard and Beck (1993) discuss the contamination of groundwater by road salt in southern Ontario.

Scott (1980b) investigated the impacts resulting from applications of deicing salts on the Don River at a site in the northern boundary of Metropolitan Toronto, Ontario. In general, maximum concentrations (>1,000 mg/L Cl^-) were associated with thaw periods in winter and early spring. Concentrations declined with April spring melt and then increased after the spring freshet. However, concentrations were less than the winter/early spring maxima, but above background concentrations estimated at 50-100 mg/L Cl^-. It was estimated that 1,112 tonnes of chloride entered the Don River from November to April affecting at least 4 km of the river with elevated chloride concentrations. More recent monitoring (Table 4-2) shows that chloride levels continue to be periodically elevated in the Don River. Chloride levels exceed the water quality guideline of 250 mg/L for 20-23% of the observations made over 1990-1996.

Kersey (1981) investigated chloride concentrations in the Humber River, northwest of Toronto, Ontario. In an area where little road salting occurred, chloride concentrations in February and March 1980 ranged from 13.8-24.1 mg/L versus 17.0-34.8 mg/L in the road-salt affected area. More recent data collected by the Ontario Ministry of the Environment reports chloride data for 3 sites on the Humber River (Table 4-2). At Albion Hills, minimum, maximum, and mean concentrations over 1990-1995 were 31, 96, and 46 mg/L, while downstream at Old Mill concentrations were 0.2, 1,680, and 175 mg/L respectively. Only at the Old Mill site were water quality chloride guidelines exceeded although not as frequently as on the Don River. Chloride concentrations at the East and West Humber River monitoring sites also tended to be low and less variable than at sites such as the Don River.

The Rouge River appears to have been less strongly impacted by road salt than the Don and Humber rivers. Four stations were monitored by the Ontario Ministry of the Environment over 1990-1996 (Table 4-2). Minimum concentrations ranged from 11-37 mg/L Cl^-, maximum from 122-970 mg/L Cl^- and mean from 50-162 mg/L Cl^-. Only occasionally, did chloride exceed the 250 mg/L guideline. Chloride concentrations also tended to remain low at the Little Rouge River monitoring site.

Chan and Clignett (1978) determined that sodium and chloride concentrations in the Niagara River increased seasonally due to the winter use of road salt. The study was conducted near the town of Niagara-on-the-Lake, Ontario. Chloride concentrations were slightly higher in February 1976 (11.2-

12.5 mg/L) than in August 1975, November 1975, and May 1976 (sodium concentrations ranged from 10.1-10.7 mg/L). Chloride concentrations in February 1996 ranged from 22.0-23.7 mg/L versus concentrations of 20.2-21.9 mg/L over August 1975, November 1975 and May 1976. No information was provided on either the sodium or chloride loadings into the river or the length of the river affected.

In a recent assessment of water quality in nearshore and tributary waters of Lake Ontario, the Environmental Monitoring and Reporting Branch (1999) of the Ontario Ministry of the Environment reported that road salt is the largest single source of chlorides entering Lake Ontario from local sources. While typical concentrations were below guidelines for the protection of aquatic life, shock loads of chloride, especially during spring snowmelt, were high enough to harm aquatic life. The highest median chloride concentrations (>55 mg/L) were observed in watersheds in the greater Toronto area. Trends of chloride increase were also observed between 1980-1982 and 1996-1998 at 71% of the monitoring sites.

Table 4-2: Chloride concentrations (mg/L) in various rivers in the Toronto Remedial Action Plan Watershed for 1990-1996. Source: Provincial Water Quality Monitoring Network.

Station Location	# of Obs.	Min.	Max.	Mean	Median	%Frequency Exceedence of Guidelines	
						250 mg/L	500 mg/L
Don River							
at Pottery Road	463	1	2,610	287	173	31	14
E. Don at Bayview and Steeles	41	290	960	158	87	20	5
W. Don at Highway #7	39	10	1,040	227	158	23	10
Humber River							
at Albion Hills	11	31	96	46	36	0	0
at Highway #7	36	28	100	51	47	0	0
at Old Mill	444	0.2	1,680	1,775	108	14	7
East Humber R. at Pine Grove	45	24	97	47	40	0	0
West Humber R. at Claireville	37	33	240	94	80	0	0
Rouge River							
at Box Grove, Markham	48	35	583	110	83	4	2
at Steeles, W of 10th Line	40	30	122	50	46	0	0
at Twin Rivers Drive	57	11	573	114	88	7	2
at Highway #2	56	37	970	162	100	16	5
Little Rouge River							
at Twyn Rivers Drive	56	27	650	81	52	5	2

In the United States, Smith et al. (1987) found a significant relationship between sodium and chloride concentrations in rivers and rates of highway road salt use. Hanes et al. (1970) reported on the results of studies investigating chloride levels in Maine rivers. Chloride concentrations in the Androscoggin and Kennebec Rivers, which flowed through areas of high road densities, increased from <1 mg/L at their headwaters to 15-18 mg/L at their mouth. In contrast, chloride levels increased only to 6-8 mg/L for the

Penoboscot, Machias, and Narraguagus rivers that were in areas of low road density. Peters and Turk (1981) investigated increases in sodium and chloride levels in the Mohawk River, New York, from the 1950s to the 1970s. Sodium increased from 7.9-13.6 mg/L while chloride increased from 8.3-20.4 mg/L. Salts from road deicing were estimated to have contributed 96% of the sodium and 69% in the chloride transport increase.

4.4 Wetlands

No studies were found of the impacts of road salt application on the chemistry of wetlands. Some studies were found on the impacts of salt depots on wetlands and are reported later in this report.

4.5 Ponds and Small Lakes

4.5.1 Maritime Regions of Canada and the United States.

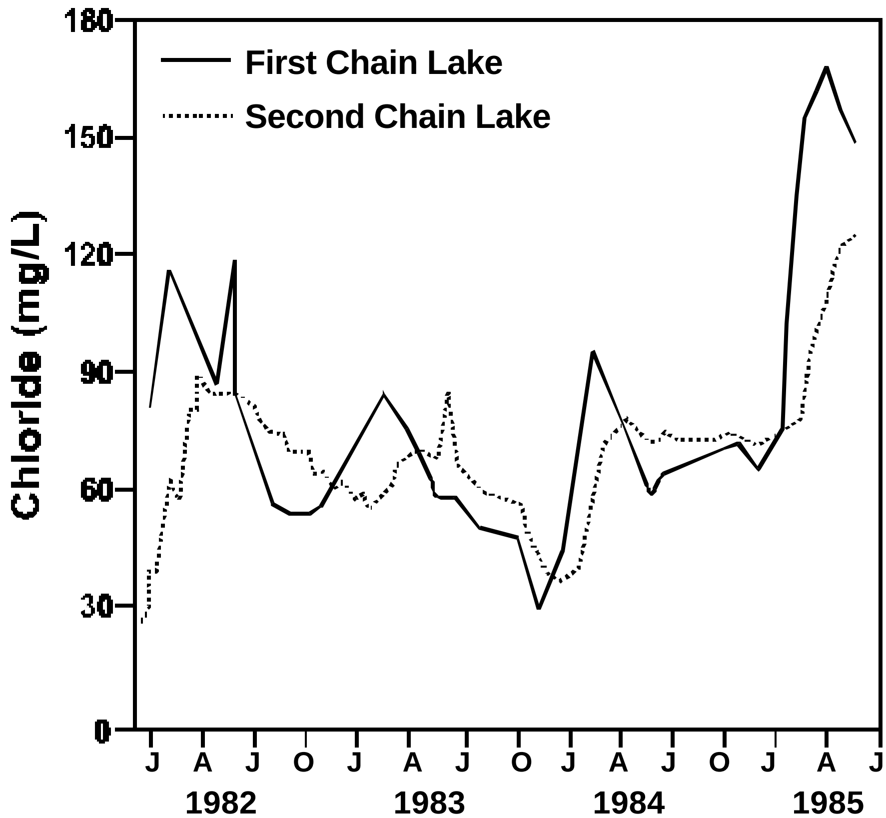
Kerekes (1974) reported that, in spring, chloride concentrations increased nearly 7-fold in Pine Hill Pond, located in Terra Nova National Park, Newfoundland. Chloride concentrations had an average of 14.0 mg/L from April to December 1969 but increased to 94.0 mg/L in March. Road salt applied to the roads during winter was identified as the probable source of this increase. Information was not provided on the duration of the elevated chloride levels nor the total road salt application rate. Pine Hill Pond is a small (surface area = 2.05 ha), shallow (maximum depth = 5.5 m) pond with an estimated volume of 450,000 m^3. The increased chloride concentration represents a 36,000 kg increase in the total chloride content of the lake. On an aerial basis, this corresponds to an increased loading rate of 1.76 kg/m^2 of chloride.

Arsenault et al. (1985) investigated five ponds in the Waterford River Basin, Newfoundland, as part of a surface water quality study during the spring of 1984. Bremigens Pond, located upstream of an industrial park, had an average chloride concentration of 6 mg/L. Barazil Pond, also located upstream of the park, had a slightly higher chloride concentration of 14 mg/L, possibly due to salt contamination from a nearby highway. Branscombe Pond, also located near a highway, had a salt concentration of 18 mg/L. Finally, District Pond, located near a salt depot, had the highest chloride concentration of 24 mg/L. No further information was provided on these ponds.

Road salt contamination was thought to be related to elevated winter and spring chloride concentrations in the Chain Lakes (First Chain and Second Chain), near Halifax, Nova Scotia, when compared to summer concentrations (Thirumurthi and Tan 1978). From June to September 1975, the chloride concentration was approximately 19 mg/L. Concentrations began to increase in November and eventually peaked in May 1976, at 43 mg/L Cl^-. Following this peak in May, the chloride concentration dropped rapidly through the rest of May and June decreasing to 19 mg/L in July. First Chain Lake (surface area = 20 ha, mean depth = 4.1 m, and volume = 810,000 m^3) is somewhat larger than Second Chain Lake (surface area = 16 ha, mean depth = 3.2 m, and volume = 330,000 m^3), but the time trends were similar for both lakes. The ca. 24 mg/L increase in chloride concentrations for both lakes represents an approximate addition of 19,440 kg of chloride to First Chain Lake and 7,920 kg for Second Chain Lake. On an aerial basis, this corresponds to 97.2 g/m^2 for First Chain Lake and 49.5 g/m^2 for Second Chain Lake.

The Chain Lakes continued to be studied through the early 1980s (Hart 1985, 1988). Chloride concentrations in both lakes increased gradually from 1976 to reach a maximum in the mid 1980s of ca. 120 mg/L in First Chain Lake and ca. 170 mg/L in Second Chain Lake (Fig. 4-4). Highest chloride concentrations were observed during spring runoff.

Figure 4-4: Whole-lake chloride concentrations for the Chain Lakes, Halifax County (after Hart 1985).



Keizer et al. (1993) investigated the water quality of 51 lakes in the Halifax-Dartmouth area in April 1991 and compared this data with a similar study conducted in 1980. The most striking difference between the two study periods was an almost doubling of conductivity. In 1980, only 12 lakes had chloride concentrations that exceeded 50 mg/L; this increased to 22 lakes in 1991. In 1980, only 3 lakes had chloride concentrations that exceeded 100 mg/L and this increased to 12 lakes in 1991. In

1981, no lakes had chloride concentrations greater than 150 mg/L while in 1991, 4 lakes had chloride concentrations which exceeded 150 mg/L with one lake reaching 197 mg/L chloride. Increases in chloride, sodium, and calcium concentrations were related to the use of road salt for deicing (sodium chloride) and dust suppression (calcium chloride). The largest increases in salt concentrations were associated with lakes in the more developed watersheds.

Underwood et al. (1986) surveyed 234 Nova Scotia lakes. Background chloride concentration was 8.1 mg/L while the maximum concentration was 28.1 mg/L. The Nova Scotia Department of the Environment (1989) indicated that waters with chloride concentrations above 25 mg/L must have been receiving chloride inputs from anthropogenic sources. In watersheds with numerous highways, road salt would have been the probable source.

In Maine, Hanes et al. (1970) discussed the effects of road salt on a variety of aquatic ecosystems. They noted that farm ponds located near highways had sodium concentrations ranging from 1.4-115 mg/L and chloride concentrations ranging from <1 mg/L to 210 mg/L. Salt concentrations were higher in April 1966 than July 1965. In 1967, chloride concentrations ranged from 1.4-221 mg/L, suggesting that chloride levels were increasing.

4.5.2 Central Canada and the United States.

The Ministry of Transport Quebec investigated the impacts of road salt on Lac a la Truit, by Highway 15 and near Sainte-Agathe-des-Monts. The lake drainage area, estimated at 728 ha, was affected by a 7-km stretch of highway, with a maximum slope of 5%. The lake had a surface area of 48.6 ha, a mean depth of 21.5 m, and an estimated lake volume is 486,000 m^3. In 1972, average chloride concentration for the lake was 12 mg/L. This increased through the 1970s to reach a maximum concentration of 150 mg/L in 1979. This corresponds to an addition of ca. 67,068 kg of salt to the lake. The amount of salt applied to the roads was reduced and chloride concentrations fell through the 1980s to reach 45 mg/L in 1990. While concentrations declined, they remained elevated at 42 mg/L at 3 m and 49 mg/L at 10 m.

In Ontario, published studies dealt with meromictic lakes and are discussed in the next section of this report. However, the Ontario Ministry of Transportation has unpublished data on the mean chloride concentrations (May-October 1995) in eight water bodies in the Humber River Watershed (Scanton 1999). Chloride concentrations averaged 10.6 mg/L at Lake St. George, 26.8 mg/L at Preston Lake, 47.0 mg/L at Wilcox Lake, 60.8 mg/L at Heart Lake, 107.9 mg/L at Claireville Reservoir, 110.3 mg/L at G. Ross Lord Dam, 174.0 mg/L at Lake Aquitane, and 408.9 mg/L at Grenedier Pond. The Humber River, like Highland Creek, is a major urban catchment basin, lying to the west rather than the east of Metropolitan Toronto. Road salt applications may be impacting the Humber River basin as has been observed for the Highland Creek basin (Howard and Haynes 1997).

In an ongoing study, Watson (2000) is investigating the impact of road salt on the water quality of 89 ponds located near roadways in southern Ontario. Samples were collected in the spring and summer

2000. This study is providing evidence that road salt runoff can affect the water quality of roadside ponds and that the effect is dependent upon highway size, among other factors. The mean chloride concentration in ponds located near two-lane roads was 95 mg/L, for ponds located near roads with more than 2 lanes but less than size lanes was 124 mg/L while for ponds located near roads with six lanes or more the mean chloride concentration was 952 mg/L. Maximum chloride concentrations reached 368 mg/L, 620 mg/L and 3,950 mg/L respectively for these three categories of ponds.

Likens (1985) reported that Mirror Lake, a small lake located in the White Mountains of north-central New Hampshire, increased in salinity over four years (1975/1976 to 1979/1980). Chloride concentrations increased from 0.94 to 2.04 mg/L (a 117% increase) while sodium concentrations increased from 1.22 mg/L to 1.65 mg/L (a 35% increase). These increases were related to runoff from road salt and/or leaching from septic tanks. Mirror Lake has a surface area of 11 ha, a mean depth of 5.75 m and a maximum depth of 11 m. Its watershed is large currently extending over 85 ha versus 103 ha prior to the construction of Interstate Highway I-93 that isolated some of the lake drainage area.

Siver et al. (1996) investigated historic changes in 42 Connecticut lakes sampled in the late 1930s, the mid to late 1970s, and the early 1990s. Since the 1970s, many of the lakes located in residential watersheds have increased in base cation concentrations by an average of 70 μeq/L, a 9% rise. Much of this increase is due to an increase in chloride concentrations in affected lakes (average increase 90 μeq/L or 3.2 mg/L). Sodium concentration generally increased concurrently with chloride concentrations (60 μeq/L or 1.4 mg/L). Siver et al. (1996) argued that much of this increase was due to road salt applications. Field et al. (1996) continued this study, focusing on changes in total phosphorus and nitrogen. They noted that other factors in addition to road salt could affect an increase in sodium and chloride concentrations. Such factors include converting forest and scrubland to agricultural land, tilling, and application of fertilizers and pesticides (Pionke and Urban 1985).

Sparkling Lake in the northern Wisconsin Lake District experienced an increase in chloride concentration due to contamination from road salt-laden groundwater (Bowser 1992). The morphometry of Sparkling Lake was not reported. The road salt was initially applied to roads above the lake and the salt apparently leached into the groundwater prior to entering the lake. Chloride concentrations in unaffected lakes and groundwater in the area were 0.3-0.5 mg/L versus a chloride concentration of 2.61 mg/L in 1982 and 3.68 mg/L in 1991 in Sparkling Lake. The load of chloride required to produce such an increase in chloride concentration between 1982 and 1991 was calculated by Bowser (1992) to be 1,200 kg/year.

Northridge Lakes in Milwaukee, Wisconsin, are artificial, interconnected, urban lakes with highly elevated chloride concentrations during winter ice-cover (Cherkauer and Ostenso 1976). The average chloride concentration for lakes in the general area was 6 mg/L. In comparison, the chloride concentration in the Northridge Lakes during March, 1975 was ~250 mg/L just below the ice and increased to ~2,500 mg/L at the lake bottom. Thus, these lakes displayed a prominent chemocline during winter ice cover, but vertical mixing of the water column occurred during spring. This may have occurred because the lakes were very shallow with a mean depth of only 2 m and because water inflow during spring was sufficient to erode the deep layer. Cherkauer (1977) later investigated the effects of urban lakes on the quality and quantity of baseflow. While these lakes had little impact of baseflow volume, they impacted chloride concentrations in downstream waters until summer. Chloride concentrations in Beaver Creek remained high (120 mg/L) 2-km downstream of the lake outflow in spring and 1.4 km downstream in summer.

Eilers and Selle (1991) compared conductivity, alkalinity, calcium, and pH data collected at 149 northern Wisconsin lakes over 1925-1931 with data collected over 1973-1983. All parameters increased between the two time periods with the greatest increases associated with increased land development on lake shorelines. Mean conductivity in developed watersheds increased from 38.1 to 47.7 μS/cm versus 14.2 to 15.3 μS/cm. Increased conductivity appeared to be associated with a combination of factors including road salt, cultural eutrophication, and changes in hydrology. The strongest increases were associated with lakes located near highways or paved roads.

4.5.3 Prairie Provinces, British Columbia and the United States

No studies were found on the effects of road salt on lakes in the Prairie Provinces and British Columbia. However, Warrington and Phelan (1998) noted that several coastal lakes in British Columbia have been impacted by snow dumping. One lake is located along the highway between Terrace and Kitimat and another is located between Port Alberni and Long Beach; additional information on the lakes was not provided. These lakes are adjacent to highways with snow and ice problems. They have roadside pullouts along the lake shore that allow for 1) salt-laden snow to be pushed over the bank during removal operations and 2) roadside snow and salt to run directly into the lake during the spring thaw. The result has been an alteration of the normal chemical equilibrium dominated by calcium and carbonate ions to a new equilibrium dominated by sodium and chloride ions.

In the United States, Hoffman et al. (1981) reported elevated chloride concentrations in several mountain lakes (i.e., Putt's Lake, Gold Run Pond, and Summit Pond) in the Sierra Nevada Mountains, California as a result of road salt application. In 1975, Putt's Lake developed a pronounced halocline (vertical gradient in the concentration of a salt) with a surface chloride concentration of 8 mg/L and a bottom concentration of 142 mg/L. Putt's Lake has a maximum depth of only 4 m. These highly elevated concentrations were recorded in early spring (i.e., April and part of May); however, normal spring mixing and flushing occurred, removing the halocline.

4.6 Meromictic Lakes

Some lakes, while developing a halocline, become meromictic with a loss in vertical spring mixing. Regional studies of meromictic lakes are described in the following section. Physical characteristics of these lakes and their watersheds are shown in Table 4-3.

Table 4-3: Meromictic lakes in Canada and the United States.

Lake	Surface Area	Drainage Basin	Mean Depth (m)	Maximum Depth (m)	Reference
Chocolate Lake, NS	82.7 ha	900 ha	3.9	12.2	Kelly et al. 1976
Lake Wabekayne, ON	19 ha	N/A	1.84	N/A	Free and Mulamoottil 1983
Little Round, ON	7.4 ha	1.09 km^2	8.3	16.8	Smol et al. 1983
Irondequoit Bay, NY	6.78 km^2	435 km^2	6.9	23	Bubeck et al. 1971
Ides Cove, NY	11.8 km^2	N/A	N/A	8.8	Bubeck et al. 1995
First Sister Lake, MI	12.9 km^2	N/A	N/A	7.2	Judd 1969
Third Sister Lake, MI	3.8 ha	N/A	8	17.1	Bridgeman et al. (2000)
Fonda Lake, MI	N/A	N/A	N/A	13	Tuchman et al. 1984; Zeeb and Smol 1991

4.6.1 Maritime Regions of Canada

Chocolate Lake, in Nova Scotia, is a Maritime lake that has been impacted by deicing salts. The lake is small (82.7 ha), shallow (mean depth = 3.9 m, maximum depth = 12.2 m), with an estimated volume of 350,000 m^3 (Kelly et al. 1976). The watershed is large (900 ha), approximately 11 times as great as the lake surface area. Chloride concentrations were highly elevated in the lake between April and August 1975. Average summer chloride concentrations were 207.5 mg/L versus an estimated background value of 15-20 mg/L for non-impacted lakes. Chloride concentrations varied with depth, ranging from 199-224 mg/L at the surface, from 189-217 mg/L at 6.1 m, and from 225-330 mg/L at 12.1 m. Sodium concentrations ranged from 108-125 mg/L, 102-125 mg/L and 116-183 mg/L at the same depths. The gradient in salt concentrations and the absolute concentration of salt in deep waters was large enough to prevent complete vernal vertical mixing of the water column; deep waters became anoxic in summer. The primary source of this excess chloride was attributed to highway deicing salt runoff. Kelly et al. (1976) estimated that 64.2 tons (58,364 kg of road salt representing 35,409 kg of chloride and 22,737 kg of sodium) were applied to the Chocolate Lake drainage basin during the winter of 1974-1975. In contrast, the estimated excess chloride in the lake in the summer of 1995 was 65,625 kg. Factors affecting the excess chloride in the lake were attributed to imprecision in salt application rates, urban inputs via a sewer line, and a gradual increase in chloride concentrations in the lake with incomplete annual flushing. The excess 65,625 kg chloride in the lake represents an aerial excess of loading of 79.4 g/m^2 for the lake.

A similar study was conducted in Williams Lake, Nova Scotia (Underwood and Josselyn 1979). Salt concentrations were about ten times higher than expected. Elevated salt concentrations were related to road salting.

4.6.2 Central Canada and the United States

Winter road salt application affected Lake Wabekayne, a storm-water impoundment in Mississauga, Ontario (Free and Mulamoottil 1983). The morphometry of Lake Wabekayne is as follows: surface area = 19 ha, mean depth = 1.84 m and volume = 35,000 m^3. Chloride concentrations were elevated from approximately February to April (e.g., 282 mg/L in February 1979), particularly on the bottom of the lake. Chloride concentrations decreased to 50 mg/L by August 1979. As a result of the salt-density gradient produced from the winter application of road salt, there was an incomplete mixing of the lake in spring and anoxic conditions developed.

Little Round Lake in central Ontario was affected by cultural disturbances that included urbanization, septic tanks, highway road salt runoff and seepage from a salt storage depot (Smol et al. 1983). This small lake (surface area = 7.4 ha, maximum depth = 16.8 m) apparently became meromictic as a result of road salt runoff and localized storage. Salt concentrations in the monolimnion or deep layer were 58.4 mg/L sodium and 103.7 mg/L chloride, well in excess of that explainable by the natural geology of the region. Road salt additions apparently resulted in the formation of meromixis some thirty years earlier.

Irondequoit Bay, near Rochester, New York, is a small embayment located on the southern shore of Lake Ontario. The morphometry of the bay was reported as follows: surface area = 6.78 km^2, maximum depth = 23.8 m, mean depth = 6.8 m, volume = 45.9 x 10^6 m^3, and retention time = 116 days. The bay has been strongly impacted by road salt application in its watershed (Bubeck and Burton 1987; Bubeck et al. 1971). The concentration of chloride measured in the bay in 1910 was 12 mg/L while concentrations of chloride in the Irondequoit Creek in 1910 were approximately 14 mg/L. Road salt began to be applied in the watershed in the 1940s resulting in an increase in chloride concentrations from ~90 mg/L in 1960 to ~125 mg/L in 1980. A maximum winter concentration of 600 mg/L was recorded in the creek. The exact loadings from the creek to the bay were not reported. However, a maximum load of 192 tonnes/day could be approximated from the maximum concentration of 600 mg/L chloride and the mean annual discharge of 3.7 m^3/s. This increase in chloride loading resulted in incomplete mixing of the bay from 1970 to 1973 when maximum concentrations of chloride were measured in the bay (e.g., surface concentration = 152 mg/L in 1971).

In a later study, Bubeck et al. (1995) investigated the physical and chemical limnology of Ides Cove, a small cove on Irondequoit Bay. The cove has a surface area of 1.18 ha and a maximum depth of 8.8 m and a bedrock sill 50 m wide and 1.5 m deep separates the cove from Irondequoit Bay. The cove is protected from strong prevailing winds by the steep slopes located on its north and west shores. During 1971-1972, the maximum annual chloride concentrations occurred in spring. Epilimnion concentrations at that time ranged from 210-225 mg/L with chloride increasing an additional 80-160 mg/L in the hypolimnion. Following decreased usage of road salt in the basin beginning in 1974, chloride inputs to the cove decreased. By 1980-1982, spring maximum epilimnion chloride concentrations were 140-150 mg/L and chloride concentrations decreased 0-90 mg/L in the hypolimnion. As a result of this, the duration and depth of spring and fall mixing increased between 1970-1982.

The induction of meromixis as a result of road salt application also has been reported for First Sister Lake in Michigan (Judd 1969). This induction occurred in the spring of 1965 and 1967 as a result of a salt-density gradient induced by salt-laden inputs that entered the lake through two drainage pipes. The eastern drain carried runoff from a subdivision and the northern drain carried runoff from a series of highways. Chloride concentrations reached a high of 177 mg/L (Judd 1969). Shortly after this study, the City of Ann Arbor, reduced its use of road salts in the subdivision and chloride concentrations in the eastern drain were reduced (Judd and Steggall 1982). However, chloride concentrations continued to increase in First Sister Lake as a result of increased inputs from the highways, reaching a deep water maximum of 720 mg/L. However, most of this runoff entered the lake through a wetland, resulting in a more diffuse entry of the salt water into the water column runoff. As a result, complete mixing in the lake occurred in the spring of 1981 (Judd and Steggall 1982).

More recently, Bridgeman et al. (2000) reported changes in the general limnology of Third Sister Lake, also in Ann Arbor, Michigan. Chloride levels increased nearly 13-fold from 19 mg/L in 1981 to 260 mg/L in 1999. This increase was sufficient to impact spring vertical mixing and the bottom waters of this lake became anoxic by January, which has impacted benthic communities. Increased chloride concentrations were related to increased road salt runoff from an area that recently experienced increased residential and commercial growth. Third Sister Lake is a small (3.8 ha) lake with a mean depth of 8 m and a maximum depth of 17.1 m.

Fonda Lake, also in Michigan, was adversely impacted by seepage from a salt storage facility which was established in 1953 (Tuchman et al. 1984; Zeeb and Smol 1991). An asphalt pad was constructed in the early the early 1970s, apparently resulting in a reduction in the salinity of this lake. Nevertheless, the average chloride concentration in 1981 remained high at 235 mg/L versus 12 mg/L at Frains Lake and 15 mg/L at Portage Lake, also located in the area. Fonda Lake has a maximum depth of 13 m and is spring fed with no permanent outlet and inlet.

4.6.3 Prairie Provinces and British Columbia

No studies were found on meromictic lakes which were created as a result of road salt contamination in these regions. Meromictic lakes, created by other processes, do occur in these regions.

4.7 The Great Lakes in Central Canada and the United States

The Laurentian Great Lakes have experienced many anthropogenic impacts over the last one hundred years, including increased chloride loadings. Of these five lakes, Lake Erie and Lake Ontario have seen the greatest changes in chloride concentrations over the last 90 to 100 years (Moll et al. 1992). In 1910, the concentration of chloride in Lake Erie was 10 mg/L and maximum concentrations of over 25 mg/L chloride were reached by the late 1960s. While road salt was one source of this chloride, other sources included urban sewage and industrial wastes, particularly from the Detroit River. Following various remedial actions, chloride concentrations dropped to 20 mg/L in 1990. Hanes et al. (1970) reported that road salt accounted for only 11% of the chloride entering Lake Erie in the early 1960s.

Chloride concentrations in Lake Ontario were less than 10 mg/L around 1900 (Moll et al. 1992) and then increased through the decades, exceeding 25 mg/L by the late 1960s. Although remedial actions have been implemented, chloride concentrations in Lake Ontario have not declined. The failure to do so is attributed to the longer retention time of water of Lake Ontario (i.e., 6 years) compared to Lake Erie (i.e., 2.6 years). Sources of chloride to Lake Ontario include road salt and urban sewage discharged to the Niagara River. Ralston and Hamilton (1978) estimated that road salt was the largest single source with 45% (219,730 tonnes per year) of chlorides entering Lake Ontario from local sources. However, this value was only 5% of the estimated 4,620,000 tonnes per year entering the lake from the Niagara River. In a later study, Fraser (1981) estimated that the road salt contributed 1.0 and 1.5 million metric tonnes of chloride annually to lakes Ontario and Erie. This loading represented 20% of the total chloride load to these lakes. Fraser also concluded that the use of deicing salt was not increasing the salt content of lakes Ontario, Erie, Huron, and Superior.

Unlike lakes Erie and Ontario, Lake Huron has seen only minor changes in chloride content (Moll et al. 1992). In 1900, the chloride concentration was 5 mg/L, which increased to 7 mg/L by the late 1960s. The concentration then returned to baseline by 1990. The major source of chloride to Lake Huron is the Saginaw Bay tributaries.

In Lake Superior, chloride concentrations have remained relatively constant at 1 mg/L over the last 200 years (Moll et al. 1992). The low, constant concentration is a result of little to no contribution of chloride from weathering of rock, small municipal and industrial discharges and minor inputs from deicing salt.

4.8 Salt Storage Sites

The effects of salt storage in two meromictic lakes, Little Round and Fonda, were already discussed in section 4.6.2. Two additional studies reported the effects of road salt depots on the salinity of aquatic ecosystems in Canada. Two studies also were located from the United States.

The Water Resources Branch of the New Brunswick Department of the Environment (1978) investigated the quantity and quality of leachate from a highway salt-treated sand pile. These sand piles were not covered, but were underlain by an impermeable butylated rubber lagoon liner. Rainfall was the primary factor affecting the loss of sodium chloride from the pile. During the 1975-1976 study year, 33% of the salt was lost from the sand pile while, in 1976-77, 57.3% of the salt was lost. This represented a total mass of 55 tons of salt in 1975-1976 and 45 tons in 1976-1977. Sodium and chloride concentrations in the leachate reached 37,000 mg/L and 66,000 mg/L, respectively. The study concluded that road salt loss could be substantially reduced by restricting the amount of precipitation which came in contact with the pile.

Arp (2001) investigated the impact of a road salt storage depot located in New Brunswick on nearby waters. Chloride concentrations of 3,000-10,000 mg/L were observed at and near the depot but were diluted to 100-300 mg/L in nearby stream waters. Salt concentrations were lowest during snowmelt and runoff events. Highest concentrations were observed in summer during to lower rainfall and highest evaporation rates. At that time, elevated chloride concentrations were observed in a nearby stream and pond.

As part of a surface water quality study in the Waterford River Basin, Newfoundland, a salt depot located in Mount Pearl was examined (Arsenault et al. 1985). This salt depot appeared to be significant in affecting relatively high sodium and chloride concentrations at a river station located approximately 1 km from the depot. Chloride concentrations ranged from 7.8-178 mg/L (mean = 42.8 mg/L) and sodium concentrations ranged from 6.1-130.0 mg/L (mean = 25.8 mg/L). Sodium and chloride concentrations in a nearby shallow well ranged from 43-180 mg/L and 61-280 mg/L respectively, with highest concentrations observed in February. In 1982, this depot was moved to the Donovans Industrial Park. Chloride concentrations in a nearby shallow well increased from 13-14 mg/L to 43-480 mg/L and sodium from 9.8-16.0 mg/L to 55-1,360 mg/L. District Pond, also located near a salt depot, had the highest chloride concentration of 24 mg/L.

Ohno (1990) investigated the contamination by four sand-salt storage sites on the chloride, sodium, and cyanide contamination of nearby (within 30 m) wetlands in Maine. At the Aton wetland, sodium concentrations at the control site were 6 mg/L versus 16-8,663 mg/L at the affected site; chloride concentrations were 8 mg/L at the control sites versus 14–12,463 mg/L at the affected sites. Concentrations varied seasonally from March to November. High concentrations recorded in June were related to the concentration of salts due to evaporation from the wetlands. High concentrations in September were related to application of fresh salt to the sand-salt storage piles.

Pinhook Bog, LaPorte County, Indiana, was adversely impacted by contamination from a salt-storage pile from the late 1960s to early 1980s (Wilcox, 1982). Chloride concentrations at control sites in 1980 and 1981 were 5-6 mg/L. This compares to a maximum single daily reading for salt-impacted locations of 1,468 mg/L chloride in 1979, 982 mg/L chloride in 1980 and 570 mg/L in 1981. The total chloride inputs to the bog over the ten-year period when salt storage occurred were estimated as follows: 2.3 million kg from the salt pile, 0.4 million kg from road salting, and 0.012 million kg from direct precipitation.

4.9 Snow Dumps and Salt-Contaminated Snow

Delisle et al. (1995) reported that of the 30,000,000 m^3 of used snow in the province of Quebec, 30% is discharged directly into lakes and rivers. However, they also notes that the Quebec Ministry of the Environment would not allow direct river dumping after 1996. Snow accumulates contaminants as it lies on the road surface and contaminant concentrations increase with snow-melt. Snow typically is high in suspended solids, chlorides, sulfate, oil, grease, calcium, sodium, and metals such as iron, lead, and chromium. Chloride concentrations in snow collected by Montreal trucks ranged from 56-10,000 mg/L and average 3,851 mg/L.

Droste and Johnston (1993) investigated four urban snow dumps in the Ottawa-Carleton region for a wide variety of parameters. Chloride concentrations ranged from 454-1,018 mg/L. Metals and suspended solids concentrations also were high and the quality of snow in the snow dump and in snow-melt waters exceeded provincial guidelines for a number of metals. However, some of this contamination could be removed by allowing suspended particulates to settle in a retention pond for a period of 2 to 6 hours, which would contaminate these sediments.

The City of Edmonton monitors melt-water quality for its snow storage sites. Data for the Kennedale site is shown in Table 4-4. Chloride concentrations ranged from 325-1,210 mg/L with highest concentrations observed in the early spring.

Table 4-4: Melt-water data (mg/L) for the Kennedale, Edmonton snow storage site for 1998. Source: Engineering Services Section, Street Engineering Branch, Edmonton Transportation.

Compound	April 30	May 12	May 26	June 9	June 25
pH	-	7.74	8.3	7.8	8.1
Arsenic	0.12	n.d.	0.02	n.d.	0.00
BOD	5.0	-4	3.5	4.0	3.7
Cadmium	n.d.	0.01	0.00	0.00	0.00
Chloride	1,210	616	325	401	338
Chromium	0.02	0.01	0.01	0.01	n.d.
Copper	0.04	0.03	0.02	0.02	0.02
Total Cyanide	0.01	0.01	0.00	0.00	<0.001
Fluorine	0.35	0.2	n.d.	0.3	0.2
Lead	0.01	0.02	0.01	n.d.	n.d.
Mercury	0.01	n.d.	n.d.	n.d.	n.d.
Oil & Grease	n.d.	8	4	6.9	<1.2
Silver	n.d.	0.05	0.00	0.00	n.d.
Total Phosphorus	0.09	0.17	0.18	0.11	0.19
Total Suspended Solids	13	8.2	28	19	9.1
Zinc	0.05	0.09	0.05	0.5	0.04

n.d. = below detection limits

Foster and Maun (1978) investigated sodium and chloride concentrations in soil, white cedar foliage, and snow at varying distances from highways 4 and 22, near London, Ontario. Chloride concentrations in snow at the pavement edge ranged from 133-4,128 mg/L; concentrations decreased to 9-79 mg/L some 8 m from the highway edge.

4.10 Highway Runoff and Storm-water Ponds

A number of studies have directly measured highway runoff, including runoff entering storm-water ponds. These studies provide information on the concentrations of road salt and other constituents directly flowing into ditches, streams, surrounding vegetation, etc.

Field and O'Shea (1992) reported a wide range in chloride concentrations in winter runoff for different areas in the Midwest. A snow pile had a chloride concentration of 1,130 mg/L, a Parkway storm drain in Des Moines, Iowa had a concentration of 2,720 mg/L, and street in Madison, Wisconsin had a concentration of 3,275 mg/L. Higher concentrations were observed in the winter runoff from a highway in Chippewa Falls, Wisconsin (10,250 mg/L) and runoff from the Expressway in Chicago, Illinois (25,100 mg/L).

Mayer et al. (1996) reported on nutrient, metal, and ionic composition of several urban storm-water ponds in the greater Toronto area. Chloride concentrations for residential areas ranged from 22-345 mg/L for Heritage Pond and 28-1,201 mg/L for Unionville Pond, while concentrations ranged from 36-617 mg/L for S. Smith Pond in an industrial area. Lowest chloride concentrations were observed in Tapscott Pond (59-216 mg/L), located in an open area. In contrast, highest metal concentrations were observed in storm-water ponds in the industrial area followed by the ponds in the residential areas.

In a later study, Mayer et al. (1998) reported on highway runoff at three study sites: the 4-lane Burlington Skyway Bridge (a high traffic area with 92,000 cars/day), the 2-lane Highway #2 at a site east of Brantford (31,100 cars/day), and the 2-lane Plains Road in Burlington (15,460 cars/day). Highest chloride concentrations (up to 10,960 mg/L) tended to be observed in runoff from the Skyway Bridge (Table 4-5), although high concentrations also were observed at the Plains Road. As expected, the highest concentrations were observed from February to April.

Table 4-5: Chloride concentrations (mg/L) in highway runoff for the Burlington area as a function of the number of lanes and traffic. Source: Mayer et al. (1998)

Date	Skyway Bridge	Highway #2	Plains Road
1997			
February 18	19,135	-	-
February 27	1,020	-	-
April 7	10,960	-	-
April 14	718	-	-
April 17	2,410	-	-
May 16	185	89	26
June 17	121	119	119
August 28	45	60	4
September 10	90	28	16
November 14	6,790	-	-
November 28	3,250	1,050	2,270
December 8	591	246	801
1998		-	
January 8	103	40	42
February 12	1,230	1,480	9,350
March 18	10,200	753	143

4.11 Replies to Requests for Information

Replies to letters sent to agencies and notices published in scientific newsletters requesting additional information on the impacts of road salt in aquatic ecosystems yielded two types of information. The first type was information in the forms of reports, papers, etc. This information is cited in the appropriate review sections in this report. The other type of information was in the form of opinion and observations. The highlights of these communications of summarized below.

Newfoundland

1) B. Moriarity with the Department of Fisheries and Oceans (letter dated December 3, 1998) noted that “in general, very little research has been carried out in the Newfoundland region on this topic” (i.e., the application of road salt and its effects of road salt on stream, lake, and wetland ecosystems). However, two studies were noted. The first was a study that assessed the use of dust suppressant chemicals in the Atlantic Region; calcium chloride was the main dust suppressant used and it was used primarily in liquid form (Kieley 1994). The second report, by Murphy (1990),

investigated the hydrogeological and geochemical characteristics of a shallow overburden aquifer that was affected by road salt applications in the Windsor area of St. John's, Newfoundland.

2) K. Robinson with the Bruce Peninsula National Park (e-mail dated July 15, 1998) noted that little was known about road salt in this Ontario park. However, studies have been conducted in Terra Nova National Park, Newfoundland. These studies indicate that road salt had effects on surrounding vegetation and salt pools acted as an attractant to moose.

Nova Scotia

1) J. Gorman with Cape Breton Highlands National Park (e-mail dated November 30, 1998) noted that road salt is used extensively between October through to May in parks and other regions in New Brunswick, Nova Scotia, and Prince Edward Island. Unfortunately "little study has been undertaken to assess or determine its impacts on the natural environment and ecosystems in this area". However, Gorman noted that heavy and continuous road salt application results in an obvious white residue or powder that, with spring rains, migrates into the soil and surrounding vegetation. Moreover, he noted that moose and deer come to the roadside to lick the road salt. Browning of vegetation also has been noted.

2) D. Rushton with Nova Scotia Transportation and Public Works (letter dated August 28, 1998) provided two reports prepared by his agency. The first was a draft report on snow and ice control operations. The second report, co-authored by the Nova Scotia Department of the Environment and the Department of Transportation and Communications (1989) discussed the environmental implications of road salting in Nova Scotia.

3) S. Parker, with the Bruce Peninsula National Park and Fathom National Marine Park (e-mail dated July 15, 1998) has noted elevated salt levels in park waters. These levels have been attributed to dust/ice control.

Ontario

1) S. Bowen with the Ecosystem Science Section, Standards Development Branch of the Ontario Ministry of the Environment (e-mail dated July 28, 1998) noted that he is working on the impact of road salts on streams draining the greater Toronto area. While background chloride concentrations range from 10-25 mg/L, increased chloride concentrations have been noted at all sites in the greater Toronto area where long-term water quality data exists.

Manitoba

1) D. Williamson with Manitoba Environment (e-mail dated September 14, 1998) noted that the Water Quality Management Section has developed guidelines for managing snow dumps to ensure that snow disposal is well back from watercourses. They have not "seen problems with high salt levels in streams, rivers, or lakes, although [they] have not done much work on wetlands in this regard". As a relatively large amount of snow accumulates through the winter, there is significant dilution of accumulated road salt with spring runoff.

2) A. Derkson with Manitoba Natural Resources, Fish Habitat Management (letter dated July 21, 1998) noted that his Branch has not conducted any surveys or research on road salts. He further noted that Winnipeg used to dump snow from street clearing into the Red River, but that this practice was discontinued some years ago because of road salt concerns. To his knowledge, road salt is not used extensively outside of Winnipeg and any which is used would be diluted during spring runoff. In the past, the Highways Department used road salt to de-ice culverts in stream crossings, apparently during spring melt. The Department was asked to use steam instead of road salt in order to prevent possible impacts to spring-spawning fish.

Alberta

1) S. Tiege with the Fisheries Management Division of Alberta Environment Protection (letter faxed July 21, 1998) noted that very little work has been done in Alberta with regard to road salt. However, Edmonton has constructed a new snow depot with a clay liner and retention ponds.

2) G. Teply with the Engineering Services Section for the City of Edmonton Transportation (letter to S. Tiege dated July 21, 1998) noted that melt-water quality is poorest at two snow storage sites: Poundmaker and Kennedale. The sites are lined with a compacted clay liner. Melt-water is collected in retention ponds and heavier solids allowed to precipitate before the water is released into the storm sewer system and ultimately into the North Saskatchewan River. It is believed that the melt-water volume is of such volume that this release of retention pond water will be sufficiently diluted before reaching the sewer outfalls and the river.

3) A. Sosiak, Water Management Division of Alberta Environment (telephone call April 4, 2000), noted that the impacts of road salt on Alberta waters have not been explicitly investigated although there are various long-term water quality data sets upstream and downstream from major urban centres in the province. For example, chloride concentrations have been measured in the Bow River upstream of Calgary (Cochrane), 45 km downstream of Calgary (Carseland), and near the river mouth (Ronalane). Average chloride concentrations in the 1970s at these sites were ca. 4.3-4.4 mg/L. Over the 1973-1997 period, chloride concentrations have increased slightly at an estimated rate of 0.057 mg/L, 0.154 mg/L and 0.171 mg/L respectively for the three sites. Increased chloride concentrations could be due to a variety of factors associated with increased urban growth.

British Columbia

1) The home page of British Columbia Ministry of Environment, Lands, and Parks was consulted for information on road salts. One document, presenting information on water quality trends (B.C. Ministry of Environment, Lands, and Parks March 1999) noted increased chloride concentrations in the Fraser River at Hope and at Marguerite and in the Thompson River at Spences Bridge: increased chloride concentrations were related to pulp mill waste abatement.

2) A second report downloaded from the web site discussed road salt and winter maintenance (B.C. Ministry of Environment, Lands, and Parks April 1999). It noted that the community of Heffley Creek suffered severe drinking water contamination from stored road salt. Moreover, the water quality of individual drinking water wells had been impaired. Total remediation costs were about two million dollars. The report also noted that two lakes, Terrace and Kitimat, between Port Alberni and Long Beach, have gone from calcium carbonate dominated to sodium chloride dominated as a result of road salt application. Snow also was pushed into these lakes.

Northwest Territories

1) C. English with Northwest Territories Transportation (letter dated October 2, 1998) provided copies of five reports that dealt with dust suppression in northern and western Canada, including the NWT.

United States

1) J. K. Crawford with the U.S. Geological Survey, Water Resources Division (letter dated August 11, 1998) provided a copy of the report authored by Harned (1988). Crawford also noted that the Susquehanna River Basin Commission was conducting a study evaluating the use of wetlands to mitigate impacts from road salts. It is interesting to note that when road salt entered First Sister Lake, Michigan through a wetland area rather than pipes, impacts of land stratification were reduced (Judd and Steggall 1982).

2) A government respondent from Washington State reported that in the spring of 1994, a hatchery used road salt to deter weed growth on its property. One thousand pounds of road salt were applied. The government respondent expressed concern that this application occurred less than 75 feet from a stream used year-round by fish.

4.12 Conclusions Regarding the Road Salt Concentrations in the Environment

Chloride concentrations vary markedly in aquatic environments. Highway runoff concentrations as high as 10,960 mg/L have been reported for the Toronto area and 25,100 mg/L for Chicago. Chloride concentrations in highway runoff decrease with decreasing highway size and traffic. Concentrations typically are highest during months of snowfall and road salt application.

1) Chloride concentrations in roadside snow range from <100 mg/L to 10,000 mg/L with concentrations typically in the 4,000 mg/L range. Snow melt from snow storage dumps have reported chloride concentration ranges of 300-1,200 mg/L. Snow and snow-melt generally are contaminated with a variety of metals, organic compounds, and particulates.

2) There is a large variation in chloride concentration in road salt-impacted streams and rivers. Highest chloride concentrations tend to be found in roadside ditches where melt-water is concentrated. The next highest concentrations (up to 4,310 mg/L) have been observed in rivers and creeks in highly populated areas with significant use of road salt. The Metropolitan Toronto area is the primary example. There is compelling evidence that many creeks and rivers in the Toronto area are significantly contaminated by chloride.

3) Some winter-deposited chloride remains in the river/stream banks and is slowly leached out through the late spring and summer. Some chloride enters the groundwater, contaminating aquifers and springs. Streams and rivers located away from densely populated urban areas have substantially lower increases in chloride concentrations as a result of road salt (e.g., as little as 5 mg/L). Nevertheless, there is evidence of widespread increases in chloride concentration in streams and rivers in many regions. Road salt has been implicated as the causal factor in many of these increases.

4) Small lakes and ponds are more strongly impacted by road salt than larger lakes, but are not as strongly impacted as creeks and rivers. For most of these small lakes, increased chloride concentrations remained below 200 mg/L. Small, moderately deep (i.e., 7-17 m) lakes located in urban areas were the most strongly impacted by road salt. Meromictic situations were created in many of these lakes, resulting in a loss of vertical mixing.

5) Larger lakes showed considerably smaller increases in chloride. Moreover, because large lakes often had a wide variety of anthropogenic activities in their watersheds, there were many potential sources of chloride (e.g., sewage, industrial discharges, etc.). These activities often were a significant or the major source of increased chloride loading to these lakes.

6) There was a surprising dearth of information on the effects of road salt application on chloride concentrations in Canadian and U.S. water bodies. In particular, no studies were located for wetland areas. Both countries have extensive water quality monitoring programs, but apparently this data has not been examined to infer trends in chloride concentrations. Nevertheless, long-term studies over broad geographic regions in the United States are showing that there has been a

gradual increase in chloride content in lake and stream waters, with greatest increases occurring in urban areas and/or near highways. Other elements, such as nitrogen and phosphorus, also have been observed to be increasing concurrently with salinity.

7) Chloride ions appear to be a strong indicator of land disturbance, some of which appears to be due to road salt application. These increases in chloride concentrations, while subtle, potentially may impact aquatic communities.

8) Road salt storage depots, if not properly constructed and maintained, can result in significant contamination of nearby waters, including lakes.

While road salt is used as a deicer, it apparently has been used for other purposes. In Manitoba, it was used in the spring to accelerate the thawing of ice-blocked culverts running under highways. In Washington state, road salt apparently was used by a hatchery as a weed killer. It is not known how widely road salt is and has been used for these other purposes.

5.0 TOXICITY OF ROAD SALTS ON AQUATIC ORGANISMS

5.1 Introduction

This section investigates the toxicity of sodium, calcium, and magnesium chlorides on freshwater organisms. The overall objective is to obtain the toxicity data to be used in the Tier 1, 2, and 3 PSL Assessments.

Unlike many of the chemicals investigated under PSL1 and now being investigated under PSL 2, magnesium, potassium, sodium, calcium, and chloride are required salts serving many essential biochemical functions (Wetzel 1983). Calcium is an essential element for algae. All plants with chlorophyll require magnesium, since it is a component of the chlorophyll molecule. Magnesium also is a micronutrient used for enzymatic transformations, especially by algae, fungi, and bacteria. Sodium and potassium are involved primarily in ion transportation and exchange in many organisms. Chloride plays a role in osmotic salinity balance and the exchange of ions.

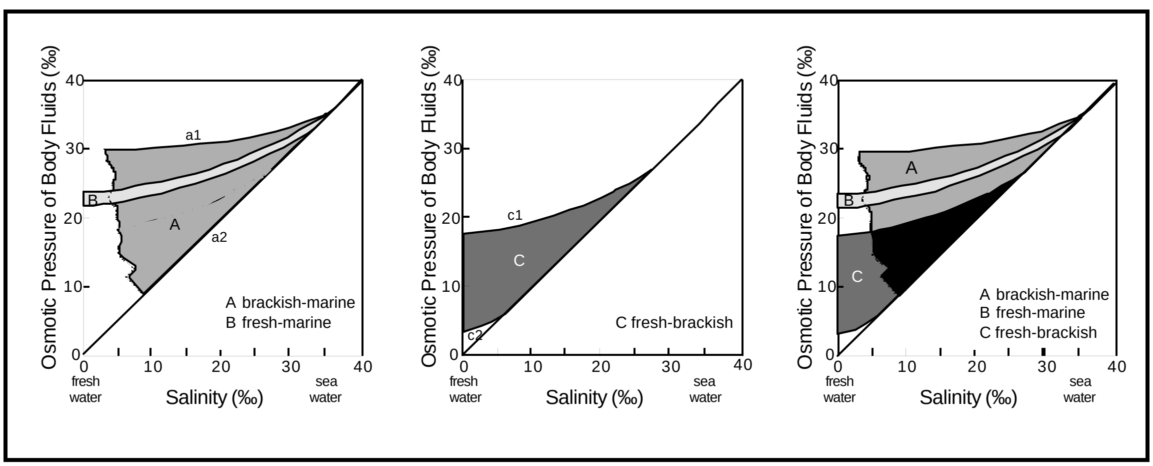
Different organisms, however, have different requirements and tolerances for these salts. Species that can tolerate a wide range of salinities are referred to as "euryhaline" (Goldman and Horne 1983). Such taxa are commonly found in estuaries, which have large gradients in salinity over short distances and, with tidal movements, time. Some taxa are found in inland waters that undergo large variations in salinity (e.g., a saline lake which, with a major flooding event, can decrease markedly in salinity). Alternately, with an extended dry period, salinity can increase markedly. Euryhaline species, found over wide gradients in salinity probably consist of several physiological races or genotypes, each adapted to a particular set of environmental conditions. In contrast, "stenohaline" species can only tolerate a narrow range of salinities. Some, such as freshwater organisms, are adapted to low-salinity environments. Others, such as marine organisms, are adapted to the high salinities associated with the world's oceans and seas.

Aquatic biota have two general strategies for dealing with high salt concentrations in their environment. The first strategy is to allow internal salt concentrations in the organism to conform to the salinities in the environment (Hammer 1986b). Usually this mechanism is found in organisms that live in fresh to mesosaline waters. In very saline waters, the internal concentrations of these "conformers" can become so great that the organism dies (Figure 5-1). Examples of conformers include the amphipod, *Gammarus duebeni*, the copepod, *Calamoecia salina*, and the freshwater caddisflies, *Limnephilus stigma* and *Anobolia nervosa* (Sutcliffe 1961b).

The second strategy involves physical mechanisms for maintaining lower ionic concentrations in the body compared to the surrounding saline environment (i.e., hypotonic regulation). To facilitate hypotonic regulation, freshwater organisms may have low surface permeability which passively controls the diffusion rate of water and ions (e.g., invertebrates with thick chitinous cuticles) (Pringle et al. 1981; Hammer 1986b). Fresh-water fish may also drink saline water and then secrete the salts in order to regain water that diffuses out of their bodies due to the concentration gradient. However, there are energetic costs associated with utilizing such a regulatory system. These energetic costs may mean that there is less energy available for other physiological processes or functions, such as reproduction.

Organisms must have a hypotonic regulating system in order to live in hypersaline ecosystems. Examples of “hyporegulators” include the chironomid, *Chironomus salinarius*, and the brine shrimp, *Artemia* spp. (Hammer 1986b). Sutcliffe (1961a) found that the caddisfly larvae, *Limnephilus affinis*, is a strong hyporegulator and this species is known to occur in both fresh- and brackish-water environments in Europe. Some fresh-water fish that have been shown to be hyporegulators (Hammer 1986b) include the three-spine stickleback (*Gasterosteus aculeatus*), the nine-spine stickleback (*Pungitius pungitius*), rainbow trout (*Oncorhynchus mykiss*), smallmouthed hardyhead (*Taeniomembras microstomus*), and the Arabian killifish (*Aphanius dispar*).

Figure 5-1: Range in osmotic pressure of body fluids (expressed as salinity) as related to the salinity of external water. Area A describes the broad range of osmotic pressure found in brackish water organisms. The pressure curve extends from a1, which is the most homoiosmotic (i.e., retains internal osmotic concentration), to a2, which is the most poikilosmotic (i.e., adjust osmotic pressure to be more or less isotonic with solution). None of these organisms have colonized freshwater systems; the left margin indicates the variability of the lower salinity limits. Area B indicates brackish water organisms that have colonized freshwater systems. These organisms are partially homoiosmotic. Area C indicates the osmotic pressure of most freshwater organisms, with extremes in osmotic pressure indicated by c1 and c2 (adapted by Wetzel, 1983).



This section presents the results of the literature review of laboratory studies investigating the toxicity of sodium, potassium, magnesium, and calcium chlorides on a wide range of aquatic organisms representing all components of aquatic ecosystems (i.e., bacteria, fungi, protozoa and flagellates, phytoplankton, macrophytes, zooplankton, benthic invertebrates, fish, amphibians and aquatic birds). The effect of a chemical on a plant or animal is directly related to (Manahan 1990):

1) The type of organism. Some organisms are more sensitive than others. Similarly, organisms may be more sensitive at certain stages in their life history or when stressed by other factors such as food limitation, low oxygen, etc.

2) The route of exposure also is important. For example, ditch vegetation may be exposed to road salt through direct spray and through snowmelt. A stream invertebrate may be exposed to a pulse of road salt draining from a rural highway or to a pulse of contaminant-laden road salt from a heavily trafficked urban bridge.

3) The amount of chemical to which the organism is exposed clearly is important. Very low concentrations may have no effect or even a beneficial effect on a organism. Toxicity will increase with increasing concentrations. After the 100% mortality concentration is reached, further increases in concentration will not affect the local population although the downstream impact will be lengthened.

4) The period of time the organism is exposed to the chemical is important. That is, a relatively high concentration of a compound is required to kill an organism exposed to that compound for a short time (e.g., one day). As exposure time is increased, a lower concentration of that compound may kill the organism. The possibility also exists for organisms to become adapted to that compound if the exposure time is long and the increase in “contaminant” concentration gradual. Each organism has its own ability to adapt to such conditions. Energetic costs may be associated with these adaptations which, in the real world, may prevent that species from competing successfully with other species in that environment.

5) The inherent ability or strength of the chemical to cause harmful effects in the organism (i.e., the chemical’s toxicity) is also important. A metal, such as arsenic, which has no known biochemical and physiological function has a greater potential to cause harmful effects to organisms than compounds, such as sodium, chloride, and potassium, which are vital to the normal biochemical and physiological functioning of most if not all organisms.

A chemical is considered to be harmful or toxic if it produces a strong negative effect when the organism is exposed to that chemical. Short-term exposures typically investigate lethal effect (i.e., death) and provide information on acute toxicity (Manahan 1990). In general, these exposure periods are in the order of minutes to a few days. Chemicals also may be toxic at lower concentrations when exposure times are longer (e.g., as little as a week to as long as several months). In such situations, the chemical is considered chronically toxic. While mortality can continue to occur over these exposure periods, the surviving population may be exhibiting other, more pronounced responses, such as reduced growth or reproductive success. Chronic, sublethal effects have the ability to greatly impact the health and survival of a population despite the fact that such effects do not result in the immediate death of an organism. For example, if a toxin affects the ability of a fish to reproduce, the population of that species may not persist in that region.

Most of the toxicity studies that were located during this literature review investigated acute lethality. These studies were based on sodium, potassium, magnesium, and chloride salts rather than the various formulations of road salt. By far the largest database was for sodium chloride toxicity. This is probably because researchers have been intrigued by the ability of organisms to survive in marine and fresh waters and, for various theoretical reasons, have investigated different aspects of salinity tolerances. This review has only touched on these studies. Fewer studies have investigated tolerances to potassium, calcium, and magnesium chlorides. These studies are included in this literature review because these chlorides are components of some road salt formulations. The largest database was obtained for fish and benthic invertebrates. However, for these other chloride salts and organisms, a sufficient number of studies were conducted to identify trends and factors influencing lethal toxicity. Lethal toxicity data is summarized in Tables 5-1 to 5-10 and listed more completely in Appendix B.

A smaller number of studies were found which located sublethal effects. These studies investigated a wide range of responses including immobilization, weakening, and increased movement. Other studies investigated changes in respiration rates, protein content, cell volume, spore production, egg production, and degradation of RNA. Data was found for bacteria, protozoa, fungi, phytoplankton, macrophytes, zooplankton, benthic invertebrates, fish, amphibians, and aquatic birds, although this data was limited for macrophytes, amphibians and aquatic birds. Laboratory data was not found on the sublethal effects of road salts on periphyton and reptiles. As well, chloride toxicity data for any one group of biota did not always include sublethal effects of all four salts (e.g., studies were not found of investigations of the sublethal effects of potassium chloride on bacteria, protozoa, or fungi). The results of the studies on sublethal effects are summarized in Appendix B.

These subsections are then followed by a discussion of three additional topics relating to the toxicity of road salts. The first is a discussion of how the interaction of various salt ions can influence the toxicity of road salts to aquatic organisms. The second discusses the toxicity of road salt itself. The third is a brief discussion of the results of recent laboratory studies investigating factors affecting intraspecies variability in toxicity. The section concludes with a summary of the highlights of these studies.

Various terms were used in the studies to define the concentration at which a lethal response occurs. These terms include threshold toxicity, minimum lethal dose, lethal concentration, and median tolerance limit. The threshold toxicity, or minimum lethal dose (MLD), is the minimum concentration required to kill one or more of the test species (McKee and Wolf 1963; Hammer 1977). The median lethal concentration (LC_{50}) or the median tolerance limit (TLm) is the concentration at which 50% of the test organisms die within a certain time period (McKee and Wolf 1963; Hammer 1977). The median effect concentration (EC_{50}) is the estimated concentration at which a sublethal or lethal toxic effect is detected in 50% of the test organisms (Environment Canada 1998). The no observable effects concentration (NOEC) is the highest concentration at which no effects were observed, the lowest observable effects concentration (LOEC) is the lowest concentration at which there is an observable effect, and the threshold effect concentration (TEC) is the geometric mean of the NOEC and LOEC.

Two units were used in reported toxicity: mg/L and moles/L or M. In order to standardize comparisons between studies, toxicity data based on molar units were converted to mg/L. These units were selected in part because water quality guidelines or criteria are expressed in mg/L. Moreover, the majority of the toxicity data was expressed in mg/L. Finally, it is more common to consider the application of road salt in units of weight (kg or tonnes) than molecules (or moles) of salt applied.

There are various limitations to this data. Many of the studies, especially the zooplankton and some of the fish studies, were conducted prior to the 1960s and provided limited information on experimental design (i.e., exposure time, water temperature, and test-water quality). This information also was lacking in review articles that cited earlier articles that could not be readily located. Other limitations of the data include the fact that specific endpoints (e.g., 50% mortality) and specific species (e.g., bluntnose minnow or *Pimephales notatus*) occasionally were not reported.

5.2 Bacteria

Only one laboratory study was found which investigated the salt tolerances of bacteria. Ito et. al (1977) determined that high concentrations of sodium chloride degrade *Escherichia coli* RNA. When the bacteria was exposed to 8,800 mg/L NaCl, 50% of the RNA in *E. coli* was degraded within two hours. However, the degradation was inhibited when 6,000 mg/L $MgSO_4$ was added in addition to the NaCl. Ito et al. hypothesized that the degradation of RNA by sodium chloride was associated with resultant losses of magnesium from the cells. *E. coli* is a predominant bacteria in human and animal intestines and incidentally in aquatic environments contaminated by such fecal matter.

5.3 Fungi

Two laboratory studies were located which investigated the effect of salts on spore production in fungi.

1) Sridhar and Barlocher (1997) found increases in sporulation of aquatic hyphomycetes when the fungi were exposed to either 659 mg/L NaCl or 554 mg/L $CaCl_2$ for 48 hours in soft water at 20 ^oC. However, sporulation decreased when the hyphomycetes were exposed to 2,215 mg/L $CaCl_2$ under the same conditions.

2) Rantamaki et al. (1992) found that the fungus, *Aphanomyces astaci*, did not produce spores when it was exposed to 1,904 mg/L $MgCl_2$ for five days in lake water at 13 ^oC.

5.4 Protozoa and Flagellates

Three studies were located which investigated the responses of flagellates to chloride salts. The first two of these studies investigated acute toxicity.

1) Gonzalez-Moreno et al. (1997) investigated the effects of sodium chloride exposure on the growth of the protozoan, *Euglena gracilis*. This organism is photosynthetic and, as a result, can also be classified as an algal species. *E. gracilis* was grown for 7 days in an acidic organotrophic medium at 27-28 ^{o}C. When grown in the light in a concentration of 5,845 mg/L NaCl, cell concentration was 84% that of the controls, whereas in the dark, cell concentration was 62% that of the controls. For the 11,690 mg/L NaCl treatment, *E. gracilis* cells attained a cell concentration of only 20% of the controls after being cultured in either the light or the dark for seven days. Other responses to elevated sodium chloride concentrations included decreases in 1) oxygen production, 2) internal concentrations of magnesium and potassium chloride, and 3) the ratio of chlorophyll a to b. Conversely, there were increases in 1) respiration, 2) cell volume, 3) internal concentrations of potassium and sodium, and 4) the total amount of both chlorophyll a and b.

2) Fuji and Hellebust (1994) investigated the effects of sodium, magnesium and potassium chloride on the growth of the euryhaline (salt-tolerant), golden-brown microflagellate alga, *Boekelovia hooglandii*. Experimental conditions involved a 12 day incubation period for the alga in culture medium at 25 ^{o}C. *B. hooglandii* grew optimally when placed in 11,690 to 23,380 mg/L NaCl, but did not grow when the sodium chloride was absent or in concentrations greater than 58,450 mg/L NaCl, nor did *B. hooglandii* grow in 13,333 mg/L $MgCl_2$ or 14,911 mg/L KCl. This data suggests that magnesium and potassium salts are toxic to *B. hooglandi* at the same concentrations at which sodium chloride provides for optimum growth.\

3) Cronkite et al. (1985) observed a 17% reduction in cell division of *Paramecium tetrourelia* when exposed to 577 mg/L NaCl. Exposure time was 5 days.

5.5 Phytoplankton

Several laboratory studies were found investigating the toxicity of road salts on non-protozoan phytoplankton growth.

1) Patrick et al. (1968) determined that the concentration of the diatom, *Nitzschi linearis*, was reduced by 50% when cells were exposed to 3,130 mg/L $CaCl_2$, 2,439 mg/L NaCl and 1,337 mg/L KCl for 120 hours. The test temperature was not given. These results suggest that *N. linearis* are most sensitive to potassium chloride, followed by sodium chloride, and finally calcium chloride. Fritz et al. (1993) investigated diatom assemblages in saline lakes of the northern Great Plains that were, in large measure, dominated by sodium, magnesium, sulfate, and carbonate ions. They estimated that *N. linearis* had a salinity optima of 2,720 mg/L and an upper and lower tolerance limit of 1,960 mg/L and 2,770 mg/L respectively in their study lakes. The difference in the tolerance

of *N. linearis* to chloride between these two field and laboratory study suggests that there are different clones of this species, with different clones having different tolerances.

2) Mohammed and Shafea (1992) investigated the effect of sodium chloride on the freshwater green alga, *Scendesmus obliquus*. The alga was exposed to the salt for seven days in a modified Beijerinck medium at a water temperature of approximately 25 ^{o}C. *S. obliquus* exposed to sodium chloride concentrations of 11,690 mg/L attained a cell concentration only 43% that of the controls. Physiological responses to high sodium chloride concentrations included decreases in oxygen production, photosynthetic pigment and dry matter. *S. obliquus* also showed an increase in respiration rates and concentrations of soluble saccharides and proteins.

3) Reynoso et al. (1982) observed a 49% reduction in growth of *Chlamydomonas reinhardtii* when exposed to a 4,965 mg/L NaCl. Exposure time was 6 days.

4) Setter et al. (1982) reported growth inhibition in *Chlorella emersonii*. Exposure concentration was 11,700 mg/L NaCl for 8-14 days.

5) Other studies reporting growth inhibition are Shitole and Joshi (1984) for *Pithophora oedogonia* and *Spirogyra setiformis*, and De Jong (1965) for *Chlorella vulgaris*. Other non-quantitative algal studies are reported in USEPA (1988) (e.g., Kalinkina 1979; Kalinkina and Strogonov 1980; Kalinkina et al. 1978 and Hosiaisluoma 1976).

5.6 Macrophytes

Wetland vegetation is sensitive to the water generated from roadside snowmelt and sodium chloride in particular. Three studies were located which investigated the sublethal toxicity of chloride salts to aquatic macrophytes.

1) *Sphagnum* species have been found to be sensitive to sodium chloride contamination of bog waters. Both *Sphagnum recurvum* (yellow-green peat moss) and *Sphagnum fimbriatum* (fringed bog-moss), from Pinhook Bog, La Porte, Indiana, experienced reduced growth in length with exposure to increasing sodium chloride concentrations (Wilcox 1984; Wilcox and Andrus 1987). *S. recurvum* grown in control bog waters increased in length by a mean of 3.22 cm versus 1.40 cm for populations exposed to 5,000 mg/L Cl^-. *S. fimbriatum* was more sensitive, attaining a mean increase in length of 2.61 cm in bog waters, but only 0.03 cm in waters with 5,000 mg/L Cl^-.

2) Stanley (1974) reported a 50% reduction in dry weight in the Eurasian millfoil, *Myriophyllum spicatum*, when exposed to sodium chloride concentrations ranging from 2,196-8,178 mg/L. Exposure time was 32 days.

3) Teeter (1965) reported reduced germination of the seeds of the pondweed, *Potemogeton pectinatus*, when exposed to 2,999 mg/L NaCl for 28 days. Reduced dry weight was observed in

a 9-week old plant exposed to the same concentrations for 35 days. 13-week old plants had reduced shoot growth and dry weight.

5.7 Zooplankton

Several laboratory studies were found which investigated the toxicity of road salts to zooplankton. Some studies were old (written prior to 1957) and were found in review articles. Such studies frequently were lacking information on exposure time, water temperature, or test-water quality (e.g., Dowden 1961, Dowden and Bennett 1965 and Biesinger and Christensen 1972). This missing information makes it difficult to directly compare the results of the various toxicity tests. Highlights of study results are as follows:

1) Kanygina and Lebedeva (1957) recorded that the cyclopoid copepod, *Cyclops serrulatus*, tolerates a maximum sodium chloride concentration of 394 mg/L NaCl at a water temperature of 20 ^{o}C; exposure time and water chemistry were not provided.

2) Arambasic et al. (1995) reported that 50% of the water flea, *Daphnia magna,* died when exposed to a 4,746$\pm$170 mg/L NaCl solution for 48 hours at 20 ^{o}C. The test water used was made from a combination of infusion water, test solution, and synthetic water. However, Birge et al. (1985), testing *Daphnia pulex* in stream water, reported a LC_{50} of 1,470 mg/L Cl^- for a 48-hour exposure. The chronic (21-day) TEC was only 372 mg/L Cl^-.

3) Fowler (1931) exposed the cladoceran, *Daphnia longispina*, to various salts using well water as the test water. Temperature was not reported, but was presumably near room temperature. *Daphnia* died following a 66-hour exposure to 2,922 mg/L NaCl. Tolerances to other chloride salts also were investigated. *D. longispina* died after less than 7 hours of exposure to 1,864 mg/L KCl, 43 hours of exposure to 3,524 mg/L $MgCl_2$, and 41 hours of exposure to 5,550 mg/L $CaCl_2$. This data suggests that *D. longispina* were most sensitive to potassium chloride and more sensitive to magnesium than calcium chloride. Differences in exposure time confound the interpretation of the relative sensitivity of *D. longispina* to sodium versus magnesium and calcium chloride. While sodium chloride causes death of *D. longispina* at a lower concentration (2,922 mg/L NaCl) compared to calcium and magnesium chloride (3,524 mg/L $MgCl_2$ and 5,550 mg/L $CaCl_2$), the exposure time was longer (66 hours for NaCl versus 41-43 hours for the other two salts).

4) Cowgill and Milazzo (1990) investigated the sensitivity of *Daphnia magna* and *Ceriodaphnia dubia* to water hardness and salinity. The 2-day LC_{50} was 7,754 mg/L NaCl for *D. magna* and 2,308 mg/L NaCl for *C. dubia*. This data suggests that *C. dubia* is particularly sensitive to salinity. Both species were less sensitive to water hardness (i.e., $CaCO_3$ was less toxic at the same concentrations as NaCl).

5) The quality of the test water has been shown to affect survivorship. Ellis (1937) reported that a sodium chloride concentration as low as 1 mg/L killed *Daphnia magna* within three hours. The

experiment was conducted in distilled water. This indicates that *Daphnia* require low concentrations of salts for their survival. Anderson (1946) noted that *Daphnia* do not survive long in distilled waters when only single salts are added as test substances. Similarly, Birge et al. (1985) reported an LC_{50} for *Daphnia pulex* of 1,470 mg/L Cl^- in stream water, but 3,100 mg/L Cl^- in reconstituted water. These tests were conducted over 48 hours.

6) Low oxygen concentrations may also lower salinity tolerance. Fairchild (1955) reported the threshold toxicity for *D. magna* was 3,170 mg/L NaCl when the test water contained 1.48 mg/L of dissolved oxygen. The low oxygen concentration is below the USEPA's (1976) water quality criterion for the protection of aquatic life of 5 mg/L dissolved oxygen. When the dissolved oxygen was increased to 6.4 mg/L, the threshold toxicity increased to 5,093 mg/L NaCl.

7) Temperature also affects the tolerance of zooplankton to salinity stress. Kanygina and Lebedeva (1957) found that *Daphnia magna* tolerate higher concentrations of sodium chloride at lower water temperatures. At 3 ^{o}C, the maximum concentration of sodium chloride tolerated by *D. magna* was 800 mg/L. At 20 ^{o}C , the maximum tolerance of *D. magna* dropped to 200 mg/L NaCl. Neither the duration of exposure nor other test conditions were reported.

8) Food levels also affects chloride toxicity. Biesinger and Christensen (1972) reported that unfed *Daphnia magna* experienced a 48-hour LC_{50} when exposed to 4,171 mg/L NaCl, but that fed *D. magna* had an LC_{50} of 4,629 mg/L NaCl. Similar results were obtained with potassium chloride (177 mg/L versus 317 mg/L KCl), magnesium chloride (549 mg/L versus 1,263 mg/L $MgCl_2$) and calcium chloride (143 mg/L versus 1,275 $CaCl_2$). *D. magna* were most sensitive to calcium followed by potassium chloride in the absence of food and most sensitive to potassium chloride and then magnesium chloride in the presence of food. Animals were substantially more tolerant to sodium chloride than the other three chloride salts.

Road salts and their additives result in the following sublethal effects in zooplankton: weakening, immobilization, failure to develop, inhibition of egg development, and suppression of feeding. With the exception of one study (Cowgill and Milazzo 1990), the other studies summarized are older (prior to 1948) and often from review articles that did not contain details on test conditions (e.g., Anderson et al. 1948; Edmister and Gray 1948). In addition, many of the test endpoints are more closely related to lethal effects (e.g., immobilization and overturn) than to sublethal effects (e.g., increased concentrations of saccharides and proteins).

1) Ramult (1925) reported weakening in the cladoceran, *Ceriodaphnia laticaudata*, at 2,992 mg/L NaCl and developmental failure for *Daphnia pulex* at 5,845 mg/L NaCl in pond water. Developmental failure and weakening also occurred for *D. pulex* at 1,854 mg/L $CaCl_2$ in pond water.

2) Anderson (1948) reported that sodium chloride concentrations ranging from 2,922-6,069 mg/L at 20-25^{o}C in Lake Erie water immobilized zooplankton.

3) Cowgill and Milazzo (1990) investigated various sensitivity responses of *Daphnia magna* and *Ceriodaphnia dubia* to water hardness and salinity. The 7-day LC_{50} was 5,777 mg/L NaCl for *D. magna* and 1,794 mg/L NaCl for *C. dubia*. The values were approximately 1.8 and 1.2 times higher respectively than the 2-day test results. Other measures were investigated. The EC_{50} for *D. magna* for dry weight was 4,310 mg/L NaCl, for total progeny was 4,282 mg/L NaCl, for mean number of broods was 5,777 mg/L NaCl and for mean brood size was 4,040 mg/L NaCl. Total progeny and mean brood size were the most sensitive measures and were 1.3-1.4 times lower than the EC_{50}. The NOEC was 1,296 mg/L NaCl for all three measures, except for the total progeny where the NOEC was 3,600 mg/L NaCl. The EC_{50} for *D. magna* total progeny was 1,794 mg/L NaCl, for mean number of broods was 1,991 mg/L NaCl and for mean brood size was 1,761 mg/L NaCl. Total progeny and mean brood size were the most sensitive measures and were similar to the EC_{50}. The NOEC was 1,296 mg/L NaCl for all three tests. Again, this data suggests *C. dubia* is particularly sensitive to salinity.

4) Biesinger and Christensen (1972) also investigated the effects of chloride salts on *Daphnia magna* reproduction. The test was conducted over 21 days and investigated the concentration at which reproduction was impaired by 16%. Concentrations were 1,730 mg/L for sodium chloride, 319 mg/L magnesium chloride and 101 mg/L for potassium chloride.

5) Anderson (1948) also reported that, in terms of sublethal effects, potassium chloride was most toxic to *D. magna*, followed by magnesium chloride, calcium chloride and finally sodium chloride. The concentrations resulting in the immobilization of 50% of the *D. magna* tested for 64 hours in Lake Erie water at 25°C were 432 mg/L KCl, 740 mg/L $MgCl_2$, 920 mg/L $CaCl_2$, and 3,680 mg/L NaCl.

6) Test concentrations of calcium chloride ranging from 920 to 3,662 mg/L $CaCl_2$, can cause weakening, inhibition of egg development, and immobilization (Naumann 1934).

7) Naumann (1934) also noted that the calcium chloride toxicity response is affected by the quality of the test water (i.e., soft versus hard waters). High concentrations of calcium chloride had a greater effect on *Daphnia magna* in soft water than in hard water. For example, no effect was noted after a 24 hours of exposure to 1,831 mg/L $CaCl_2$ in hard water, but a weakening was observed when *D. magna* were placed in soft waters at the same salinity. Potassium chloride, like calcium chloride, has a greater effect in soft water than in hard water. *D. magna* were irritated by 746 mg/L KCl in hard water, but were immobilized by the same concentration of KCl in soft water. Finally, 1,571 mg/L $MgCl_2$ was harmful to *D. magna* in both soft and hard water which is comparable to Anderson's (1948) results above. However, exposure time and water temperature were not reported for this study. Nauman's study suggests that *D. magna* are most sensitive to potassium chloride, least sensitive to calcium chloride, and of intermediate sensitivity to magnesium chloride.

8) Salinity tolerances may vary within a given species as shown by Hutchinson (1933) who investigated the tolerances of two species of *Daphnia* to magnesium chloride. The response observed was the continued production of living young. *Daphnia longispina* (clone 2) had the lowest tolerance at a

concentration of 119 mg/L $MgCl_2$ whereas *D. magna* had the highest tolerance at a concentration of 952 mg/L $MgCl_2$. Water temperatures and exposure times were not given.

5.8 Benthic Invertebrates

Several studies were found on the lethal effects of road salts on benthic invertebrates (Table 5-1and 5-2; Appendix B). Only the information from one of these studies (Kanygina and Lebedeva 1957) was of a secondary nature, being obtained from the review article by McKee and Wolf (1963). As a result, the data for exposure time, water temperature and test-water quality is more complete, making interpretation more straight forward. The results of the various studies indicate that lethal toxicity of road salts and their additives to benthic invertebrates is influenced by the type of invertebrate, type and concentration of salt, quality of test water and the temperature of test water.

1) Birge et al. (1985) reported an LC_{50} for the isopod, *Lirceus fontinalis*, at 2,950 mg/L Cl^- (4,863 NaCl) and the snail, *Physa gyrina*, at 2,540 mg/L Cl^-. The tests were conducted over 96-hours in reconstituted water.

2) Jones (1940; 1941) investigated sodium, calcium, magnesium, and potassium chloride toxicity to the planarian, Polycelis nigra. *P. nigra* was exposed to different salt solutions for 48 hours at 15-18 ^oC. It was most sensitive to potassium chloride (1,259 mg/L for 48-hour survival), followed by magnesium chloride (3,798 mg/L), calcium chloride (7,200 mg/L) and finally sodium chloride (11,109 mg/L).

3) Waller et al. (1996) investigated the effects of sodium, calcium and potassium chloride on the zebra mussel, *Dreissena polymorpha*. The zebra mussel is an exotic species that invaded the Great Lakes during the early 1990s. Experiments were conducted at a water temperature of 12 ^oC, a water hardness of 140$\pm$10 mg/L $CaCO_3$, and a 24-hour exposure time. A concentration of 2,500 mg/L KCl resulted in 100% mortality of both veligers and settlers. A concentration of 10,000 mg/L $CaCl_2$ was required to for the same results. A concentration of 10,000 mg/L NaCl resulted in 100% mortality of veligers and 98% mortality in settlers under the same environmental conditions. Thus, potassium chloride was the most toxic to the zebra mussel, followed by calcium chloride and then sodium chloride.

4) Dowden and Bennett (1965) observed a 48-hour LC_{50} of 10,254 mg/L for the mosquito larvae, *Culex* sp. Animals were reared in an artificial water media at room temperature.

5) Hamilton et al. (1975) investigated the sodium chloride and potassium chloride toxicity of the chironomid, *Cricotopus trifascia*, the caddisfly, *Hydroptila angusta*, and the oligochaete worm, *Nais variabilis*. Organisms were exposed to a range of either sodium chloride or potassium chloride concentrations in filtered lake water for a 48-hour period at 12 ^oC. A regression equation was then developed to estimate the concentrations causing 100% mortality. *N. variabilis* was the most sensitive species, experiencing a 100% mortality when exposed to 3,735 mg/L NaCl and 204

mg/L KCl, followed by *C. trifascia*, at 8,865 mg/L NaCl and 4,896 mg/L KCl and then *H. angusta*, at 10,136 mg/L NaCl and 6,317 mg/L KCl, under the same test conditions.

6) Sutcliffe (1961b) investigated the tolerances of two species of caddisflies to sodium chloride. Both *Anaobolia nervosa* and *Limnephilus stigma* experienced 75% mortality when exposed to a 9,936 mg/L NaCl for 72 hours. The test solution was a mixture of tap and sea water and the experiment was conducted at 14-17 ^{o}C. Mortality rates were 50% at a concentration of 7,014 mg/L NaCl under the same test conditions.

7) Thorton and Sauer (1972) found that the chironomid, *Chironomus attenatus*, experienced 100% mortality when cultured in 12,000 mg/L NaCl for 12 hours in dechlorinated, oxygenated water at 25 ^{o}C. Mortality rates were 50% when the same species were cultured at 9,995 mg/L NaCl under the same test conditions.

8) Tolerances are affected by the quality of the test water. Khanna et al. (1997) reported that the type of test water used influenced the tolerance of the nematode, *Caenorhabditis elegans*, to sodium chloride. Nematodes were cultured in moderately hard reconstituted water (MHRW) (96 mg $NaHCO_3$ + 60 mg $CaSO_4{\bullet}2H_2O$ + 60 mg $MgSO_4$ + 4 mg KCl per litre distilled water) and in K-medium (2.36 g KCl + 3.0 g NaCl per litre distilled water) at 20^{o}C. Up to 20,500 mg/L NaCl (24 hours) to 20,950 mg/L NaCl (96 hours) and 18,850 mg/L KCl (24 hours) to 18,900 mg/L KCl (96 hours) were tolerated in MHRW with mortalities not significantly different from controls. The higher mortality in K-medium rather than MHRW was likely associated with the relatively high potassium chloride concentrations of the former. The study also shows that *C. elegans* is more sensitive to potassium than sodium chloride.

9) Tolerances also are affected by water temperature. Two studies have identified that benthic invertebrates are more tolerant of salts when water temperature is lower.

   a. Kanygina and Lebedeva (1957) found that the maximum concentration of sodium chloride tolerated by the chironomid, *Stictochironomus* was 1,000 mg/L when the test water was 3 ^{o}C. However, the maximum tolerance dropped to 788 mg/L NaCl when the temperature was 20 ^{o}C. Kanygina and Lebedeva found similar results using an oligochaete worm.

   b. Waller et al. (1996) also found that increases in temperature from 12 to 17 ^{o}C shortened the length of time from 24 hours to 6 hours to achieve nearly 100% mortality in zebra mussel veligers and settlers exposed to water with 10,000 mg/L NaCl. Potassium chloride was most toxic to the zebra mussel, followed by calcium chloride and sodium chloride.

10) Finally, current velocity may affect salinity tolerances for stream invertebrates. Lowell et al. (1995) report that the EC_{50} (48 hours) for the mayfly, *Baetis tricaudatus*, increased from 4,704 mg/L NaCl in the absence of a current, to 5,330 mg/L NaCl at a current velocity of 6 cm/sec, and to 5,440 mg/l NaCl at a current velocity of 12 cm/sec. Increased current velocity increases the rate at

which freshly oxygenated water flows over the invertebrate, which may improve the physiological ability of the organism to manage saline stress.

Table 5-1: Responses of benthic invertebrates to various concentrations of sodium chloride.

Exposure Concentration (NaCl mg/L)	Exposure Concentration (NaCl moles/L)	Toxic Response	Species of Concern	Exposure Time	Water Temperature (°C)	Reference
1,319	0.023	survival and pupate	*Hydropsyche betteni* (caddisfly)	240 hours (10 days)	unknown	Kersey 1981
1,319	0.023	survival and pupate	*Hydropsyche bronta* (caddisfly)	240 hours (10 days)	unknown	Kersey 1981
1,319	0.023	survival and pupate	*Hydropsyche slossonae* (caddisfly)	240 hours (10 days)	unknown	Kersey 1981
3,735	0.064	100% mortality	*Nais variabilis* (oligochaete)	48 hours	12	Hamilton et al. 1975
4,121	0.071	20% mortality	*Gammarus pseudolimnaeus* (amphipod)	24 hours	unknown	Crowther and Hynes 1977
4,863	0.083	50% mortality	*Lirceus fontinalis* (isopod)	96 hours	unknown	Birge et al. 1985
5,330	0.091	50% mortality (stream velocity = 6 cm/sec)	*Baetis tricaudatus* (mayfly)	48 hours	unknown	Lowell et al. 1995
5,440	0.093	50% mortality (stream velocity = 12 cm/sec)	*Baetis tricaudatus* (mayfly)	48 hours	unknown	Lowell et al. 1995
7,014	0.120	50% mortality	*Anaobolia nervosa* (caddisfly)	72 hours (3 days)	14-17	Sutcliffe 1961b

Table 5-1: Continued.

Exposure Concentration (NaCl mg/L)	Exposure Concentration (NaCl moles/L)	Toxic Response	Species of Concern	Exposure Time	Water Temperature (°C)	Reference
7,014	0.120	50% mortality	*Limnephilus stigma* (caddisfly)	72 hours (3 days)	14-17	Sutcliffe 1961b
7,996	0.137	50% mortality (TLm)	*Chironomus attenatus* (chironomid)	48 hours	25	Thornton and Sauer 1972
8,865	0.152	100% mortality	*Cricotopus trifascia* (chironomid)	48 hours	12	Hamilton et al. 1975
9,890	0.169	80% mortality	*Hydropsyche betteni* (caddisfly)	144 hours (6 days)	unknown	Kersey 1981
9,936	0.170	75% mortality	*Anaobolia nervosa* (caddisfly)	72 hours (3 days)	14-17	Sutcliffe 1961b
9,936	0.170	75% mortality	*Limnephilus stigma* (caddisfly)	72 hours (3 days)	14-17	Sutcliffe 1961b
9,995	0.171	50% mortality (TLm)	*Chironomus attenatus* (chironomid)	12 hours	25	Thornton and Sauer 1972
10,000	0.171	veligers = 100% mortality; settlers = 70% mortality	*Dreissena polymorpha* (zebra mussel)	6 hours	17	Waller et al. 1996
10,000	0.171	veligers = 100% mortality; settlers = 98% mortality	*Dreissena polymorpha* (zebra mussel)	24 hours	12	Waller et al. 1996
10,136	0.173	100% mortality	*Hydroptila angusta* (caddisfly)	48 hours	12	Hamilton et al. 1975
10,254	0.175	50% mortality	*Culex sp.* (mosquito) larvae	48 hour	unknown	Dowden and Bennett 1965
11,109	0.190	survival for 48 hours	*Polycelis nigra* (planarian)	48 hours	15-18	Jones 1940

Table 5-1: Continued.

Exposure Concentration (NaCl mg/L)	Exposure Concentration (NaCl moles/L)	Toxic Response	Species of Concern	Exposure Time	Water Temperature (°C)	Reference
12,000	0.205	100% mortality	*Chironomus attenatus* (chironomid)	12 hours	25	Thornton and Sauer 1972
15,460	0.265	mortality not significantly different from controls	*Caenorhabditis elegans* (nematode)	24 hours	20	Khanna et al. 1997
15,500	0.265	mortality not significantly different from controls	*Caenorhabditis elegans* (nematode)	96 hours	20	Khanna et al. 1997
20,000	0.342	veligers = 100% mortality; settlers = 99% mortality	*Dreissena polymorpha* (zebra mussel)	6 hours	17	Waller et al. 1996
20,500	0.351	mortality not significantly different from controls	*Caenorhabditis elegans* (nematode)	24 hours	20	Khanna et al. 1997
20,950	0.358	mortality not significantly different from controls	*Caenorhabditis elegans* (nematode)	96 hours	20	Khanna et al. 1997

Only two quantitative studies were found investigating the sublethal effects of sodium, calcium, and potassium chlorides on benthic invertebrates. Responses investigated included movement, natality, and the ratio of the mean number of newborn clams to the number of parents in those studies where exposure times were reported.

1) Mackie (1978) found the natality of the clam, *Musculium securis*, was negatively affected by high sodium and calcium chloride concentrations. Experiments were conducted for 60-80 days. Natality was reduced with increasing concentrations of sodium chloride, with zero natality occurring at 1,000 mg/L NaCl. Similarly, natality was reduced with increasing concentrations of calcium chloride, up to 400 mg/L $CaCl_2$. However, at higher calcium chloride concentrations, natality increased, but not to the level of the control clams. The specific reason for improved natality above 400 mg/L $CaCl_2$ was not determined although Mackie suggested that calcium may have a stimulating effect on reproduction, while chloride may have an adverse effect. If this hypothesis is correct, then the adverse effects of chloride may be more pronounced at low calcium chloride concentrations, while the stimulating effects of calcium are more pronounced at high chloride concentrations.

2) Exposure to high concentrations of potassium chloride resulted in the increased movement of the water beetle, *Laccophilus maculosis*. Exposure to potassium chloride concentrations of 5,800 mg/L KCl resulted in the increased movement of 50% of test animals (Hodgson 1951). Exposure times were not given, but most likely were in the acute range (i.e., one day or less).

Table 5-2: Responses of benthic invertebrates to various concentrations of potassium chloride.

Exposure Concentration (KCl mg/L)	Exposure Concentration (KCl moles/L)	Toxic Response	Species of Concern	Exposure Time	Water Temperature (°C)	Reference
204	0.003	100% mortality	*Nais variabilis* (oligochaete)	48 hours	12	Hamilton et al. 1975
940	0.013	50% mortality	*Physa heterostropha* (fresh-water snail)	96 hours	20±2	Patrick et al. 1968
1259	0.017	Survival for 48 hours	*Polycelis nigra* (planarian)	48 hours	15-18	Jones 1940
2,500	0.034	Veligers = 100% mortality; settlers = 100% mortality	*Dreissena polymorpha* (zebra mussel)	24 hours	12	Waller et al. 1996
2,500	0.034	Veligers = 100% mortality; settlers = 96% mortality	*Dreissena polymorpha* (zebra mussel)	24 hours	17	Waller et al. 1996
4,896	0.066	100% mortality	*Cricotopus trifascia* (chironomid)	48 hours	12	Hamilton et al. 1975
6,317	0.085	100% mortality	*Hydroptila angusta* (caddisfly)	48 hours	12	Hamilton et al. 1975
10,000	0.134	Veligers = 100% mortality; settlers = 89% mortality	*Dreissena polymorpha* (zebra mussel)	3 hours	12	Waller et al. 1996
11,510	0.154	mortality not significantly different from controls	*Caenorhabditis elegans* (nematode)	24 hours	20	Khanna et al. 1997
11,510	0.154	mortality not significantly different from controls	*Caenorhabditis elegans* (nematode)	96 hours	20	Khanna et al. 1997
18,850	0.253	mortality not significantly different from controls	*Caenorhabditis elegans* (nematode)	24 hours	20	Khanna et al. 1997
18,900	0.253	mortality not significantly different from controls	*Caenorhabditis elegans* (nematode)	96 hours	20	Khanna et al. 1997

5.9 Fish

A substantial number of the references on the toxicity of road salts and their additives to fish found to date are older and often from review articles. As a result, information on experimental characteristics (i.e., exposure time, water temperature, and test-water quality) is often absent and specific endpoints (e.g., 50% mortality) or specific species (e.g., bluntnose minnow, *Pimephales notatus*), are often not reported. Due to the large quantity of data available on fish, the results of these studies have been summarized according to the type of salt and by the length of exposure. Short-term exposure has been defined here as an exposure of 24 hours or less, while long-term exposure has been classified as a time period greater than 24 hours. As with zooplankton and benthic invertebrates, the lethality of road salts is influenced by type of fish, type and concentration of salt, and quality and temperature of the test water.

5.9.1 Sodium Chloride

Short-term Exposure (24 hours)

There have been a number of studies conducted investigating mortality of fish when exposed to elevated sodium chloride concentrations for short time periods of one day or less. (Table 5-3 and Appendix B-1).

1) The lowest sodium chloride concentration at which a response was observed was at 5,496 mg/L NaCl. Exposure to this concentration resulted in 50% mortality of *Salmo gairdneri* (rainbow trout) (Kostecki and Jones 1983). The next lowest sodium chloride concentration at which a response was observed was at 7,500 mg/L NaCl. Exposure resulted in 50% mortality of the fry of three species of Indian carp (*Catla catla, Labeo rohoto,* and *Cirrhinius mrigalo*) (Gosh and Pal 1969). Test water temperature was 26 ^oC. This high mortality for a relatively low salinity may be associated more with the life stage than species tested. Most adult fish appear to be able to sustain short-term exposures to considerably higher salinities than reported by Gosh and Pal for Indian carp fry.

2) Waller et al. (1996) investigated the salinity tolerances of channel catfish (*Ictalurus punctatus*), bluegill sunfish (*Lepomis macrochirus*), smallmouth bass (*Micropterus dolomieu*), rainbow trout (*Oncorhynchus mykiss*), yellow perch (*Perca flavescens*), fathead minnows (*Pimephales promelas*), brown trout (*Salmo trutta*), lake trout (*Salvelinus namaycush*) and walleye (*Stizostedion vitreum*). Fish were exposed to 10,000 or 20,000 mg/L NaCl, for 6 or 24 hours, at water temperatures of 12 or 17 ^oC. Water hardness was held at 140±10 mg/L $CaCO_3$ for all tests. The only test conditions that all fish were tested under were 10,000 mg/L NaCl for an exposure time of 24 hours at a temperature of 12 ^oC. There was negligible mortality for the various species tested at sodium chloride concentrations of 10,000 mg/L NaCl.

3) A number of studies were found investigating salinity tolerance between 11,500 mg/L and 15,000 mg/L NaCl. Minnows, bluegill sunfish, and goldfish all began to experience significant mortality at these salinities (e.g., Ellis 1937; LeClerc 1960; Dowden and Bennett, 1965; Kszos et al. 1990).

4) Exposure times, resulting in mortality, were shorter for experiments conducted at salinities of 20,000 mg/L NaCl and greater. Channel catfish and fathead minnows appeared to be more sensitive than bluegill sunfish and rainbow trout (e.g., Wiebe et al. 1934; Phillips 1944).

5) Very high salinities (35,100-50,000 mg/L NaCl) resulted in death in a matter of minutes (Powers 1917; Phillips 1944).

Table 5-3: Short-term responses (24 hours or less) of fish to various concentrations of sodium chloride.

Exposure Concentration (NaCl mg/L)	Exposure Concentration (NaCl moles/L)	Toxic Response	Species of Concern	Exposure Time	Water Temperature (°C)	Reference
5,496	0.094	50% mortality	*Salmo gairdneri* (rainbow trout)	24 hours	unknown	Kostecki and Jones 1983
7,500	0.128	50% mortality	*Catla catla, Labeo rohoto, Cirrhinius mrigalo* (three species of Indian carp fry)	24 hours	26	Gosh and Pal 1969 (in Hammer 1977)
10,000	0.171	0% mortality	*Ictalurus punctatus* (channel catfish)	24 hours	12 and 17 (two different experiments)	Waller et al. 1996
10,000	0.171	0% mortality	*Lepomis macrochirus* (bluegill sunfish)	24 hours	12 and 17 (two different experiments)	Waller et al. 1996
10,000	0.171	3.3% mortality	*Micropterus dolomieu* (smallmouth bass)	24 hours	12	Waller et al. 1996
10,000	0.171	killed or immobilized	Minnows	6 hours	unknown	LeClerc 1960; LeClerc and Devlaminck 1950; 1955 (in McKee and Wolf 1963)

Table 5-3: Continued

Exposure Concentration (NaCl mg/L)	Exposure Concentration (NaCl moles/L)	Toxic Response	Species of Concern	Exposure Time	Water Temperature (°C)	Reference
10,000	0.171	0% mortality	*Oncorhynchus mykiss* (rainbow trout)	24 hours	12 and 17 (two different experiments)	Waller et al. 1996
10,000	0.171	0% mortality	*Perca flavescens* (yellow perch)	24 hours	12 and 17 (two different experiments)	Waller et al. 1996
10,000	0.171	0% mortality	*Pimephales promelas* (fathead minnow)	24 hours	12 and 17 (two different experiments)	Waller et al. 1996
10,000	0.171	0% mortality	*Salmo trutta* (brown trout)	24 hours	12	Waller et al. 1996
10,000	0.171	0% mortality	*Stizostedion vitreum* (walleye)	24 hours	12 and 17	Waller et al. 1996
11,500	0.197	killed or immobilized	minnows	6 hours	unknown	LeClerc 1960; LeClerc and Devlaminck 1950; 1955 (in McKee and Wolf 1963)
11,765	0.201	killed or immobilized	*Carassius auratus* (goldfish)	17 hours	unknown	Ellis 1937 (in McKee and Wolf 1963)

Table 5-3: Continued

Exposure Concentration (NaCl mg/L)	Exposure Concentration (NaCl moles/L)	Toxic Response	Species of Concern	Exposure Time	Water Temperature (°C)	Reference
13,480	0.231	50% mortality	*Carassius auratus* (goldfish)	24 hours	unknown	Dowden and Bennett 1965
14,100	0.241	50% mortality (TLm)	*Lepomis macrochirus* (bluegill sunfish)	24 hours	unknown	Abegg 1949; 1950 (in Doudoroff and Katz 1953)
14,194	0.243	50% mortality	*Lepomis macrochirus* (bluegill sunfish)	24 hour	unknown	Dowden and Bennett 1965
14,612–29,224	0.250-0.500	death	*Orizias latipes* (small freshwater cyprinodont)	24 hours	unknown	Iwao 1936 (in Doudoroff and Katz 1953)
15,000	0.257	average survival time	*Notemigonus crysoleucas* (golden shiners)	4.73 hours	22-22.5	Wiebe et al. 1934
20,000	0.342	100% mortality	*Ictalurus punctatus* (channel catfish)	6 hours	17	Waller et al. 1996
20,000	0.342	47% mortality	*Lepomis macrochirus* (bluegill sunfish)	6 hours	17	Waller et al. 1996
20,000	0.342	average survival time	*Notemigonus crysoleucas* (golden shiners)	1.33 hours	22-22.5	Wiebe et al. 1934
20,000	0.342	40% mortality	*Oncorhynchus mykiss* (rainbow trout)	6 hours	17	Waller et al. 1996

Table 5-3: Continued

Exposure Concentration (NaCl mg/L)	Exposure Concentration (NaCl moles/L)	Toxic Response	Species of Concern	Exposure Time	Water Temperature (°C)	Reference
20,000	0.342	100% mortality	*Pimephales promelas* (fathead minnows)	6 hours	17	Waller et al. 1996
30,000	0.513	survival and recovery	*Salvelinus fontinalis* (brook trout)	0.5 to 1 hour	unknown	Phillips 1944
35,100	0.601	death	*Carassius auratus* (goldfish)	0.47 - 0.63 hours	~21	Powers 1917 (in Hammer 1977; Doudoroff & Katz 1953)
50,000	0.855	50% mortality	*Salvelinus fontinalis* (brook trout)	0.25 hour	unknown	Phillips 1944

Long-term Exposure (2 days to one month)

Various tests of sodium chloride toxicity to fish have been conducted using an exposure time of greater than 24 hours (Table 5-4, Appendix B-1). The range of concentrations causing death for these longer exposures is 2,500-17,500 mg/L NaCl.

1) No mortality was observed for rainbow and brown trout fingerlings exposed to salinities of 800-1,000 mg/L NaCl for approximately eight days (Camargo and Tarazona 1991).

2) As salinity increased above 1,000 mg/L, mortality began to be observed. Pickering et al. (1996) reported that the no observed effects concentration (NOEC) for 1-7 day old fathead minnow was 4,000 mg/L NaCl and the lowest observed effects concentration (LOEC) was 8,000 mg/L. Other studies conducted in this range were less quantitative.

3) Similarly, Birge et al. (1985) reported a 96-hour LC_{50} for fathead minnow of 10,830 mg/L NaCl (6,570 mg/L Cl^-) and 9,628 mg/L NaCl (5,840 mg/l Cl^-) for bluegill sunfish. Experiments were conducted in reconstituted water which provided a two times greater estimate of LC_{50} than experiments conducted when *Daphnia pulex* was the test organism.

4) Birge et al. (1985) also reported that sodium chloride concentrations as low as 1,650 mg/L NaCl had an adverse impact of the 33-day survival of fathead minnow eggs (mean survival ca. 20%) with no survival at 2,308 mg/L NaCl. Moreover, fry were statistically smaller at test concentrations of 1,210 mg/L NaCl (15.91 mm) and 1,743 mg/L NaCl (15.62 mm) than in the control and concentrations ranging from 425-880 mg/L NaCl (16.92-17.73 mm). The chronic TEC for fathead minnow was 492 mg/L NaCl.

5) Adelman et al. (1976) reported a 96-hour LC_{50} of 7,650 mg/L NaCl for fathead minnows and 7,341 mg/L NaCl for goldfish. Threshold LC_{50} of 7,650 mg/L NaCl and 7,322 mg/L NaCl were reached in six days.

6) Hinton and Eversole (1978) determined that the glass eel (larval) stage of the American eel (*Anguilla rostrata*) has a 96-hour LC_{50} of 17,880 mg/L NaCl. In contrast, the black eel stage had a 96-hour LC_{50} of 21,450 mg/L NaCl. Adult eels live in fresh and salt waters, being found in the Great Lakes, offshore of the Maritimes, and as far north as Greenland and as far south as the north coast of South America; eels spawn in freshwater (Scott and Crossman 1973).

7) In order to improve estimates of salinity tolerances of fish between ambient sodium chloride concentrations and salinities up to 8,000 mg/L NaCl, Beak International (1999) was contracted to conduct a series of tests using fathead minnow and rainbow trout. Experimental concentrations were control, 250, 500, 1,000, 2,000, 4,000, and 8,000 mg/L NaCl. Experimental results were as follows:

   a) Fathead minnow larvae (<24-hours old) exposed to elevated salinities for 7 days experienced no mortality at salinities up to 1,000 mg/L NaCl. Mean mortality was 10% at 2,000 mg/L NaCl, 28% at 4,000 mg/L NaCl, and 75% at 8,000 mg/L NaCl.

   b) Surviving fish had impaired growth at 4,000 and 8,000 mg/L NaCl with mean body weight approximately 50% of the controls. Swimming behaviour was also impaired at salinities of 4,000 and 8,000 mg/L NaCl.

   c) The estimated 7-day NOEC was 2,000 mg/L NaCl, the LOEC was 4,000 mg/L NaCl and the TEC concentration was 2,830 mg/L NaCl. The LC_{50} was 5,490 mg/L NaCl. In contrast, Birge et al. (1985) reported a 33-day TEC of 492 mg/L NaCl for fathead minnows reared in reconstituted water.

   d) Potassium chloride was used as a reference toxicant and had a 7-day LC_{50} of 861 mg/L KCl. This compares favorably to Pickering et al.'s (1996) NOEC of 500 mg/L for KCl.

   e) A 7-day embryo larval and teratogenicity test was conducted with fathead minnow embryos <36 hours old. There was no significant mortality at salinities up to 1,000 mg/L NaCl, 90% mortality at 2,000 mg/L NaCl, and 100% at both 4,000 and 8,000 mg/L NaCl. A number of deformed larvae were observed at 2,000 mg/L NaCl. The final mortality, which included the combined number of dead embryos and dead and deformed larvae was estimated at 97.5% at 2,000 mg/L NaCl and 100% at 4,000 and 8,000 mg/L NaCl. The calculated NOEC was 1,000 mg/L NaCl, the LOEC was 2,000 mg/L NaCl and the TEC was 1,410 mg/L NaCl. Fathead minnow embryos appear to be more sensitive to elevated sodium chloride concentrations than larvae.

f) Similar 7-day tests were conducted with fertilized rainbow trout eggs. There was no difference between mean mortality of the control and eggs exposed up to 2,000 mg/L NaCl (range of mortality 2.5-4.1%). Mean mortality at 4,000 mg/L NaCl was 86.3% and 100% at 8,000 mg/L NaCl. The estimated EC_{25} was 1,630 mg/L NaCl and the EC_{50} was 2,400 mg/L NaCl.

g) Finally, a 27-day embryo-alvin test was conducted with rainbow trout. The mean percentage of nonviable embryos ranged from 10.8-18.5% at salinities ranging from the control to 1,000 mg/L NaCl. There was no relationship between mortality and salinity. Mean mortality (uncorrected for controls) was 31% at 2,000 mg/L NaCl, 90.8% at 4,000 mg/L NaCl and 100% at 8,000 mg/L NaCl. The estimated EC_{25} was 18,630 mg/L NaCl and the EC_{50} was 2,630 mg/L NaCl.

Overall, these tests suggest that mortality of early life history stages of fish increase sharply between 2,000 and 4,000 mg/L NaCl when exposed for periods of approximately seven days.

8) Other researchers have investigated salinity tolerances at sodium chloride concentrations of 10,000 mg/L NaCl and higher. Bluegill sunfish had 50% mortality at salinities of ca. 13,000 mg/L NaCl when exposed for 96 hours while mosquito fish had ca. 50% mortality when exposed to 17,500 mg/L NaCl for the same time period (Patrick et al. 1968; Trama 1954; Wallen et al. 1957). Perch also appear to be fairly tolerant of sodium chloride, surviving 14 days in 14,000 mg/L NaCl (Black 1950). When the sodium chloride concentration was gradually increased from 9,100 to 17,500 mg/L, yellow perch survived (Young 1923).

9) As with invertebrates, chloride tolerance varies with the species tested and the type of salt. Edmister and Gray (1948) investigated the tolerances of lake whitefish (*Coregonus clupeaformis*) and walleye (*Stizostedion vitreum*) fry to potassium, sodium, and calcium chlorides. Walleye fry were the most sensitive, becoming immobilized at 751 mg/L KCl, 3,859 mg/L NaCl, and 12,060 mg/L $CaCl_2$. Lake whitefish fry were substantially more tolerant becoming immobilized at 10,368 mg/L KCl, 16,500 mg/L NaCl, and 22,080 mg/L $CaCl_2$. Furthermore, these tests indicate the fry were most sensitive to potassium chloride, followed by sodium chloride and then calcium chloride. Exposure time was not given. An additional study by Threader and Houston (1983) was located, but exposure times were unknown.

Table 5-4: Long-term responses (greater than 24 hours) of fish to various concentrations of sodium chloride.

Exposure Concentration (NaCl mg/L)	Exposure Concentration (NaCl moles/L)	Toxic Response	Species of Concern	Exposure Time	Water Temperature (^oC)	Reference
734	0.013	smaller size	*Pimephales promelas* (fathead minnow) fry	33 days	unknown	Birge et al. 1985
800	0.0137	survival (no effect)	*Oncorhynchus mykiss* (rainbow trout) fingerlings	196 hours (~8 days)	15-16	Camargo and Tarazona 1991
1,000	0.017	survival (no effect)	*Pimephales promelas* (fathead minnow) embryo <24 hours	168 hours (7 days)	23	Beak 1999
1,000	0.017	survival (no effect)	*Pimephales promelas* (fathead minnow) embryo <36 hours	168 hours (7 days)	23	Beak 1999
1,000	0.017	survival (no effect)	*Salmo trutta* (brown trout) fingerlings	196 hours (~8 days)	15-16	Camargo and Tarazona 1991
1,057	0.018	smaller size	*Pimephales promelas* (fathead minnow) fry	33 days	unknown	Birge et al. 1985
1,054 - 1,060	0.018	80% mortality	*Pimephales promelas* (fathead minnow) eggs	33 days	unknown	Birge et al. 1985

Table 5-4: Continued.

Exposure Concentration (NaCl mg/L)	Exposure Concentration (NaCl moles/L)	Toxic Response	Species of Concern	Exposure Time	Water Temperature (°C)	Reference
2,000	0.034	31% mortality	*Oncorhynchus mykiss* (rainbow trout) eggs (embryo-alvin test)	168 hours (7 days)	23	Beak 1999
2,000	0.034	10% mortality	*Pimephales promelas* (fathead minnow) embryo <24 hours	168 hours (7 days)	23	Beak 1999
2,000	0.034	90% mortality	*Pimephales promelas* (fathead minnow) embryo <36 hours	168 hours (7 days)	23	Beak 1999
2,500	0.043	threshold toxicity	*Notropis atherinoides* (lake emerald shiner)	120 hours	18	Van Horn et al. 1949
2,500	0.043	death	*Notropis blennius* (river shiner)	216-576 hours	room temperature	Garrey 1916 (in Hammer 1977; Doudoroff and Katz 1953)
2,500	0.043	threshold toxicity	*Notropis spilopterus* (spotfin shiner)	120 hours	18	Van Horn et al. 1949
4,000	0.068	86.3% mortality	*Oncorhynchus mykiss* (rainbow trout) eggs	168 hours (7 days)	23	Beak 1999

Table 5-4: Continued.

Exposure Concentration (NaCl mg/L)	Exposure Concentration (NaCl moles/L)	Toxic Response	Species of Concern	Exposure Time	Water Temperature (°C)	Reference
4,000	0.068	90.8% mortality	*Oncorhynchus mykiss* (rainbow trout) eggs (embryo-alvin test)	168 hours (7 days)	23	Beak 1999
4,000	0.068	28% mortality surviving fish had impaired growth and swimming behavior	*Pimephales promelas* (fathead minnow) embryo <24 hours	168 hours (7 days)	23	Beak 1999
4,000	0.068	100% mortality	*Pimephales promelas* (fathead minnow) embryo <36 hours	168 hours (7 days)	23	Beak 1999
4,000	0.068	NOEC	*Pimephales promelas* (fathead minnow) embryo <24 hours	168 hours (7 days)	23	Beak 1999
4,000	0.068	LOEC	*Pimephales promelas* (fathead minnow) embryo	168 hours (7 days)	25 ± 1	Pickering et al. 1996
5,000	0.086	Survival	*Carassius auratus* (goldfish)	240 hours	unknown	Ellis 1937 (in Hammer 1977)

Table 5-4: Continued.

Exposure Concentration (NaCl mg/L)	**Exposure Concentration (NaCl moles/L)**	**Toxic Response**	**Species of Concern**	**Exposure Time**	**Water Temperature (°C)**	**Reference**
5,000	0.086	survival (no effect)	*Carassius auratus*. (goldfish)	600 hours (25 days)	unknown	Ellis 1937 (in Hanes et al. 1970 and Anderson 1948)
5,000	0.086	0% mortality	*Micropterus salmoides* (largemouth black bass)	200-250 hours	22 - 22.5	Wiebe et al. 1934
5,000	0.086	average survival time	*Notemigonus crysoleucas* (golden shiners)	148 hours	22 - 22.5	Wiebe et al. 1934
5,840	0.100	50% mortality	*Lepomis macrochirus* (bluegill sunfish)	96 hours	unknown	Birge et al. 1985
6,000	0.103	50% mortality	*Catla catla, Labeo rohoto, Cirrhinius mrigalo* (three species of Indian carp fry)	48 hours	unknown	Gosh & Pal 1969 (in Hammer 1977)
7,341	0.126	50% mortality	*Carassius auratus* (goldfish)	96 hours	25	Adelman et al. 1976
7,650	0.131	50% mortality (LC_{50})	*Pimephales promelas* (fathead minnows)	96 hours	25	Adelman et al. 1976
8,000	0.137	100% mortality	*Oncorhynchus mykiss* (rainbow trout) eggs	168 hours (7 days)	23	Beak 1999

Table 5-4: Continued.

Exposure Concentration (NaCl mg/L)	**Exposure Concentration (NaCl moles/L)**	**Toxic Response**	**Species of Concern**	**Exposure Time**	**Water Temperature (°C)**	**Reference**
8,000	0.137	100% mortality	*Oncorhynchus mykiss* (rainbow trout) eggs (embryo-alvin test)	168 hours (7 days)	23	Beak 1999
Gradual increase from 9,100 to 17,500	gradual increase from 0.156 to 0.299	survival	*Perca flavescens* (yellow perch)	720 hours (1 month)	unknown	Young 1923 (in Hanes et al. 1970)
8,000	0.137	75% mortality surviving fish had impaired growth and swimming behavior	*Pimephales promelas* (fathead minnow) embryo <24 hours	168 hours (7 days)	23	Beak 1999
8,000	0.137	100% mortality	*Pimephales promelas* (fathead minnow) embryo <36 hours	168 hours (7 days)	23	Beak 1999
8,000	0.137	LOEC	*Pimephales promelas* (fathead minnow) 1 to 7 days old	168 hours (7 days)	25±1	Pickering et al. 1996
10,000	0.171	death	*Carassius auratus.* (goldfish)	240 hours or less (10 days or less)	unknown	Ellis 1937 (in Hanes et al. 1970 & Anderson 1948)
10,000	0.171	100% mortality	*Micropterus salmoides* (largemouth black bass)	142-148 hours	22-22.5	Wiebe et al. 1934

Table 5-4: Continued.

Exposure Concentration (NaCl mg/L)	Exposure Concentration (NaCl moles/L)	Toxic Response	Species of Concern	Exposure Time	Water Temperature (^{o}C)	Reference
10,000	0.171	average survival time	*Notemigonus crysoleucas* (golden shiners)	97 hours	22-22.5	Wiebe et al. 1934
11,690	0.200	survival	*Anguilla japonica* (young eel)	50 hours	20-22	Oshima 1931 (in Doudoroff & Katz 1953)
11,700	0.200	death	*Carassius auratus* (goldfish)	17 - 154 hours	~21	Powers 1917 (in Hammer 1977; Doudoroff and Katz 1953)
12,200	0.209	50% mortality (LC_{50})	*Lepomis macrochirus* (bluegill sunfish) young of the year	288 hours (12 days)	18.8-20.1	Kszos et al. 1990
12,946	0.221	50% mortality (LC_{50})	*Lepomis macrochirus* (bluegill sunfish)	96 hours	18 ± 2	Patrick et al. 1968
12,964	0.222	50% mortality (TLm)	*Lepomis macrochirus* (bluegill sunfish)	96 hours	unknown	Trama 1954 (in McKee and Wolf 1963)
14,000	0.240	0% mortality	bass	336 hours (14 days)	unknown	Black 1950 (in Hanes et al. 1970)
14,000	0.240	0% mortality	perch	336 hours (14 days)	unknown	Black 1950 (in Hanes et al. 1970)
17,500	0.299	50% mortality (TLm)	*Gambusia affinis* (mosquito-fish)	96 hours	unknown	Wallen et al. 1957 (in Hammer 1977; McKee and Wolf, 1963)

5.9.2 Calcium Chloride

Short-term Exposure (24 hours or less)

A few studies have investigated the short-term toxicity (24 hours or less) of calcium chloride (Table 5-5 and Appendix B-2). Waller et al. (1996) conducted the most detailed studies. Unfortunately, the lowest test exposure was 7,752 mg/L $CaCl_2$. Highlights are as follows:

1) Three older studies, Ellis (1937) and Abegg (1949; 1950) suggest that harmful effects can be observed on goldfish and bluegill sunfish at concentrations of 7,550-8,400 mg/L $CaCl_2$. The first study was conducted in distilled water where as the latter two were conducted in synthetic river water. An additional study by Cairns and Scheier (1959) was located, but exposure times were unknown.

2) Dowden and Bennett (1965) observed 50% mortality in bluegill (*Lepomis macrochirus*) when exposed to a concentration of 8,363 mg/L $CaCl_2$.

3) Waller et al. (1996) conducted an extensive suite of tests involving calcium chloride and the same fish species as were tested with sodium chloride (i.e., channel catfish, bluegill sunfish, smallmouth bass, rainbow trout, yellow perch, fathead minnows, brown trout, lake trout and walleye). Test conditions involved 10,000 or 20,000 mg/L $CaCl_2$, exposure times of 3, 12 or 24 hours, and water temperatures of 12 or 17 ^oC. Water hardness was again held at 140 ± 10 mg/L $CaCO_3$ for all tests.

    a) Channel catfish and fathead minnows experienced 100% mortality when exposed for 24-hours at 12^oC to 10,000 mg/L $CaCl_2$ water. Yellow perch (80% mortality), brown trout (20% mortality), rainbow trout (16% mortality), bluegill sunfish (3.3% mortality), walleye (3.3% mortality) and smallmouth bass (0% mortality) were less sensitive to these conditions. These same species of fish where less sensitive to sodium chloride under the same experimental conditions, experiencing no mortality. This is in contrast to Edmister and Gray's (1948) study that found walleye and lake whitefish fry to be more sensitive to sodium chloride than calcium chloride. Whitefish fry were immobilized at 12,060 mg/L $CaCl_2$ versus 16,560 mg/L NaCl while walleye fry were immobilized at 22,080 mg/L $CaCl_2$ versus 3,859 mg/L NaCl. However, Edmister and Gray's study was an older, less comprehensive study and unknown factors may have confounded study results.

    b) Fathead minnow experienced a mean mortality of 47%, channel catfish a mean mortality of 37%, and yellow perch a mean mortality of 10% when exposed for 12-hours at 12^oC to 10,000 mg/L $CaCl_2$. Yellow perch apparently are more sensitive that fathead minnow to calcium chloride.

    c) The results of tests conducted by Waller et al. (1996) also illustrate how the sensitivity of fish to calcium chloride increases with increasing temperatures. Channel catfish experienced a mortality of 37% during a 12-hour exposure to 10,000 mg/L $CaCl_2$ at 12°C. Mortality increased to 63% when the same experiment was conducted at 17 °C.

Table 5-5: Short-term responses (24 hours or less) of fish to various concentrations of calcium chloride.

Exposure Concentration ($CaCl_2$ mg/L)	Exposure Concentration ($CaCl_2$ moles/L)	Toxic Response	Species of Concern	Exposure Time	Water Temperature (°C)	Reference
7,752	0.070	killed or injured	*Carassius auratus* (goldfish)	22-27 hours	unknown	Ellis 1937 (in McKee and Wolf 1963)
8,363	0.143	50% mortality	*Lepomis macrochirus* (bluegill sunfish)	24 hour	unknown	Dowden and Bennett 1965
8,400	0.076	50% mortality (TLm)	*Lepomis macrochirus* (bluegill sunfish)	24 hours	unknown	Abegg 1949, 1950 (in Doudoroff and Katz 1953)
10,000	0.090	0% mortality	*Lepomis macrochirus* (bluegill sunfish)	12 hours	12 and 17 (two different experiments)	Waller et al. 1996
10,000	0.090	0% mortality	*Micropterus dolomieu* (smallmouth bass)	12 hours	12	Waller et al. 1996
10,000	0.090	0% mortality	*Stizostedion vitreum* (walleye)	12 hours	12 and 17 (two different experiments)	Waller et al. 1996
10,000	0.090	10% mortality	*Oncorhynchus mykiss* (rainbow trout)	12 hours	12	Waller et al. 1996
10,000	0.090	10% mortality	*Perca flavescens* (yellow perch)	12 hours	12	Waller et al. 1996

Table 5-5: Continued.

Exposure Concentration ($CaCl_2$ mg/L)	Exposure Concentration ($CaCl_2$ moles/L)	Toxic Response	Species of Concern	Exposure Time	Water Temperature (°C)	Reference
10,000	0.090	37% mortality	*Ictalurus punctatus* (channel catfish)	12 hours	12	Waller et al. 1996
10,000	0.090	47% mortality	*Pimephales promelas* (fathead minnow)	12 hours	12	Waller et al. 1996
10,000	0.090	0% mortality	*Micropterus dolomieu* (smallmouth bass)	24 hours	12	Waller et al. 1996
10,000	0.090	0% mortality	*Salvelinus namaycush* (lake trout)	24 hours	12	Waller et al. 1996
10,000	0.090	3.3% mortality	*Lepomis macrochirus* (bluegill sunfish)	24 hours	12	Waller et al. 1996
10,000	0.090	3.3% mortality	*Stizostedion vitreum* (walleye)	24 hours	12	Waller et al. 1996
10,000	0.090	16% mortality	*Oncorhynchus mykiss* (rainbow trout)	24 hours	12	Waller et al. 1996
10,000	0.090	20% mortality	*Salmo trutta* (brown trout)	24 hours	12	Waller et al. 1996
10,000	0.090	80% mortality	*Perca flavescens* (yellow perch)	24 hours	12	Waller et al. 1996
10,000	0.090	100% mortality	*Ictalurus punctatus* (channel catfish)	24 hours	12	Waller et al. 1996

Table 5-5: Continued.

Exposure Concentration ($CaCl_2$ mg/L)	Exposure Concentration ($CaCl_2$ moles/L)	Toxic Response	Species of Concern	Exposure Time	Water Temperature (°C)	Reference
10,000	0.090	100% mortality	*Pimephales promelas* (fathead minnow)	24 hours	12 and 17 (two different experiments)	Waller et al. 1996
10,000	0.090	40% mortality	*Oncorhynchus mykiss* (rainbow trout)	12 hours	17	Waller et al. 1996
10,000	0.090	63% mortality	*Ictalurus punctatus* (channel catfish)	12 hours	17	Waller et al. 1996
10,000	0.090	83% mortality	*Perca flavescens* (yellow perch)	12 hours	17	Waller et al. 1996
10,000	0.090	0% mortality	*Lepomis macrochirus* (bluegill sunfish)	24 hours	17	Waller et al. 1996
10,000	0.090	0% mortality	*Stizostedion vitreum* (walleye)	24 hours	17	Waller et al. 1996
10,000	0.090	49% mortality	*Oncorhynchus mykiss* (rainbow trout)	24 hours	17	Waller et al. 1996
10,000	0.090	death	*Tinca vulgaris* (tench)	3 hours	20	Wiebe et al. 1934
10,000	0.090	average survival time	*Notemigonus crysoleucas* (golden shiners)	27.6 hours	22-22.5	Wiebe et al. 1934
13,874	0.125	death	*Oriziass latipes* (small freshwater cyprinodont)	24 hours	unknown	Iwao 1936 (in Doudoroff and Katz 1953)

Table 5-5: Continued.

Exposure Concentration ($CaCl_2$ mg/L)	Exposure Concentration ($CaCl_2$ moles/L)	Toxic Response	Species of Concern	Exposure Time	Water Temperature (°C)	Reference
15,000	0.135	average survival time	*Lepomis macrochirus* (bluegill sunfish)	17.7 hours	22-22.5	Wiebe et al. 1934
15,000	0.135	average survival time	*Notemigonus crysoleucas* (golden shiners)	17 hours	22-22.5	Wiebe et al. 1934
20,000	0.180	17% mortality	*Stizostedion vitreum* (walleye)	3 hours	12	Waller et al. 1996
20,000	0.180	20% mortality	*Oncorhynchus mykiss* (rainbow trout)	3 hours	12	Waller et al. 1996
20,000	0.180	83% mortality	*Ictalurus punctatus* (channel catfish)	3 hours	12	Waller et al. 1996
20,000	0.180	97% mortality	*Lepomis macrochirus* (bluegill sunfish)	3 hours	12	Waller et al. 1996
20,000	0.180	100% mortality	*Pimephales promelas* (fathead minnow)	3 hours	12 and 17 two different experiments)	Waller et al. 1996
20,000	0.180	63% mortality	*Oncorhynchus mykiss* (rainbow trout)	3 hours	17	Waller et al. 1996
20,000	0.180	100% mortality	*Stizostedion vitreum* (walleye)	3 hours	17	Waller et al. 1996
20,000	0.180	average survival time	*Notemigonus crysoleucas* (golden shiners)	6.4 hours	22-22.5	Wiebe et al. 1934
20,000	0.180	average survival time	*Lepomis macrochirus* (bluegill sunfish)	19.5 hours	22-22.5	Wiebe et al. 1934

Long-term Exposure (50 hours to 7 weeks)

Some studies were located which investigated the longer-term responses (50 hours to 7 weeks) of fish to elevated calcium chloride concentrations (Table 5-6 and Appendix B-2). These tests were conducted at concentrations ranging from 277 - 13,400 mg/L $CaCl_2$. Highlights are as follows.

1) The river shiner (*Notropis blennis*) died when exposed to calcium chloride concentrations as low as 277 mg/L for 7 weeks (Garrey 1916). This was an older study and details on the experimental design were limited. The tests were conducted in distilled water, which is a poor rearing medium because it lacks essential salts.

2) Cairns and Scheir (1958) report a 50% mortality of bluegill sunfish exposed to 9,500 mg/L $CaCl_2$ for 96 hours, an estimate that is not much different from Patrick et al.'s (1968) observation of 50% mortality at 10,650 mg/L. In contrast, 50% mortality of bluegill sunfish when exposed to elevated concentrations of sodium chloride for 96 hours occurs at ca. 12,964 mg/L NaCl (Table 5-4). This suggests that bluegill sunfish may be less tolerant of calcium chloride than sodium chloride.

3) As expected, the tolerance to calcium chloride decreases with exposure time. Wiebe et al. (1934) reported that golden shiners (*Notemigonus crysoleucas*) survived for 143.5 hours in a mixture of distilled and tap water containing 5,000 mg/L $CaCl_2$. They only survived 28 hours when the test water contained 10,000 mg/L $CaCl_2$ and for 6.4 hours when the test water contained 20,000 mg/L $CaCl_2$.

Table 5-6: Long-term responses (greater than 24 hours) of fish to various concentrations of calcium chloride.

Exposure Concentration ($CaCl_2$ mg/L)	Exposure Concentration ($CaCl_2$ moles/L)	Toxic Response	Species of Concern	Exposure Time	Water Temperature (°C)	Reference
277	0.002	death	*Notropis blennius* (river shiner)	840-1176 hours (5-7 weeks)	room temperature	Garrey 1916 (in Doudoroff & Katz 1953)
555	0.005	killed or injured	*Ambloplites rupestris* (rock bass)	168 hours (1 week)	unknown	Ellis 1937 (in McKee and Wolf 1963)
832	0.007	death	*Notropis blennius* (river shiner)	336 - 504 hours (14-21 days)	room temperature	Garrey 1916 (in Doudoroff and Katz 1953)
2,775	0.025	death	*Notropis blennius* (river shiner)	48-96 hours (2-4 days)	room temperature	Garrey 1916 (in Doudoroff and Katz 1953)
5,000	0.045	average survival time	*Notemigonus crysoleucas* (golden shiners)	143.5 hours	22-22.5	Wiebe et al. 1934
5,000	0.045	death	*Notemigonus crysoleucas* (golden shiners)	143 hours	unknown	Ellis 1937 (in Anderson 1948)
9,500	0.086	50% mortality (TLm)	*Lepomis macrochirus* (bluegill sunfish)	96 hours	unknown	Cairns & Scheier 1958 (in McKee & Wolf 1963)
Gradual increase from 9,500 to 13,500	gradual increase from 0.086 to 0.122	death	fish	within 288 hours (within 12 days)	unknown	Young 1923 (in Hanes et al. 1970)
10,000	0.090	average survival time	*Lepomis macrochirus* (bluegill sunfish)	48.8 hours	22-22.5	Wiebe et al. 1934
10,650	0.096	50% mortality (LC_{50})	*Lepomis macrochirus* (bluegill sunfish)	96 hours	18 ± 2	Patrick et al. 1968
11,099	0.100	survival	*Anguilla japonica* (young eel)	50 hours	20-22	Oshima 1931 (in Doudoroff & Katz 1953)
13,400	0.121	50% mortality (TLm)	*Gambusia affinis* (mosquito-fish)	96 hours	unknown	Wallen et al. 1957 (in McKee & Wolf 1963)

5.9.3 Magnesium Chloride

Compared to the other salts, there is a limited amount of information on the effects of magnesium chloride on fish. As well, the available data was taken from old references (i.e., prior to 1958).

Short-term Exposure (24 hours or less)

Two studies were found on the short-term exposure of fish to magnesium chloride (Table 5-7 and Appendix B-3).

1) Iwao (1936) found *Orizias latipes*, a small freshwater cyprinodont, died within 24 hours of exposure to 23,809 mg/L $MgCl_2$. Water temperature and general water quality were not reported for this study.

2) Wiebe et al. (1934) investigated golden shiner (*Notemigonus crysoleucas*) survival times for various exposure times and concentrations of magnesium chloride. Test exposures extended to 96 hours. Average survival times of 96.5 hours, 4.6 hours, 0.8 hours and 0.5 hours for golden shiners were associated with exposures to 5,000 mg/L, 10,000 mg/L, 15,000 mg/L and 20,000 mg/L $MgCl_2$ respectively. Test conditions involved test water composed of an aerated mixture of distilled and tap water at 22-22.5 ^oC. In contrast, golden shiners survived for an average time of 27.6 hours in 10,000 mg/L $CaCl_2$ and 97 hours in 10,000 mg/L NaCl at 22-22.5 ^oC. This again suggests that magnesium chloride is most toxic to fish, followed by calcium chloride and then sodium chloride.

Table 5-7: Short-term responses (24 hours or less) of fish to various concentrations of magnesium chloride.

Exposure Concentration ($MgCl_2$ mg/L)	Exposure Concentration ($MgCl_2$ moles/L)	Toxic Response	Species of Concern	Exposure Time	Water Temperature (^oC)	Reference
10,000	0.105	average survival time	*Notemigonus crysoleucas* (golden shiners)	4.6 hours	22 - 22.5	Wiebe et al. 1934
15,000	0.158	average survival time	*Notemigonus crysoleucas* (golden shiners)	0.8 hours	22 - 22.5	Wiebe et al. 1934
20,000	0.210	average survival time	*Notemigonus crysoleucas* (golden shiners)	0.5 hours	22 - 22.5	Wiebe et al. 1934
23,809	0.250	death	*Oriziass latipes* (small freshwater cyprinodont)	24 hours	unknown	Iwao 1936 (in Doudoroff and Katz 1953)

Long-term Exposure (3-21 days)

Five studies were located on the long-term exposure of fish to magnesium chloride (Table 5-8 and Appendix B-3). The deaths of various fish species in many of these long-term exposure tests were associated with relatively small concentrations of magnesium chloride. Highlights are as follows:

1) Garrey (1916) reported the mortality of the river shiner (*Notropis blennius*) occurred after exposure to 476 mg/L $MgCl_2$ over 4 to 6 days at room temperature; however, this experiment was conducted in distilled water. Ellis (1937) also conducted experiments using distilled water and noted that minnows experienced mortality after being exposed for 4-6 days to 476 mg/L $MgCl_2$.

2) As previously noted, Wiebe et al. (1934) found that golden shiners (*Notemigonus crysoleucas*) survived an average of 96.5 hours when exposed to 5,000 mg/L $MgCl_2$ at 22 ^oC.

3) Comparatively, young eels (*Anguilla japonica*) survived 50 hours of exposure to 9,523 mg/L $MgCl_2$ in a water temperature of 20-22 ^oC (Oshima 1931); water quality was not reported. Mosquito-fish (*Gambusia affinis*) also had a high reported tolerance to magnesium chloride (Waller et al. 1957).

Table 5-8: Long-term responses (greater than 24 hours) of fish to various concentrations of magnesium chloride.

Exposure Concentration ($MgCl_2$ mg/L)	Exposure Concentration ($MgCl_2$ moles/L)	Toxic Response	Species of Concern	Exposure Time	Water Temperature (°C)	Reference
476	0.005	death	minnows	96-144 hours (4-6 days)	unknown	Ellis 1937 (in McKee and Wolf 1963)
476	0.005	death	*Notropis blennius* (river shiners)	96-144 hours (4-6 days)	room temperature	Garrey 1916 (in Doudoroff and Katz 1953)
2,381	0.025	death	*Notropis blennius* (river shiners)	~48 hours (2 days)	room temperature	Garrey 1916 (in Doudoroff and Katz 1953)
5,000	0.053	average survival time	*Notemigonus crysoleucas* (golden shiners)	96.5 hours	22-22.5	Wiebe et al. 1934
6,757	0.071	death	*Carassius auratus* (goldfish)	72-504 hours (3-21 days)	unknown	Ellis 1937 (in McKee and Wolf 1963)
9,523	0.100	survival	*Anguilla japonica* (young eel)	50 hours	20-22	Oshima 1931 (in Doudoroff and Katz 1953)
13,400	0.141	50% mortality (TLm)	*Gambusia affinis* (mosquito-fish)	96 hours	unknown	Wallen et al. 1957 (in McKee and Wolf 1963)

5.9.4 Potassium Chloride

Short-term Exposure (24 hours or less)

A number of studies were located which investigated the tolerance of fish to short-term exposure to potassium chloride. Waller et al. (1996) conducted the most comprehensive studies. The range of test concentrations was as low as 74.6 mg/L and as high as 32,800 mg/L KCl (Table 5-9 and Appendix B-4). Highlights are as follows:

1) A concentration of 74.6 mg/L KCl lead to the death of goldfish (*Carassius auratus*) within 4.5 to 15 hours (Ellis 1937). Distilled water was used in this test. While fish are often tolerant of distilled water for short periods (i.e., less than 1 day), there is the possibility that the distilled water may have influenced the results.

2) Dowden and Bennett (1965) observed 50% mortality in bluegill (*Lepomis macrochirus*) when exposed to a concentration of 5,546 mg/L KCl.

3) Waller et al. (1996) conducted extensive studies on the effects of potassium chloride on various fish species over exposure times of up to 24 hours. Test conditions were 2,500 or 10,000 mg/L KCl, exposure times of 3, 6 or 24 hours, water temperatures of 12 or 17 ^{o}C, and a water hardness of 140$\pm$10 mg/L $CaCO_3$. Walleye experienced 100% mortality when exposed for 24 hours to 2,500 mg/L KCl at 12 ^{o}C. Yellow perch (46.7% mortality), channel catfish (3.9% mortality), and smallmouth bass (3.9%) were less sensitive. The least sensitive were bluegill sunfish, rainbow trout, fathead minnows, brown trout, lake trout which experienced no mortality under these conditions.

4) The sensitivity to potassium chloride increased with temperature (Waller et al. 1996). While yellow perch experienced a mortality rate of 46.7% at concentrations of 2,500 mg/L KCl, exposure times of 24 hours and water temperatures of 12^{o}C, mortality increased to 80% when the experiment was conducted at 17 ^{o}C. Similarly, bluegill mortality increased from 0% to 20% for the two temperatures and test conditions.

5) Waller et al.'s (1996) study also provides information on the relative sensitivity of yellow perch to chloride salts. As noted above, yellow perch experienced 80% mortality when exposed to 2,500 mg/L KCl for 24 hours at 17^{o}C. In contrast, yellow perch required exposure to 10,000 mg/L $CaCl_2$ to attain a mortality of 83%. Yellow perch exposed for 24 hours to 10,000 mg/L NaCl experienced 0% mortality. This suggest that potassium chloride is more toxic to yellow perch than calcium chloride and sodium chloride is the least toxic.

Table 5-9: Short-term responses (24 hours or less) of fish to various concentrations of potassium chloride.

Exposure Concentration (KCl mg/L)	Exposure Concentration (KCl moles/L)	Toxic Response	Species of Concern	Exposure Time	Water Temperature (°C)	Reference
74.6	0.001	death	*Carassius auratus* (goldfish)	4.5-15 hours	unknown	Ellis 1937 (in McKee and Wolf 1963)
373	0.005	death	minnows	12-29 hours	unknown	Ellis 1937 (in McKee and Wolf 1963)
400	0.005	death	*Notropis blennius* (river shiner)	12-29 hours	room temperature	Garrey 1916 (in Hammer 1977; Doudoroff and Katz 1953)
1937	0.026	death	*Oriziass latipes* (small freshwater cyprinodont)	24 hours	unknown	Iwao 1936 (in Doudoroff and Katz 1953)
2,500	0.034	3.9% mortality	*Ictalurus punctatus* (channel catfish)	24 hours	12	Waller et al. 1996
2,500	0.034	0% mortality	*Lepomis macrochirus* (bluegill sunfish)	24 hours	12	Waller et al. 1996
2,500	0.034	3.9% mortality	*Micropterus dolomieu* (smallmouth bass)	24 hours	12	Waller et al. 1996
2,500	0.034	0% mortality	*Oncorhynchus mykiss* (rainbow trout)	24 hours	12 and 17 (two different experiments)	Waller et al. 1996

Table 5-9: Continued

Exposure Concentration (KCl mg/L)	Exposure Concentration (KCl moles/L)	Toxic Response	Species of Concern	Exposure Time	Water Temperature (°C)	Reference
2,500	0.034	0% mortality	*Pimephales promelas* (fathead minnow)	24 hours	12 and 17 (two different experiments)	Waller et al. 1996
2,500	0.034	46.7% mortality	*Perca flavescens* (yellow perch)	24 hours	12	Waller et al. 1996
2,500	0.034	0% mortality	*Salmo trutta* (brown trout)	24 hours	12	Waller et al. 1996
2,500	0.034	0% mortality	*Salvelinus namaycush* (lake trout)	24 hours	12	Waller et al. 1996
2,500	0.034	100% mortality	*Stizostedion vitreum* (walleye)	24 hours	12 and 17 (two different experiments)	Waller et al. 1996
2,500	0.034	0% mortality	*Ictalurus punctatus* (channel catfish)	24 hours	17	Waller et al. 1996
2,500	0.034	20.0% mortality	*Lepomis macrochirus* (bluegill sunfish)	24 hours	17	Waller et al. 1996
2,500	0.034	80% mortality	*Perca flavescens* (yellow perch)	24 hours	17	Waller et al. 1996
5,500	0.074	50% mortality (TLm)	*Lepomis macrochirus* (bluegill sunfish)	24 hours	unknown	Abegg 1949, 1950 (in Doudoroff and Katz 1953)

Table 5-9: Continued

Exposure Concentration (KCl mg/L)	Exposure Concentration (KCl moles/L)	Toxic Response	Species of Concern	Exposure Time	Water Temperature (°C)	Reference
5,546	0.095	50% mortality	*Lepomis macrochirus* (bluegill sunfish)	24 hours	Unknown	Dowden and Bennett 1965
7,700	0.103	death	*Carassius auratus* (goldfish)	4.6 - 15 hours	~21	Powers 1917 (in Hammer 1977) Doudoroff and Katz 1953)
10,000	0.134	0% mortality	*Salvelinus namaycush* (lake trout)	3 hours	12	Waller et al. 1996
10,000	0.134	0% mortality	*Stizostedion vitreum* (walleye)	3 hours	12	Waller et al. 1996
10,000	0.134	0% mortality	*Pimephales promelas* (fathead minnow)	6 hours	12	Waller et al. 1996
10,000	0.134	0% mortality	*Salmo trutta* (brown trout)	6 hours	12	Waller et al. 1996
10,000	0.134	0% mortality	*Salvelinus namaycush* (lake trout)	6 hours	12	Waller et al. 1996
10,000	0.134	22.1% mortality	*Oncorhynchus mykiss* (rainbow trout)	6 hours	12	Waller et al. 1996
10,000	0.134	50% mortality	*Micropterus dolomieu* (smallmouth bass)	6 hours	12	Waller et al. 1996
10,000	0.134	63.3% mortality	*Stizostedion vitreum* (walleye)	6 hours	12	Waller et al. 1996

Table 5-9: Continued

Exposure Concentration (KCl mg/L)	Exposure Concentration (KCl moles/L)	Toxic Response	Species of Concern	Exposure Time	Water Temperature (°C)	Reference
10,000	0.134	30% mortality	*Oncorhynchus mykiss* (rainbow trout)	3 hours	17	Waller et al. 1996
10,000	0.134	6.7% mortality	*Stizostedion vitreum* (walleye)	3 hours	17	Waller et al. 1996
10,000	0.134	93.3% mortality	*Oncorhynchus mykiss* (rainbow trout)	6 hours	17	Waller et al. 1996
10,000	0.134	93.3% mortality	*Stizostedion vitreum* (walleye)	6 hours	17	Waller et al. 1996
10,000	0.134	0% mortality	*Ictalurus punctatus* (channel catfish)	3 hours	12 and 17 (two different experiments)	Waller et al. 1996
10,000	0.134	0% mortality	*Lepomis macrochirus* (bluegill sunfish)	3 hours	12 and 17 (two different experiments)	Waller et al. 1996
10,000	0.134	0% mortality	*Pimephales promelas* (fathead minnow)	3 hours	12 and 17 (two different experiments)	Waller et al. 1996
10,000	0.134	0% mortality	*Ictalurus punctatus* (channel catfish)	6 hours	12 and 17 (two different experiments)	Waller et al. 1996

Table 5-9: Continued

Exposure Concentration (KCl mg/L)	**Exposure Concentration (KCl moles/L)**	**Toxic Response**	**Species of Concern**	**Exposure Time**	**Water Temperature (°C)**	**Reference**
10,000	0.134	0% mortality	*Lepomis macrochirus* (bluegill sunfish)	6 hours	12 and 17 (two different experiments)	Waller et al. 1996
10,000	0.134	0% mortality	*Perca flavescens* (yellow perch)	6 hours	12 and 17 (two different experiments)	Waller et al. 1996
12,060	0.162	50% mortality (TLm)	*Stizostedion vitreum* (walleye)	24 hours	unknown	ORVWSC 1950 (in McKee and Wolf 1963)
32,800	0.440	death	*Carassius auratus* (goldfish)	0.23 - 0.28 hours	~21	Powers 1917 (in Hammer 1977; Doudoroff and Katz 1953)

Long-term Exposure (50-168 hours)

Seven studies were found on the long-term exposure of fish to potassium chloride. Test concentrations ranged from 500-6,209 mg/L KCl and exposure times of 50-168 hours (Table5-10 and Appendix B-4). Highlights are as follows:

1) Fathead minnows had no detectable mortality when exposed to 500 mg/L KCl for 7 days. The LOEC was 1,000 mg/L for a 7-day exposure (Pickering et al. 1996); minnows were 1 to 7 days old. Beak (1999) used potassium chloride as a reference toxicant in its 7-day fathead minnow survival and growth test, using larva that were less than 1 day old. In this instance, the 7-day LC_{50} was 861 mg/L KCl. This concentration was considerably lower than the LC_{50} of 5,490 mg/L NaCl.

2) Fifty percent of mosquito-fish died when exposed for 96 hours to 920 mg/L KCl in turbid waters (Wallen et al. 1957).

Table 5-10: Long-term responses (greater that 24 hours) of fish to various concentrations of potassium chloride.

Exposure Concentration (KCl mg/L)	Exposure Concentration (KCl moles/L)	Toxic Response	Species of Concern	Exposure Time	Water Temperature (^oC)	Reference
500	0.007	NOEC	*Pimephales promelas* (fathead minnow) 1-7 days old	168 hours (7 days)	25$\pm$1	Pickering et al. 1996
920	0.012	50% mortality (TLm)	*Gambusia affinis* (mosquito-fish)	96 hours	unknown	Wallen et al. 1957 (in Hammer 1977; McKee and Wolf 1963)
1,000	0.013	LOEC	*Pimephales promelas* (fathead minnow) 1-7 days old	168 hours (7 days)	25$\pm$1	Pickering et al. 1996
1,360	0.018	death	*Perca flavescens* (yellow perch)	72 hours (3 days)	unknown	Young 1923 (in McKee and Wolf 1963)
2,000	0.027	50% mortality (TLm)	*Lepomis macrochirus* (bluegill sunfish)	96 hours	unknown	Trama 1954 (in Hammer 1977)
2,010	0.027	50% mortality (TLm)	*Lepomis macrochirus* (bluegill sunfish)	96 hours	unknown	Anon. 1960 (in McKee and Wolf 1963)
2,010	0.027	50% mortality	*Lepomis macrochirus* (bluegill sunfish)	96 hours	18$\pm$2	Patrick et al. 1968
6,209	0.083	survival	*Anguilla japonica* (young eel)	50 hours	20-22	Oshima 1931 (in Doudoroff and Katz 1953)

3) Bluegill sunfish experienced a 50% mortality rate when exposed to ca. 2,000 mg/L KCl for 96 hours (Trama 1954; Anon. 1960; Patrick et al. 1968).

4) Young eels were able to survive for 50 hours in 6,209 mg/L KCl (Oshima 1931).

5.10 Amphibians

Two studies were located which investigated the toxicity of chloride salts to amphibians. While useful, neither was sufficient in quantifying the effects of high chloride concentrations on egg survivorship. Accordingly, additional data was obtained by contacting Beak International to conduct additional toxicity testing. Highlights of the studies are as follows:

1) Padhye and Ghate (1992) investigated the short-term effects of different concentrations of sodium and potassium chloride on embryos and tadpoles of the frog, *Microhyla ornata*, a species found in India. Embryos treated with 2,000 mg/L (0.2%) KCl experienced swollen heads which increased their buoyancy, causing them to float with their ventral sides upward. Exposure to sodium chloride at concentrations of 5,000-6,000 mg/L (0.5% to 0.6%) resulted in incomplete closure of the neural tube. Both potassium and sodium chloride caused significant reduction in swelling of the perivitelline space. Tadpoles were observed to be somewhat more resistant to both salts compared to embryos. However, potassium chloride was found to be more toxic than sodium chloride. For late gastrula stage embryos, the 24-hr LC_{50} was 6,482 mg/L NaCl and 5,000 mg/L KCl. The 96-hr LC_{50} decreased to 2,711 mg/L NaCl and 1,414 mg/L KCl respectively. Eight-day old tadpoles had an LC_{50} of 5,027 mg/L NaCl and 1,593 mg/L KCl while hind-limb tadpoles had an LC_{50} of 6,929 mg/L NaCl and 2,539 mg/L KCl. Embryos were more sensitive than tadpoles to sodium and potassium chloride, but both were more sensitive to potassium than sodium chloride.

2) Exposure to high concentrations of sodium and potassium chloride reduced enzyme activity in the estivating Couch's spadefoot toad, *Scaphiopus couchi*, and the non-estivating leopard frog, *Rana pipiens* (Grundy and Storey 1994). The former species is found in the southeastern United States while the latter is found throughout much of provincial Canada and the central United States. Estivation is the summer equivalent to hibernation and is done to avoid heat and/or drought. The addition of 11,690 mg/L NaCl (200 mM) resulted in an 81% reduction in the maximum reaction rate, or Vmax, of pyruvate kinase for the toad and an 87% reduction for the frog. For the enzyme phosphofructokinase, these reductions were 55% for the toad and 86% for the frog. The addition of 12,000 mg/L KCl (200 mM) resulted in a 26% reduction in the toad and a 35% reduction in the frog maximum reaction rate of pyruvate kinase. These additions reduced the maximum reaction rate of phosphofructokinase by 42% for the toad and 12% for the frog. Overall, sodium chloride had a stronger impact on enzyme function than potassium chloride. The implication of this to frog and toad survivorship was not investigated.

3) The Beak study (1999) investigated the 7-day survivorship of tadpoles (<2 weeks old) of the African clawed frog, *Xenopus leavis*, in 6 concentration levels (250, 500, 1,000, 2,000, 4,000, and 8,000 mg/L NaCl) sodium chloride in addition to control populations. Experiments were conducted at 23^{o} C. The studies were initiated too late to use a native Canadian frog species. Survivorship at the end of 7 days ranged from 90-97% at concentrations up to 2,000 mg/L NaCl, decreasing to 6.7% at 4,000 mg/L NaCl and to 0% for the 8,000 mg/L NaCl treatment. For the 4,000 and 8,000 mg/L NaCl treatments, most mortality occurred on day 2. Impaired swimming behaviour also was noted. The 7-day LC_{50} was estimated at 2,940 mg/L NaCl and the 7-day EC_{50} was estimated at 2,510 mg/L NaCl. Potassium chloride was used as a reference toxicant and had similar toxicity as sodium chloride (i.e., an LC_{50} and EC_{50} of 2,230 mg/L KCl). This is in contrast to the Padhye and Ghate (1992) study that reported a greater sensitivity of the frog, *Microhyla ornata*, to potassium than sodium chloride when mortality was the test response. Grundy and Storey (1994) noted a greater reduction in enzyme function when Couch's spadefoot toad and the leopard frog were exposed to sodium rather than potassium chloride.

5.11 Aquatic Birds

One study was found that investigated the sublethal effects of sodium chloride on aquatic birds. Hughes et al. (1991) determined that Peking ducks, *Anas platyrhynchos*, that drank salty (17,354 mg/L NaCl) water for one month had 73% of their body mass as body water compared to 63% for ducks that drank freshwater.

5.12 Additional Considerations

5.12.1 Ion Interactions

A physiologically-balanced salt solution contains different salts, particularly sodium, calcium, magnesium and potassium, in proportions that neutralize or reduce the specific toxicity of each through antagonistic actions (Doudoroff and Katz 1953). Therefore, the toxicity of cations to aquatic organisms can be counteracted or sometimes enhanced by other cations in solution (Doudoroff and Katz 1953). Three studies have investigated ion interactions and toxicity.

1) Garrey (1916) noted that a relatively small amount of calcium chloride (20-40 mg/L $CaCl_2$) added to a solution of sodium, magnesium or potassium chloride in distilled water reduced the toxicity of that chloride solution to minnows (*Notropis* sp.).

2) Grizzle and Mauldin (1995) found that increasing the concentration of calcium ions decreased the toxicity of sodium chloride to juvenile striped bass (*Morone saxatilis*). The lethal concentration resulting in 50% mortality of juveniles increased from 1,400 mg/L NaCl to 18,200 mg/L NaCl as the calcium concentration increased from 3.0 to 100 mg/L Ca^{2+}. Test conditions involved 24-hour

exposures to sodium chloride and calcium chloride added to water from a hatchery reservoir at 21-22 ^{o}C. The decreasing toxicity was thought to be a result of the reduction in the ratio of $Na^{+}:Ca^{2+}$ resulting from the addition of the calcium. A low-calcium/high-sodium environment also adversely affected red drum (*Sciaenops ocellatus*) and channel catfish (*Ictalurus punctatus*). The $Na^{+}:Ca^{2+}$ ratio was considerably lower in certified NaCl (33,713) and food grade salt (17,282) than rock salt (85) and rock salt mixed with calcium chloride (53). The authors hypothesized that rock salt has enough calcium to prevent NaCl toxicity under the conditions employed in the study.

3) Borgmann (1996) determined the aqueous ionic requirements of the freshwater amphipod, *Hyalella azteca*. Sodium (13.8 mg/L) and bicarbonate (122.0 mg/L) were essential ions with amphipods dying rapidly when reared in distilled water. Magnesium (8.51 mg/L) improved survival and growth, but was toxic in the absence of calcium. Potassium (1.6 mg/L) improved growth and the production of young, but had no effect on survivorship. In contrast, sulphate (33.6 mg/L) and chloride (22.7 mg/L) had no effect on amphipod survivorship, growth, and reproduction.

4) Saline lakes support a variety of plant and animal species although diversity tends to be lower in comparison to fresh water. These lakes, while generally carbonate, sulfate, or chloride dominated, typically are elevated in all the major salts (Cumming and Smol 1993; Fritz et al. 1993; Herbst 1988; Wood and Talling 1988). Complex interactions among sodium, potassium, magnesium, and chloride ions may play an unknown role in extending the salinity tolerances of species inhabiting these lakes.

5.12.2 Road Salt Toxicity

Few studies have investigated the toxicity of pure formulations of road salt and direct road salt runoff. These studies are as follows:

1) Peter Meier (School of Public Health, Ann Arbor, Michigan) and his graduate student, Blaise Blastos, have been comparing the toxicity of sodium chloride, calcium chloride, deicing salt, and road salt. For Ceriodaphnia dubia, calcium chloride was the most toxic followed by deicing salt, sodium chloride and road salt; however, toxicity varied by only a factor of 1.6 across the four salts considered (Table 5-11). The LC_{50} for sodium chloride was similar to that (2-day) observed by Cowgill and Milazzo (1990) for C. dubia. Statistical tests were not performed to assess whether these differences were significant.

2) For the fathead minnow, Meier and Blastos determined that deicing salt was the most toxic, followed by sodium chloride, and then calcium chloride; toxicity varied by a factor of 1.8 across the three salts considered. The LC_{50} for fathead minnow was similar to that (96-hour) reported by Adelman et al. (1976). Overall, there was neither a large or consistent difference between road salt and sodium chloride toxicity, with sodium chloride some 1.2-1.4 times more toxic than road salt. Fathead minnows were more tolerate of all three salts than C. dubia.

3) Meier and Blastos also exposed *Daphnia magna* and fathead minnows to road salt. The LC_{50} for road salt was 4,390 mg/L for *D. magna* and 9,410 mg/L for fathead minnow. Fathead minnows appeared to be more sensitive to deicing salts than road salt. Cowgill and Milazzo (1990) reported a 2-day LC_{50} of 7,754 mg/L NaCl. This may suggest that *D. magna* are more sensitive to road salt than sodium chloride.

Table 5-11: Toxicity of the zooplankton, *Ceriodaphnia dubia* and *Daphnia magna*, and the fathead minnow, *Pimephales promelas*, to sodium chloride, calcium chloride, road salt and deicing salt.

	Ceriodaphnia dubia		*Daphnia magna*		*Pimephales promales*	
	EC_{50} (gm/L)	95% Confidence Interval	EC_{50} (gm/L)	95% Confidence Interval	EC_{50} (gm/L)	95% Confidence Interval
Sodium Chloride (NaCl)						
Test 1	2.28	2.14-2.43	-	-	7.17	6.92-7.44
Test 2	2.40	2.30-2.50	-	-	7.24	6.90-7.60
Mean	2.34	-	-	-	7.21	-
Calcium chloride ($CaCl_2$)						
Test 1	1.03	0.89-1.19	-	-	9.52	8.68-10.45
Test 2	1.84	1.64-2.08	-	-	-	-
Mean	1.44	-	-	-	9.52	-
Deicing salt						
Test 1	2.14	1.76-2.60	-	-	5.96	5.26-7.74
Test 2	1.78	1.52-2.08	-	-	4.33	3.78-4.96
Test 3	-	-	-	-	9.23	7.98-10.67
Test 4	-	-	-	-	5.50	5.05-5.98
Mean	1.96		-	-	6.26	
Road Salt						
Test 1	-	-	4.42	4.21-4.63	9.25	8.93-9.58
Test 2	-	-	4.45	4.18-4.75	9.57	9.91-9.96
Test 3	-	-	4.31	4.06-4.58	-	-
Mean	-	-	4.39		9.41	

Source: P. Meier and B. Blastos (School of Public Health, the University of Michigan, Ann Arbor, Michigan), with permission.

4) Rokosh et al. (1997) also conducted bioassays using road salt. Rainbow trout, *Daphnia*, and *Ceriodaphnia dubia* test results were similar to those reported using the same species of animals and sodium chloride. Given that road salts consist of various formulations (i.e., mixtures of sodium and calcium chlorides), depending on the manufacturer, these comparative data sets are too small to convincingly quantify differences in toxicity between sodium chloride and road salt formulations. Differences, if they exist, are probably small.

5.12.3 Road Runoff Toxicity

A number of researchers have investigated the toxicity of road runoff (Buckler and Granato 1999) although such studies have not always considered snowmelt contaminated by road salt. Runoff from such roads contains not only sodium and calcium chlorides but other salts, ferrocynanides, trace metals, organic contaminants (i.e., polynuclear aromatic hydrocarbons) and nutrients (Marsalek et al. 1999 a, b). Fuel is a source of lead, brake linings a source of copper, vehicles types a source of zinc and cadmium, while deicing salts can contain chromium and copper (Maltby et al. 1995a). Therefore, snowmelt highway runoff could be expected to be more toxic than sodium chloride alone, particularly when the runoff originates from heavily used highways. A few studies were reviewed for this document. Study highlights are as follows.

1) Isabelle et al. (1987) found that species diversity, evenness, and richness decreased significantly when increasing proportions of snowmelt were used as the test water. Only two species, the common cattail, *Typha latifolia*, and the purple loosestrife, *Lythrun salicaria*, germinated in pure roadside snowmelt. However, no analyses were conducted on the snowmelt to determine the chemical composition of the test water. Field observations by Isabelle et al. also revealed that species such as the flat-topped white aster, *Aster umbellatus* and the three-way sedge, *Dulichium arundinaceum* were intolerant to snowmelt and are uncommon in wet roadside ditches in urban Ontario. Dominant species are those most tolerant to snowmelt (e.g., *Lythrum salicaria* and *Typha latifolia*). It is not clear whether toxicity was a result of exposure to road salts or to other contaminants, such as metals that are associated with snowmelt from road surface.

2) Kszos et al. (1990) investigated the toxicity of Chautauqua Lake Bridge, New York, runoff to young-of-the-year bluegill sunfish (*Lepomis macrochirus*). Sodium chloride concentrations were highest in winter (mean 47,200 mg/L NaCl) than in spring (mean 16,780 mg/L NaCl) and autumn (mean 1,652 mg/L NaCl). Fish exposed to 25% of fall bridge runoff and to 50% concentrations of spring and fall runoff actually had a greater survivorship than control fish. This improved survivorship was related to the beneficial effect of sodium chloride that reduced the incidence of disease and bacterial infections in the experimental fish populations. Similarly, Rantamaki et al. (1992) noted that the addition of 1,900 mg/L $MgCl_2$ prevented the transmission of the crayfish plague fungus, *Aphanomyces astaci*, to the freshwater crayfish, *Astacus astacus*. For the winter bioassay, fish exposed to 50% winter runoff had lower survival than the controls. This toxicity was related to sodium chloride concentrations. However, zinc, copper, and cadmium were present is sufficient concentrations to have acted additively or synergistically with sodium chloride.

3) Maltby et al. (1995 a, b) investigated the effects of motorway runoff on freshwater ecosystems in northern England; road salt was not measured in this study. Stream waters downstream of the motorway were elevated in polyaromatic hydrocarbons (PAH), zinc, lead, and copper. Copper, zinc, and PAH concentrations were higher in downstream than upstream sediments. Survivorship of the amphipod, *Gammarus pulex*, was slightly reduced when exposed to these contaminated sediments for 14 days. Toxicity was associated primarily with the PAH fractions. This small stream (about 0.9 m wide and 0.02 m deep) received runoff water from a 1,500 m length of the M1 motorway.

4) Rokosh et al. (1997) tested the toxicity of road runoff to zooplankton and rainbow trout. Stormwater runoff occasionally was toxic to rainbow trout, more commonly toxic to *Daphnia magna*, and most commonly (6 out of 15 samples) toxic to *Ceriodaphnia dubia.* Snodgrass et al. (2000) determined that while toxicity of road salt runoff was associated with chloride concentrations, salt concentration alone did not always explain toxicity. Metals associated with the runoff, such as copper and zinc, apparently enhanced toxicity.

5) Marsalek et al. (1999 a, b) investigated the toxicity of urban and highway runoff at several sites in southern Ontario. Runoff from multilane divided highway (MLDH) sites was considerably more toxic than combined sewer overflows (Marsalek et al. 1999a). In a related study, Marsalek et al. (1999b) found that almost 20% of MLDH samples were severely toxic in comparison to 1% of stormwater samples. A battery of tests was employed using *Daphnia magna,* Microtox (a bacteria), sub-mitochondrial particles, and the SOS Chromotest for genotoxicity. Winter runoff from sites such as the Skyway Bridge in Burlington, Ontario, were considerably more toxic than summer runoff. Increased winter toxicity was associated with the accumulation of contaminants in snow, high concentrations of road salt, and the enhanced mobility of chloride laded runoff (Marsalek et al. 1999b).

6) Novotny et al. (1998) examined concentrations of salts and metals in urban snow melt. Concentrations were elevated along roadways, with higher concentrations occurring in commercial versus residential areas.

5.12.4 Indirect Toxicity Effects of Road Salts

Road salts can have indirect effects on aquatic systems. Chloride salts, for example, tend to be more soluble than carbonate salts. Chlorides can thus, through various reactions, enhance the mobility of trace metals in aquatic ecosystems. Road salt, by affecting the density of water, can affect mixing process in lakes. This, in turn, can affect many aspects of the ecological functioning of that ecosystem. A few examples of such studies are as follows:

1) Increased salt concentrations on the bottom of lakes or streams can lead to the release of metals from sediments. Wang et al. (1991) found that 709 mg/L Cl^- (or 0.02 M) substantially enhanced the

release of mercury from freshwater sediments. Mercury can be acutely toxic to invertebrate species and fish in concentrations as low as 0.002 mg/L and 0.02 mg/L, respectively (CCME 1991). Since mercury accumulates in the tissues of fish, the Canadian Council of Ministers of the Environment (CCME) has established a criterion of 0.0001 mg/L of total mercury for the protection of consumers of fish (CCME 1991). Sodium chloride also enhances mercury mobilization from soils (MacLeod et al. 1996).

2) By competing for particulate binding sites, sodium chloride acts as an enhancer of dissolved and potentially bioavailable trace metals such as cadmium, copper and zinc in aquatic ecosystems (Warren and Zimmerman 1994). Cadmium, copper, and zinc are acutely toxic to aquatic organisms at concentrations as low as 0.001 mg/L (rainbow trout), 0.0065 mg/L (*Daphnia magna*) and 0.09 mg/L (rainbow trout), respectively (CCME 1991).

3) The formation of meromixis can have a number of impacts on lakes. The low oxygen conditions which develop below the chemocline can result in the loss of all, but the most resilient, deep water benthic species. Zooplankton may be excluded from their deep water daytime refuges, being forced to live in the well-lit surface layers where they may become more vulnerable to size-selective fish predation. Hypolimnetic fish species, such as lake trout, may also be adversely affected.

4) The onset of lake meromixis will affect sediment water exchanges. Phosphorus and various metals are more readily released from low oxygen than well oxygenated sediments (Wetzel 1983). This increase in phosphorus release from the sediments may enhance the productivity of the lake, particularly if there is sufficient exchange at the chemocline (Smol et al. 1983). Furthermore, regenerated nitrogen will be dominated by ammonia rather than nitrate.

5.13 Summary of Chloride Salt Toxicity

General trends as illustrated by the data collected on lethal toxicity of road salts and their additives are as follows:

1) The tolerance to elevated salt concentrations decreases with increasing exposure time. Short-term exposures to concentrations of salts in the hypersaline range (>50,000 mg/L salinity) may kill adult fish and other organisms rapidly (e.g., 15 minutes).

2) As exposure time increases, salinity tolerance decreases.

3) Tolerance to salinity can be increased through the gradual increase in salinity allowing the organism to develop mechanisms for dealing with the osmotic shock and other physiological stresses.

4) Aquatic biota are more tolerant of salts in water at higher oxygen concentrations than lower concentrations.

5) While some studies suggest that organisms are more tolerant to salinity at lower water temperatures, other studies have shown the converse.

6) Zooplankton and benthic invertebrates appear to be relatively more sensitive to sodium chloride concentrations than fish.

7) Within a given taxonomic category (e.g., benthic invertebrates or fish) there is significant species variation in salinity tolerances.

8) Potassium chloride tends to be the most toxic salt to aquatic organisms. Magnesium chloride is next in toxicity followed by calcium chloride and then sodium chloride for invertebrates and adult fish. Fish fry may be more tolerant of high concentrations of calcium rather than sodium chloride.

9) Limited studies have been conducted of the toxicity of road salts and deicing salts to aquatic organisms. In general, toxicity is within the same general range of that observed for sodium and calcium chlorides.

10) Road salts, by increasing the mobilization of metals, may enhance the toxicity and adverse environmental impacts of road runoff. Nutrients and organic contaminants may also be carried with this runoff, especially from heavily trafficked highways. This also can contribute to toxic stress.

Road salt, by affecting the density of water, can create meromictic conditions in lakes. This can create new forms of environmental stress, particularly to deep water communities

6.0 BIOLOGICAL EFFECTS OF ROAD SALT APPLICATION ON STREAM, WETLAND, AND LAKE ECOSYSTEMS IN CANADA

6.1 Introduction

This section discusses the effects of road salts on the biological components of lakes, streams and wetlands within the various regions of Canada (i.e., Maritimes, Central Canada, Prairie Provinces, West Coast and Rocky Mountains, as well as Northern Canada). It concludes with a broader discussion of salt in the environment.

A very small number of studies were located which investigated the impacts of road salt on aquatic ecosystems in Canada. Accordingly, studies conducted in the United States also are included in this section of the report. Included among these are studies investigating the impacts of saline seepage on stream communities. The biological components affected include densities of bacteria and algae, drift of stream benthic invertebrates, as well as diversity and community structure of aquatic invertebrates.

The majority of studies found relating to the effect of road salts were conducted in Central Canada (i.e., Ontario and Quebec) and were based on investigations of the use of road salts in winter for the deicing of roads. The results of the various studies are summarized in the text provided below, as well as, in Appendix C. This appendix includes a description of the ecological characteristic or species affected, the location of the effect, the response of the ecosystem to the salt loading, the duration of the response, the baseline or upstream concentration, the value of the main loadings into the ecosystem, the new concentration after loading, and the duration of the new concentration. Rarely were values provided for the main loadings into the ecosystem. No studies were found regarding the consequences to aquatic ecosystems on the application of calcium chloride as a dust suppressant.

6.2 Streams

Several studies investigated the impacts of road salt on stream ecosystems. These studies illustrate the complex ways in which road salt may affect these ecosystems. Results are as follows:

1) In 1973, Dickman and Gochnauer (1978) conducted a 28-day experiment on the density of bacteria and algae in Heyworth Stream, near Heyworth, Quebec. The authors noted that the National Association of Corrosion Engineers had studied the chloride content of 25 separate storm sewers over a three and a half-year period during which 175 samples were analyzed. Seventy-four samples (42%) had chloride concentrations in excess of 1,000 mg/L. The experiment was conducted from July 24 to October 1, 1973. Sodium chloride was added at four locations along the stream in order to maintain chloride concentrations of 1,000 mg/L (1,653 mg/L NaCl). The stream was described as being shallow and fast flowing and densely shaded. Artificial substrates (tiles) were placed in the creek at the experimental site (F) and an upstream control (A) and recovered at weekly intervals. Sodium chloride concentrations at site A were 2-3 mg/L. Conclusions were as follows:

a) Algal diversity was consistently lower at site F than site A. This decrease in diversity was related to increased osmotic stress.

b) Autrophic (photosynthetic periphyton or algae) standing crops were significantly lower on tiles incubated at the experimental site where sodium chloride concentrations were elevated. This decrease in standing stock also was related to osmotic stress.

c) Auxospore (a resting stage) formation of the diatom *Cocconeis placentula* were noted at site F on day 28, but not site A. Auxospore formation is often triggered by environmental stress.

d) Bacteria density was enhanced by exposure to 1,000 mg/L NaCl. This was believed to be due to a reduction in the grazing pressure on the bacterial population because of the reduced number of grazers, such as flagellates, ciliates, and rhizopods. Sodium chloride, per se, was not believed to have stimulated bacterial growth.

e) The incidence of diatom parasitism was lower at site F (2%) than at site A (7%), possibly because the sodium chloride inhibited fungal growth. Other researchers have related reductions in fungal and disease infections to elevated chloride levels (Rantamaki et al. 1992; Kszos et al. 1990).

Overall, these studies suggest that continuous exposure to sodium chloride concentration as low as 1,000 mg/L NaCl (580 mg/L Cl^-) for time periods as short as one week can result in changes in stream periphyton communities. Furthermore, the periphyton community may not recover from these stresses, with community composition continuing to remain different from upstream controls one month after continued exposure to this stress. This further suggests that pulses of sodium chloride-laden water during spring melt will have pronounced effects on the periphyton community as will the continued released of high concentrations of sodium chloride from stream banks during summer and autumn months. Periphyton form the base of the food web in many creeks, being grazed upon by invertebrates, which in turned serve as forage for fish. Reduced periphyton algal concentrations may have adverse effects at higher trophic levels. Leaf litter is another important source of food for stream communities. However, invertebrates obtain their nutrition from the microbial and fungal community rather than the litter itself. Reduced fungal biomass, as a result of increased NaCl concentrations may also impact invertebrate, forage fish, and, ultimately, predatory fish communities.

2) Dussart (1984) conducted a study of the effects of motorway runoff on the ecology of stream algae in various streams in Cumbria and North Lancashire, England. This study, along with a companion invertebrate study, was conducted in response to concerns that M6 highway runoff was adversely affecting trout stocks. Seven streams were sampled on March 26, 1976 and one stream resampled on April 5, 1976. Streams were sampled upstream and downstream of the highway. Chloride concentrations were not reported in this manuscript, but possibly are available in other literature reported in this note. Highlights of the study are as follows.

a) Total algal and filamentous algae were significantly more abundant downstream than upstream of the motorway. Species diversity also was higher.

b) The distribution patterns of six algal species appeared to be affected by the motorway, but these effects were not described.

c) Although road salt was used on the motorway in the weeks preceding the study, Dussart did not believe that chloride had directly affected the stream algae community because there was no evidence of a reduction of diversity and biomass downstream of the motorway. Rather, he hypothesized that the algae were responding to nutrient enhancement. Vehicle oils contain organic phosphates and some of these phosphates may enter streams with runoff, enhancing algal growth, especially nutrient-poor streams as in this study.

d) The companion study reported a decrease in macroinvertebrate diversity downstream of the motorway. This decline in invertebrate species may have resulted in decreased grazing pressure on the algae, resulting in increased abundance and biomass downstream of the motorway. The reason for the decreased invertebrate diversity was not discussed, but could be related to increased benthic drift, loss of species, etc., as described below from other studies.

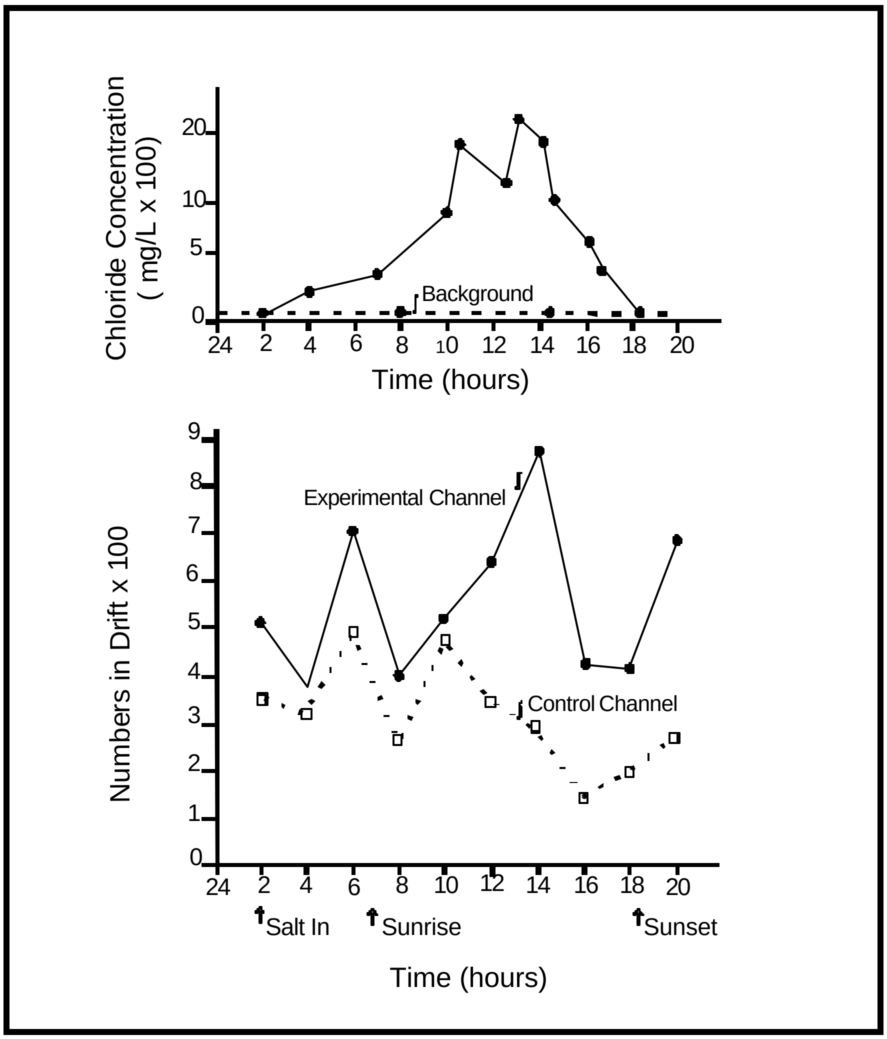
3) Crowther and Hynes (1977) experimentally investigated the effect of chloride concentrations on the drift of benthic invertebrates in Laurel Creek, located in southern Ontario. This creek is a small (annual discharge = 19 x 10^6 m^3) tributary to the Grand River that passes through urban Waterloo. At that time, Waterloo had a population of 32,000. Therefore, the general results of Crowther and Hynes' study may apply to the creeks flowing through the numerous small towns in certain regions, such as southern Ontario, Quebec, and the Maritimes, where road salt is heavily used. As previously noted (section 5), chloride levels in Laurel Creek vary seasonally with peak concentrations as high as 1,770 mg/L Cl^- being observed for a period spanning January 5 to February 6, 1975. A series of experiments were conducted in Lutteral Creek, a spring fed tributary with fauna characteristic of a small healthy trout stream, in order to assess the impacts of pulses of sodium chloride-laden water on benthic drift. Fifteen meters of Lutteral Creek were divided longitudinally by sheets of corrugated steel and benthic drift assessed on the control side and experimental side of the partition. Highlights are as follows:

a) In November 1974 and February 1975, there was no difference in benthic drift between control and experimental channels when chloride levels were raised to 500 and 750 mg/L Cl^-, respectively.

b) In March 1975, chloride levels were gradually raised to 2,165 mg/L Cl^-. Benthic drift was observed in both streams (Figure 6-1). Benthic activity, relative to the control stream, began to increase between 850 and 1,200 mg/L Cl^-. Maximum drift (ca. a factor of 3) relative to the control stream was observed when chloride concentrations reached 2,165 mg/L Cl^-. As chloride levels declined, so did benthic drift. Nevertheless, benthic drift continued to remain high relative to the control stream.

c) The composition of drift was similar in control and experimental channels. Drift was comprised of several species of Plecoptera (stoneflies), Ephemeroptera (mayflies), Trichoptera (caddisflies), and Diperta (chironomids). This was unexpected since various laboratory studies have determined that freshwater invertebrates vary greatly in their salinity tolerances (see section 5 and the additional references cited by Crowther and Hynes (1977)). The researchers suggest that the invertebrates inhabiting Laurel Creek were equally sensitive to elevated chloride levels.

Figure 6-1: Chloride concentrations and drift in Lutteral Creek, Ontario (adapted from Crowther and Hynes 1977).



Crowther and Hynes (1977) concluded their study by noting that as chloride levels in the stream approach 1,000 mg/L Cl^-, deleterious effects, as described above, probably will begin to appear. During their study, they noted that most of these effects would be expected to occur during winter and early spring. However, they warned that as groundwater continued to become contaminated with chloride salts, stream flows might be contaminated by chloride-rich groundwater inflows during summer low flow periods.

4) Williams et al. (1997) detected significant variations in macroinvertebrate community structure related to different concentrations of chloride in 20 groundwater-fed springs in southeastern Ontario. The chloride content of these springs ranged from 8.1-1,149 mg/L Cl^-. Tipulidae and Ceratopogonidae (two chironomid families) were two taxa associated with springs containing higher chloride concentrations, whereas taxa such as, *Gammarus pseudolimnaeus* (an amphipod) and Turbellaria (a flatworm), were only found in springs with low chloride concentrations. The higher chloride concentrations found in some of the springs originated from groundwater that apparently had been contaminated by road salt.

5) Demers (1992) investigated the effects of elevated chloride levels on the aquatic macroinvertebrates inhabiting four streams near the town of Newcomb in the Adirondack region of northern New York. All four streams were located along a 2-km stretch of state highway 28N. Artificial substrates were placed in riffle or fast flowing sections of the stream, six upstream and six downstream of the highway. Flow rates ranged from 0.01-0.22 m^3/sec. Samplers were within 50-100 m of the road and were left in place between April 22 to June 3, 1988. At the end of the experiment, the samplers were recovered, the colonizing invertebrates removed and later identified. Chloride concentrations in the creeks were measured at weekly intervals during the course of the study. Highlights of the study results are as follows.

   a) The overall mean concentration of chloride in upstream locations was 0.61 mg/L compared to 5.23 mg/L in downstream areas. Chloride concentrations were as much as 66 times higher in downstream than upstream samples. The load to the streams and the duration of the new concentrations were not reported.

   b) Benthic diversity was lower in downstream than upstream sites. Chironomidae comprised an average of 83% of the individuals in the upstream sites versus 90% in downstream sites. In contrast, Perlodidae (a stonefly family) declined from 4.5 to 2.9% of the population and Ephemerellidae (a mayfly family) from 3.9% to <2% downstream of the highway. No Name Brook, which ran parallel to the highway for 257 m and had the lowest flow rate, was the stream most strongly affected by highway runoff. It is interesting to note that Kersey (1981, see below) noted an increase in chironomid and decrease in mayfly and stonefly dominance in the Humber River between November, when road salt had not yet been applied, and the following March, after road salt had been applied to an Ontario road.

6) Smith and Kaster (1983) investigated the effects of highway runoff on stream benthic invertebrates in Sugar Creek, Wisconsin (56 km southwest of Milwaukee). Four sites were examined: a control site, site 2 which received a slight amount of runoff from Highway 15, site 3 which received intermediate amounts of road runoff, and site 4 which received the greatest amount of runoff. Macroinvertebrates were sampled at monthly intervals from June 1980 to June 1981. Twenty-eight

physical and chemical parameters were monitored at these stations during a snowmelt event in 1980 as well as during two rainfall events in March and April 1981. Highlights of the study are as follows:

a) Peak concentrations of chloride (53 mg/L) and sodium (28 mg/L) were recorded during snowmelt events in March and April of 1981. No baseline concentrations were reported. Current speeds averaged (annual) 0.32 m^3/sec at site 1, 0.23 m^3/sec at site 2, 0.50 m^3/sec at site 3 and 0.64 m^3/sec at site 4.

b) Differences in mean annual benthic numbers at site 1 (4,155/m^2, 10,500 mg/m^2) and site 2 (2,611/m^2, 5,200 mg/m^2) were related to the slower current velocity and greater siltation at site 1. Site 4 had the greatest abundance and biomass (11,291/m^2, 62,200 mg/m^2) apparently because of the higher current velocity and better habitat (i.e., a larger coverage by cobble).

c) Site 3 had a mean annual numerical standing stock and biomass of 4,624/m^2 and 10,500 mg/m^2, values similar to site 1. However, pollution sensitive taxa were half as abundant (595/m^2) at this site as at site 1 (595/m^2). The largest differences were associated with Tricoptera (stonefly) and Coleoptera (beetle) larvae.

7) Molles and Gosz (1980) investigated the number and biomass of benthic invertebrates above and at 6 sites up to 2.2 km downstream of the Rio en Medio, a mountain stream in the Santa Fe, New Mexico ski area. Water quality parameters were measured weekly while benthic invertebrates were sampled in May and October 1977. Highlights of this study are as follows:

a) Chloride concentrations were low, averaging 0.38 mg/L Cl^- above the road and 8.61 mg/L Cl^- 200 m below the road, with concentrations declining to 5.63 mg/L Cl^- 2.2 km below the road. Higher concentrations downstream of the road were related to road salting. Chloride continued to be flushed from the soil during summer precipitation events.

b) Suspended sediment concentrations varied seasonally, reaching up to 160 mg/L Cl^- during episodic events during spring and summer. Sediments were readily eroded from the cobbly loam stream banks. During periods of high suspended sediment load, concentrations tended to be lower above the road than 200 m below the road.

c) Ephemeroptera, Coeloptera, Diptera, and Turbellaria occurred in lower numbers and biomass below the ski area than above, while Tricoptera and Nematoda occurred in higher numbers and biomass. Plecoptera and Oligochaeta occurred in similar numbers.

d) Decreases in invertebrate abundance and biomass along the stream length were related to suspended sediments rather than road salt impacts.

This study, like Smith and Kaster (1983), showed that many factors can affect differences in the invertebrates and other organisms along stream lengths. Natural factors, such as stream flow rate, substrate composition, and suspended sediments all have pronounced effects on stream communities. In contrast, chloride effects may be relatively small and, without a carefully designed experiment, difficult to quantify.

8) Maltby et al. (1995a) investigated the effects of motorway runoff on water quality, sediment quality, and the biota of three small streams in the M1 highway drainage basin in northern England. Studies were conducted over a 12-month period. Highlights are as follows:

   a) Median chloride values upstream of Pigeon Bridge Brook were 65.1 mg/L Cl^- versus 229.1 mg/L Cl^- downstream. Butterthwaite Ditch (86.2 mg/L Cl^- and 86.4 mg/L Cl^- respectively) and Rockley Dike (105.7 mg/L Cl^- and 112.1 mg/L Cl^-, respectively) showed smaller differences in upstream and downstream chloride concentrations.

   b) Calcium, magnesium, and copper occurred in statistically higher concentrations downstream than upstream of Pigeon Brook Bridge. Calcium occurred in higher concentrations and iron concentration were lower downstream from Butterthwaite Ditch. Zinc concentrations were significantly lower downstream than upstream of Rockley Dike.

   c) There was no evidence of differences in macroinvertebrate assemblages upstream and downstream of Butterthwaite Ditch and Rockley Dike. However, macroinvertebrates were reduced in abundance downstream of Pigeon Brook Bridge. Pollution-sensitive taxa (Plecoptera, Amphipoda, Trichoptera, and (Mollusca) declined in relative abundance while chironomid larvae and tubificid worms (Oligochaeta) became relatively more abundant. Moreover, there was a change in functional feeding groups (i.e., decrease in the relative abundance of scrapers and shredders and an increased abundance of collectors).

   d) Fungal assemblages exhibited small differences in diversity and composition between study sites. Diversity was higher downstream (8.52) than upstream (3.81) of Pigeon Bridge Brook. However, there were no differences in epilithitic algal assemblages. Leaf litter processing was lower downstream than upstream of Pigeon Bridge Brook, possibly because motorway runoff inhibited macroinvertebrate-meditated leaf decomposition at this site.

   e) Downstream impacts at Pigeon Bridge Brook were related to higher polynuclear aromatic hydrocarbon concentrations at this site, in particular, naphthalene, fluoranthene, and pyrene. Zinc, cadmium, chromium, and lead also were elevated at this site. Sediment size was not an important factor.

   f) The possible impacts of road salt were not investigated during this study. Since water samples were collected at three-month intervals between October 1990 and July 1991 (i.e., once in October, January, April, and July), the marked difference in median upstream and downstream chloride concentration at Pigeon Bridge Brook is suggestive of episoidic and massive chloride loading events.

6.3 Rivers

Only one study was found explicitly investigating the impact of road salt on rivers. Results are as follows:

Kersey (1981) investigated the impact of deicing road salts on benthic invertebrates inhabiting the Humber River, Ontario. Invertebrate community structure was investigated at three sites (Cedar Mills, side road; Cedar Mills, Highway 50; and Albion Hills) in June and November 1979, but only at Cedar Mills, Side Road, in March 1980. Samples were collected downstream of bridges in shallow, fast-flowing water. Chloride concentrations were measured in February and March 1980. Salt toxicity tests were performed on three species of caddisflies. Highlights of this study are as follows.

1) Chloride concentrations at Cedar Mills were 23.1 mg/L Cl^- upstream of the side road and 34.8 mg/L Cl^- downstream of the side road on February 21, 1980. On March 23, 1980, chloride concentrations upstream of the side road were 17.0 mg/L Cl^- versus 17.8 mg/L Cl^- downstream. Similarly, small differences in chloride levels upstream and downstream of Highway 50 and Albion Hills were noted. Low chloride concentrations were, in part, related to atypically low use of road salt and the fact that major snowmelt events were missed.

2) There were no obvious differences in benthic diversity between the three study sites in June and November. At Cedar Mills, side road, there were only small changes in species diversity between November 1979 and March 23, 1980. If the comparisons are based only on insect taxa, chironomids increased in percent composition (70.6% to 81.0%) while Tricoptera (25.5% to 15.8%) and Ephemeroptera (3.6% to 1.4%) decreased in percent composition between the two time periods. These differences were related to sample variability and to increased turbidity in March which reduced the ability to select suitable microhabitats for sampling.

3) Laboratory studies suggest that the larvae of the caddisflies, *Hydropsyche bronta*, *Hydropsyche betteni*, and *Hydropsyche slossonae*, can withstand exposure to 800 mg/L Cl^- for ten days without incurring significant mortality.

6.4 Wetlands

No studies were found on the effect of road salts on Canadian wetlands. Only one study was found investigating the effects of sodium chloride contamination by a road salt storage pile in a United States wetlands. This was the study conducted at Pinbook Bog, LaPorte County in Indiana (Wilcox 1982). This was a protected wetland which was included in the Indian Dunes National Lakeshore in 1966. However, in 1963, an uncovered sodium chloride storage depot was established overlooking the bog for use on the Indiana Toll Road I-80/90 (Wilcox 1982). Salt-laden runoff from the storage pile runoff resulted in major alterations in the bog vegetation within a 2 ha area, as did runoff from the highway. The salt pile was covered in 1972 and, after winter 1980-1981, road salt ceased to be stored at this site. Highlights of the study, which began in 1979 and continued until 1983, are as follows:

1) The total chloride inputs to the bog over the ten-year period when the salt storage pile was uncovered (1963-1972) were estimated as follows: 2.3 million kg from the salt pile, 0.4 million kg from road salting, and 0.012 million kg from direct precipitation.

2) Sodium concentrations as high as 468 mg/L and chloride concentrations as high as 1,215 mg/L were recorded in interstitial waters of the bog mat in areas of the strongest road salt impact (Wilcox 1982, 1984). These readings were made in 1979. High sodium and chloride concentrations may have occurred in earlier years.

3) Native species, such as *Sphagnum* spp. (bog moss) and *Larix laricina* (tamarack), were absent from the impacted areas of the bog and salt tolerant species, such as *Typha angustifolia* (cattail), invaded the bog. Salinity tolerances of various plant species were defined (Table 6-1 and Wilcox 1986) and many taxa were shown to be sensitive to sodium chloride concentrations as low as 280-400 mg/L Cl^-.

4) Chloride concentrations at control sites in 1980 and 1981 were 5-6 mg/L. This compares to a maximum single daily reading for salt-impacted locations of 1,468 mg/L Cl^- in 1979, 982 mg/L Cl^- in 1980 and 570 mg/L CL^- in 1981. The maximum chloride concentration in 1983 was 610 mg/L.

5) As salt concentrations decreased some 50% over 1980-1983, many endemic bog plants, including *Sphagnum* recolonized the bog (Wilcox 1986). *Sphagnum* began growing on low hummocks in areas where interstitial chloride concentrations had dropped approximately 300 mg/L.

Table 6-1: Sodium chloride tolerance, mean cover, and frequency (in parentheses) of selected plant species in the salt-impacted mat zone of Pinhook Bog, Indiana. A= occasional, B = frequent. Source: Wilcox (1986).

Scientific Name	Common Name	NaCl Tolerance (mg/L)	1980	1981	1982	1983
Bidens connata	purple-stemmed tickseed	1,030	A (22)	<1 (33)	<1 (33)	<1 (22)
Pyrus floribunda	purple chokeberry	1,070	B (78)	10 (78)	13 (78)	15 (89)
Hypericum virginicum	marsh St. John's wort	1,070	A (44)	1 (67)	1 (56)	1 (56)
Sphagnum	bog moss	770	A (22)	4 (67)	5 (67)	5 (89)
Solidago graminifolia	grass-leaved goldenrod	760	A (44)	3 (56)	3 (56)	1 (56)
Vaccinium corymbosum	highbush blueberry	580	A (11)	5 (56)	10 (56)	12 (56)
Vaccinium atrocuccum	black highbush blueberry	400	-	-	1 (44)	1 (44)
Drosera intermedia	oblong leafed sundew	360	A (33)	2 (44)	5 (44)	5 (44)
Nemopanthus mucronata	mountain holly	280	A (11)	<1 (22)	<1 (22)	<1 (22)
Larix larcina	tamarack	280	A (22)	3 (33)	3 (33)	4 (33)

6.5 Lakes

Only five studies were found which investigated the impact of road salt on the biological communities of lakes; the impacts of road salt on meromixis formation have been discussed earlier. Two recent studies have investigated paleolimnological changes in algal populations with changing pH and increasing phosphorus concentrations and chloride concentrations. Results of the lake studies are as follows:

1) Benthic invertebrate diversity decreased with the anoxic bottom conditions which developed in Lake Wabekayne, a storm-water impoundment in Mississauga, Ontario (Free and Mulamoottil 1983). Chloride concentrations were elevated from approximately February to April (e.g., 282 mg/L Cl^- in February, 1979), particularly on the bottom of the lake. Concentrations in August 1979 were much lower at 50 mg/L Cl^-.

2) Tuchman et al. (1984), used core samples from lake sediments, to investigate the historic salinization of Fonda Lake, Michigan. A salt storage facility has been located adjacent to Fonda Lake since

1953. A reduction in algal species diversity began in 1960; this reduction was thought to be related to increased salinity resulting from the slow leaching of salts from the storage facility. Diatom diversity reached a minimum in 1968, when a variety of salt-tolerant (or halophilic) taxa (e.g., *Diatoma tenue*, *Navigula gregaria* and *Synedra fasciculata*) attained their highest relative abundance. In later years, diversity increased slightly and some halophilic taxa decreased in relative abundance, suggesting a decrease in salt loading to the lake. A possible reason for the reduction in salt loading may have been the construction of an asphalt pad for the salt storage facility in the early 1970s. Aside from the changes in chloride concentration in Fonda Lake, the results of this study also indicate that subdominant diatom species may be more sensitive to salinity changes than dominant, euryhaline taxa. Unfortunately, lake salinity was not determined as part of this study.

3) Zeeb and Smol (1991) continued this study, investigating scaled chrysophytes. *Mallamonas caudata*, a chloride indifferent taxon, dominated at all depths in the core. *Mallamonas elongata*, a widely distributed taxon found more commonly in eutrophic and alkaline waters, and *Mallamonas pseudocoronta*, a chloride intolerant taxon, declined during the period when chloride concentrations apparently were highest. *Mallamonas tonsurrata*, which has a world wide distribution and occurs mainly in eutrophic lakes, became more abundant during this period. Overall, the response of the chrysophyte community was similar to that observed by Tuchman et al. (1984) using diatoms. It is worthwhile to note that major shifts in diatom and chrysophyte assemblages were associated with relatively small changes in salinity (i.e., from ca. 12-235 mg/L Cl^- and ca. 20-387 mg/L NaCl).

4) Smol et al. (1983) investigated the impact of cultural disturbances on Round Lake, a small meromictic lake in central Ontario. Prior to European settlement, the lake was oligotrophic and dominated by the diatom, *Cyclotella* spp., and the chrysophyte, *Mallamonas pseudocoronata*. With land clearing and settlement, the lake became more productive with the diatom, *Synedra* spp., and the chrysophyte, *Mallamonas tonsurata*, becoming more dominant. The lake then became eutrophic with the diatom, *Stephanodiscus hantzschii*, dominating and *Syndera* spp. being reduced to trace levels. Increased productivity was related to urbanization, including nutrient inputs from septic tanks. This eutrophic period was followed by a return to oligomesotrophic conditions indicated by the recurrence of *Cyclotella* and *Mallamonas*, most notably *M. fastigata*. This was, in turn, related to meromictic conditions that apparently had developed in the lake as a result of road runoff and seepage from a salt storage shed. Salt concentrations in the monimolimnion were 58.4 mg/L Na^+ and 103.7 mg/L Cl^-. Stratification, by preventing spring and autumn overturn of the water column, prevented the vertical exchange of nutrients regenerated at the sediment-water interface with the upper, well-lit layer of the water column where most algal growth would have occurred. As with the study on Fonda Lake, it is worthwhile to note that major shifts in algal assemblages and productivity were associated with relatively small changes in salinity.

5) Bridgeman et al. (2000) investigated recent changes in the limnology of Third Sister Lake, Michigan, as a result of increased chloride loadings. This increased loading was associated with residential and commercial growth in the lake watershed. Chloride concentrations increased from 19 mg/L in 1981 to 260 mg/L in 1988 and bottom waters became anoxic because of reduced spring mixing. This

reduced mixing and concurrent nutrient exchanges may have affected a reduction in primary productivity rates. More striking was the sharp decrease in benthic species diversity from at least 12 species in 1927 to 4-5 species in 1999 and average densities from 167,000/m^2 to 15,144/m^2. Moreover, the region of densest benthic population abundance shifted from deep waters (16-18 m) to shallower depths.

6) Dixit et al. (1999) investigated long-term water quality changes in 257 lakes in northeastern United States using a paleolimnological approach. This approach was based on earlier work which assessed long-term changes in lake acidity (Dixit et al. 1992). Using current diatom assemblages in the 257 lakes, these researchers were able to estimate the total phosphorus, chloride, and pH optima for a wide variety of diatom species. They then used this information to infer changes in the water quality of the study lakes base on changes in the sedimentary diatom record. Marked deterioration was reported in the water quality in these lakes over the past 150 years. The nature of the change varied with the ecoregion. Chloride and phosphorus concentrations have increased, with the greatest increases in lakes that now have the higher chloride and phosphorus concentrations. Diatom communities displayed difference responses to increased chloride concentrations than to increased phosphorus concentrations. The greatest changes have been in the proportion of lakes with chloride concentrations of 100-200 uequiv./L (3.6-7.1 mg/L) with a smaller increase in the proportion of lakes with chloride concentrations of >200 uequiv./L (>7.1 mg/L). Sodium and chloride concentrations were highly correlated in these lakes, suggesting that increased chloride concentrations were associated with road salting. However, it was also recognized that agriculture, silviculture, and urbanization can affect increases in chloride concentration. While these increases were modest, they were sufficient to affect algal community composition. In an ongoing study, Dr. Peter Dillon with the Ontario Ministry of the Environment (e-mail to M. Evans dated December 12, 2000) has noted that 20-25% of lakes in the Muskoka region of Ontario have elevated sodium and chloride concentrations. These increased sodium and chloride levels appear to be related to road salt. Other counties have shown fewer affected lakes.

7) In a more recent study, Dixit et al. (2000) investigated water quality changes in three lakes in northeastern United States. Increased chloride concentrations were related to road salt use although other watershed disturbances were recognised as also having some potential importance. Chloride concentrations ranged from 6.9 – 27.4 mg/L with the highest concentration in the lake with the urban development.

6.6 General Conclusions

Few studies have been conducted investigating the impacts of road salts on aquatic ecosystems in Canada. The studies that have been conducted indicate that high concentrations of salt (ca. 1,000 gm/L NaCl) can affect algal and bacteria densities, macrophyte and benthic composition. At lower concentrations (ca. 250 mg/L), increased chloride concentrations can cause small lakes to become meromictic with pronounced effects on benthic populations, algal standing stocks and composition, and primary production levels. Moreover, even very small increases in chloride concentration (2- 10 mg/L) can affect changes in phytoplankton community structure in low salinity lakes.

The literature review was broadened to the saline literature on lakes and rivers in order to determine whether additional insights could be found on the potential impacts of road salt on aquatic ecosystems. As previously noted, there are many naturally saline ecosystems in various arid regions of the world. In many such areas, rivers are becoming more saline as a result of water diversions. Natural saline seepage may occur in certain bedrock or are associated with various hydrocarbon operations. Finally, there is emerging literature that describes investigations of the variables affecting standing stocks and composition of organisms across a variety of geological and other settings. This literature was examined to determine whether or not chloride or conductivity was a significant environmental variable and whether the authors identified road salt as an important anthropogenic impact. This review is brief but should form the basis for a more in depth review for interested parties.

6.7 Species Diversity as a Function of Salinity

Studies conducted in fresh and salt waters have shown a gradual loss in the number of fresh water taxa as salinity increases (Figure 6-2 and Table 6-2). Similarly, there is a decrease in the number of marine species as salinity decreases (Wetzel 1983). For freshwater species, the greatest decrease in species numbers appears to occur between ca. 2 and 6 g/L. Similarly, saline lakes generally contain a lower number of species compared to fresh water lakes (Wetzel 1983; Goldman and Horne 1983). In general, as salinity increases, the number of species decreases. Blue-green algae and a few zooplankton species (e.g., *Artemia* spp. or brine shrimp) are often found in saline waters (Goldman and Horne 1983). Fish, however, are typically absent from meso- and hypersaline lakes due to osmotic stress. Even though the number of species may be lower in saline lakes, productivity of these lakes is often comparable to or may even exceed that of freshwater lakes (Wetzel 1983; Goldman and Horne 1983).

Figure 6-2: Species diversity across a salinity and chloride gradient (adapted from Wetzel 1983).

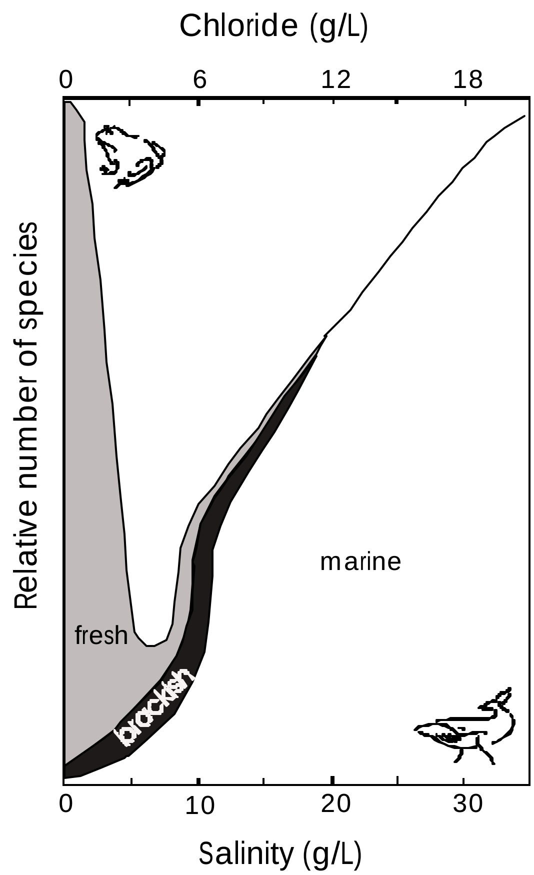


Table 6-2: Numbers of Animal Species Recorded Within Various Salinity Ranges for the Saline Lakes in Victoria, Australia. Source: Williams 1978 (in Goldman and Horne 1983)

Salinity (mg/L)	Number of Species
1.000 - 10.000	71
10.000 - 100.000	36
100.000 - 200.000	8
200.000 - 300.000	1
> 300.000	0

A brief review of the saline lake literature was conducted in an effort to find more detailed information on the loss of species with increasing salinity, particularly in the 250–5,000 mg/L NaCl range. This generally did not provide the detailed data desired. Highlights of these "low-salinity" studies are as follows:

1) Cumming and Smol (1993) investigated diatom assemblages in a wide variety of saline lakes in British Columbia. Salinity optima were calculated for various diatom taxa and they noted that the majority had salinity optima of <3,000 mg/L. Species diversity was weakly, but significantly correlated with salinity. Fritz et al. (1993) also calculated salinity optima for various diatom species in the northern Great Plains lakes with similar results. Both publications provide data on salinity ranges and optima for a number of diatom species.

2) Blinn et al. (1981) investigated the seasonal dynamics of phytoplankton at three locations along the Chevelon Creek system in Arizona. Chloride concentrations at site 1 were 88 mg/L at baseflow compared to 900-1,100 mg/L Cl^- at site 3, located 10.8 km downstream of site 1 and 3.5 km downstream of site 2 (chloride concentration at site 2 was not given). The elevated chloride at site 3 was due to numerous seeps and springs from the canyon wall. There were marked differences in phytoplankton between the three sites with species associated with particular salinities.

3) Short et al. (1991) investigated benthic invertebrates in a Kentucky stream subject to chloride seepage from nearby oil field operations. Ephemeropterans were the group least tolerant of elevated sodium chloride levels and were absent in regions where salinity exceeded 2,000 mg/L Cl^-. This absence was related to direct salt toxicity. Fish appeared to be more tolerant of these elevated salinities.

4) Colburn (1988) noted that species diversity in saline lakes in Death Valley decreased with increasing salinity, but indicated that salinity alone could not account for these observations. Some taxa had broader distributions than predicted based on experimental studies testing salinity and temperature tolerances. These broader distributions were related to decreased competition and predation that occurred at the high salinities. Amphibians, Ephemeroptera, amphipods, and cladocerans were generally absent at salinities of 10,000-20,000 mg/L. Odonata and Tricoptera also were absent at the higher salinities.

5) Galat et al. (1988) investigated the salinity tolerances of three benthic invertebrates from Pyramid Lake, Nevada. Multiple generation toxicity tests using mesocosms indicated a much higher sensitivity to elevated salts than simple short-term bioassays. Shifts in feeding habits and reduced predation as a result of the loss of predatory fish species were major factors affecting these responses. Melack (1988) also found complex trophic responses to salinity stresses in a shallow, equatorial lake in Kenya, Africa.

6) Hart et al. (1990) investigated the implications of increased salinification of Australia's aquatic ecosystems. They concluded that adverse effects are most likely to begin occurring around 1,000 mg/L salinity for macrophytes, algae, and macroinvertebrates. They also suggested that the

microbial community would not be directly affected because of their rapid generation times and ability to adapt.

7) Williams (1987) suggested that the increased salinification of irrigation waters would begin to have adverse impacts on the crop yield of sensitive crops at 500-1900 mg/L and many crops at 1000-2000 mg/L. He also reported that, in Sunday River, South Africa, salt intolerant diatoms began to be replaced by more tolerant species in salinity ranges of 500-2000 mg/L.

8) Metlzeling (1993) reported little change in macroinvertebrate structure in Australian systems of differing salinities in the 51-1,100 mg/L range, although changes became evident at higher salinities. Additional detailed discussions of the effects of elevated salinities in Australian aquatic ecosystems can be found in Williams et al. (1990 and 1991).

9) Leland and Fend (1998) investigated benthic distributions in the San Joaquin River Valley, California. Total dissolved solids concentrations ranged from 55-1,700 mg/L; distinct assemblages were observed at the high and low salinities. Ephemeroptera rarely occurred at salinities above 1,000 mg/L.

10) Rowe and Dunson (1993) conducted a study investigating breeding effort and various abiotic characteristics of 35 temporary wetlands in central Pennsylvania. The number of egg masses of the spotted salamander, *Ambystoma maculatum*, was negatively correlated with higher concentrations of total cations (Na^+, K^+, Mg^{2+}, Ca^{2+}) and positively correlated with pH and pond volume. These wetlands were not affected by road salt. Sodium concentrations ranged from 0.34-0.82 mg/L and pH from 4.2-6.2.

6.8 Conductivity and Freshwater Lakes

A number of studies have shown that conductivity (or total dissolved solids) can be a good predictor of fish (Ryder et al. 1974; Matuszek 1978; Hanson and Leggett 1982) and possibly phytoplankton (Oglesby 1977) biomass. In Great Slave Lake, Rawson (1953, 1956) related higher standing stocks of plankton and benthos in Christie Bay than McLeod Bay to the greater mineral concentrations in the waters of the former bay. Mineral content also appeared to be important in affecting regional differences in plankton, benthos, and fish standing stocks in lakes in northern Saskatchewan (Rawson 1960). In general, these regression-based studies determined that biomass increases with conductivity, although the causal mechanism was not determined. While all these systems were carbonate-dominated, it is unlikely that carbonates per se affected productivity. It is more probable that that conductivity integrates considerations of nutrient concentration, depth, and lake size or depth. More recent studies have shed some light on these issues.

Chow-Fraser (1991) determined that total nitrogen and phosphorus concentrations in a series of 161 Canadian lakes were strongly correlated with total dissolved solids (TDS) concentrations. In addition, the morphoedaphic index (MEI), based on dividing TDS by mean lake depth, explained 85% of the variation in TN and TP concentrations and 50% of the variation in chlorophyll a. However, the MEI explained none of the variation in zooplankton biomass. Similarly, Downing et al. (1990) determined that fish production was more strongly correlated with annual phytoplankton production, mean total phosphorus concentration, and fish standing stock than to the MEI.

More recent limnological studies have begun to investigate community structure in a wide variety of lakes and as a function of a number of limnological variables, such as conductivity, nutrient concentrations, and major ions. This literature was briefly examined to determine whether additional information could be found on the responses of aquatic organisms to small gradients in salinity. Since these studies were conducted in fresh water lakes, conductivity was determined primarily by calcium carbonates. The following are noted:

1) Moser et al. (1998) investigated the physical and chemical limnology of 35 lakes in Wood Buffalo National Park, Alberta. This study illustrated the relationship between conductivity, major ions, nutrient concentrations, chlorophyll, and other limnological measures in a series of pristine lakes located on and off the Canadian Shield. Shield lakes had lower conductivity and dissolved nutrient concentrations than lakes located on limestone and gypsum. Productivity and algal standing stocks also were lower in Shield lakes. However, phosphorus and nitrogen concentrations were not directly related to geology although conductivity and major ion concentrations were strongly related to geological factors.

2) Chetelat et al. (1999) investigated periphyton biomass and community composition in 13 rivers in southern Ontario that differed in their nutrient concentrations and conductivity. Biomass was strongly correlated with total phosphorus ($r^2 = 0.56$) and even more strongly correlated with conductivity ($r^2 = 0.71$). They suggested that the predominance of *Cladophora* at high conductivity sites was related to the higher calcium concentrations at this site. Conductivity ranged from 65-190 µS/cm. Major differences in species composition were observed between low and high conductivity sites.

3) Rott et al. (1998) investigated periphyton assemblages in the Grand River, Ontario. Chloride concentrations ranged from 7.7-85.0 mg/L and conductivity from 180-540 µS/cm. Canonical correspondence analysis showed that the largest portion of variability in species composition in the river over the study period could be explained by a seasonal gradient related to temperature and latitudinal gradients of nitrite-nitrate, conductivity, and chloride. Phosphorus, ammonia, pH, turbidity, and oxygen were of lesser importance.

4) Stemberger and Lazorchak (1994) investigated zooplankton assemblage responses to disturbance gradients in 19 New England lakes. Lakes were classified as warm water and cold water lakes. Chloride concentrations varied over a relatively small range, from 0.3-60.5 mg/L (8-1,707 µeq/L). Chloride was not used as a variable in the multivariate analyses, but the percentage of disturbed lake shoreline was used in these analyses. This variable, along with total phosphorus, chlorophyll concentration, and variables related to fish species were important is explaining the variation in zooplankton species assemblages.

5) As previously noted, Dixit et al. (1999) assessed changes in water quality in the northeastern United States using diatoms that had become deposited in lake sediments as indicators of change. The composition of diatoms in the surface sediments was related to current water quality parameters including pH, chloride, and total phosphorus (TP) concentrations. Optima were developed for 235 of the common species (Table 6-3). It is apparent from this study that even small changes in chloride concentration may affect subtle changes in algal community structure. This must occur for other organisms such as other algal groups, plants, benthic invertebrates, and zooplankton.

Table 6-3: The pH, total phosphorus and chloride optima for selected diatom species in the northeastern United States. Source: Dixit et al. (1999).

Taxon	pH	TP (µg/L)	Cl^- (mg/L)
Achnanthes altaica	6.8	7	0.7
Achnanthes clevei	8.1	7	8.7
Amphora ovalis	8.0	22	4.2
Amphora perpusilla	8.3	25	21.0
Cyclotella meneghiniana	8.3	66	39.5
Cymbella cesatii	7.8	10	0.7
Fragilaria crotonensis	8.0	14	6.9
Navicula bremensis	6.2	7	0.5
Nitzschia linearis	7.4	8	0.8
Stephanodiscus niagrae	8.1	16	10.5
Synedra ulna	7.9	15	4.5
Tabellaria fenestrata	7.5	13	1.8
Tabellaria quadriseptata	5.5	11	1.1

It is difficult to draw conclusions from most of these studies because they are too few in numbers to be comprehensive and because they were not designed to investigate the implications of road salt in the environment. Nevertheless, the limited data suggests that small increases in chloride concentration can result in subtle changes in community composition and biomass. Whether these increases are directly due to sodium and/or chloride or are simply a surrogate measure for increased nutrient loading from land disturbance could not be determined in this literature review. However, the recent research of Dixit et al. (1999) is highly suggestive that road salt has a major role in changes in diatom assemblages in northeastern United States. This, combined with other studies, such as Crowther and Hynes (1977), Dickman and Gochnauer (1978), Smol (1983) Wilcox (1986) and Zeeb and Smol (1991), suggest that increases in chloride from preindustrial baseline levels of 10 mg/L or less to concentrations of 200-1,000 mg/L will have measurable impacts on aquatic ecosystems.

Finally, it is important to note that road salt contains various impurities including phosphorus (26 mg/kg) and nitrogen (7-4,200 mg/kg) (Michigan Department of Transportation 1993). Thus, a 1,000 mg/L increase in sodium chloride concentration (or 606 mg/L chloride) could result in a 26 ug/L increase in phosphorus concentration and 7-4,200 ug/L in nitrogen concentration. A 250 mg/L increase in chloride concentration could result in a 10.7 ug/L increase in phosphorus concentration and 2.9 – 1,731 ug/L in nitrogen concentration. These increases would be sufficient to cause an oligotrophic aquatic ecosystem to become mesotrophic or even eutrophic (Wetzel 1983).

7.0 RISK CHARACTERIZATION

7.1 Introduction

The objective of this chapter is to determine the likelihood and magnitude of adverse effects occurring in aquatic ecosystems as a direct result of runoff from road salt applications and from salt storage depots. This approach is based on the preceding literature on the impacts of road salt on chloride levels in the environment, laboratory studies assessing toxicity to chloride salts, and field studies investigating ecological impacts as a result of road salt applications. This chapter concerns itself only with toxicity resulting from chloride in road salt. Additives such as ferric cyanide compounds are dealt with by Letts (2000 a, b). The approach for assessing toxicity follows that developed for priority substances under CEPA (Environment Canada 1997c). In essence, it is tiered approach, involving three levels or tiers.

7.2 Tier 1 Assessment

A Tier 1 assessment is based on a hyperconservative approach for assessing CEPA toxicity for a wide range of trophic levels and toxicity responses. Mortality is one such response. Other potentially useful responses are growth, reproduction, fecundity, longevity, diversity, productivity, community structure, and diversity. Modifying factors such as temperature, water hardness, etc., need to be considered in these assessments. Wherever possible, assessments are based on data pertaining to Canadian species and conditions.

Body burden, while a useful endpoint for some studies, is not suitable for road salt assessments. Chlorides, unlike lipophilic organic contaminants such as PCBs and DDT, are not strongly bioaccumulated and biomagnified by organisms.

Tier 1 assessments also require estimates of the maximum concentration of the chemical in the environment as a result of the anthropogenic release of that compound. Tier 1 is a hyperconservative estimate of risk or CEPA toxicity. It involves calculating a quotient for each assessment endpoint by dividing a single EEV by an ENEV where:

EEV = estimated exposure value

ENEV = estimated no effects value.

For Tier 1, the EEV is the maximum measured concentration in the Canadian Environment. The ENEV is estimated by dividing a Critical Toxicity Value (CTV) by an Application Factor (AF). Recommended AF’s are provided in Table 7-1 of the CEPA Guidance Manual Version 1.0 (Environment Canada 1997c) and are described below. If the resulting quotient is >1, the substance is viewed as being potentially CEPA “toxic” and the risk characterization proceeds to Tier 2. If the quotients are <1, adverse effects are viewed as being unlikely and the substance is not considered to be CEPA “toxic”.

7.2.1 Critical Toxicity Values and Application Factors

The CEPA Guidance Manual Version 1.0 (Environment Canada 1997c) recommends that, under ideal situations, toxicity values are derived from low-toxicity, chronic tests. That is, from tests which extend over the full life cycle or a major portion of the organism's life cycle rather than acute tests in which exposure is only for a short duration of the organism's life cycle. For bacteria, test duration could be as short as one day; for zooplankton test duration could be as short as three or four weeks. Benthic organisms can live for one year or more, amphibians and forage fish such as minnows for a few years, while larger fish can live for twenty or more years. The Guidance Manual also recommends that toxicity tests should involve five or more treatments, specify a statistical model, and estimate parameters through regression analysis. Examples of test endpoints are:

LC_{25} = lethal concentrations

EC_{25} = effective concentrations, but not lethal (e.g., immobilization)

IC_{25} = inhibiting concentration (e.g., reduced biomass)

A maximum AF of 10 is recommended when such LC_{25}, EC_{25} or IC_{25} tests are available and where the base data set is large containing a number of taxa (e.g., fish, daphnid, and algal species). A maximum AF of 100 is recommended for data sets based on the lowest acute LC_{50} or EC_{50} from a database comprising taxa such as fish, daphnids, and algae species. A maximum AF of 1,000 is to be used for the lowest acute LC_{50} or EC_{50} from a data set of only one or two species.

If LC_{25}, LC_{50}, EC_{25}, and EC_{50} data is not available, a less preferred approach for estimating toxicity is the LOEL (lowest observed effects level) or NOEL (no observed effects level), but such an approach has several shortcomings including:

1) NOELs and LOELs are test concentrations that do not correspond to specified effects levels from one test to the next;

2) Poor experimental design could mistakenly indicate that a substance is less toxic than it really is;

3) Most information from the toxicity testing is not used.

7.2.2 Approach to Toxicity Data

Tier 1 evaluations in this report are based on laboratory studies. While a number of field studies were located which demonstrated that diversity, standing stocks, behaviour, and survivorship were impacted by salinity, these studies were not as quantitative as laboratory investigations of toxicity. Therefore, only laboratory data is used in the Tier 1 assessments. Assessments are performed for all major components of aquatic communities (e.g., phytoplankton, zooplankton, benthos, amphibians, and fish) for which appropriate toxicity data was located. For example, no studies were found testing the tolerances of protozoans, reptiles, and birds to sodium, potassium, magnesium, or calcium chloride.

7.2.3 Maximum Estimated Concentration Relevant to the Canadian Environment (EEV)

In this literature review, the highest reported sodium chloride concentrations in the aquatic environment were associated with highway runoff, road salt-contaminated snow, and with contaminated waters near salt storage depots. These studies provide for the range of estimates of the maximum estimated chloride concentration relevant to the Canadian Environment.

In Maine, chloride concentrations in surface waters of four wetlands located within 30 m of salt storage depots reached concentrations of 13,500 mg/L: concentrations remained elevated throughout the March-October sampling period (Ohno 1990). Evaporation may have played a major role in these elevated chloride concentrations. Arp (2001) reported that chloride concentrations in seepage from a New Brunswick salt depot reached 10,000 mg/L.

In Ontario, chloride concentrations as high as 19,135 mg/L have been reported in highway runoff from the Skyway Bridge (Table 4.5; Mayer et al 1998); concentrations of 10,200 mg/L and 10,960 mg/L have also been observed at this bridge. Delisle and André (1995) reported that chloride in snow carried by Montreal trucks occurred in an average concentration of 3,851 mg/L with a maximum reported concentration of 10,000 mg/L. Mayer et al. (1999) reported chloride concentrations as high as 89,000 mg/L in meltwater from some city streets.

For the purposes of Tier 1 assessments, an EEV of 10,000 mg/L Cl^- is used to represent estimated maximum chloride concentrations in snowmelt draining from highways into small water bodies, such as ditches and creeks. Such a concentration also approximates that observed near road salt storage depots.

7.2.4 Salts Other Than Sodium Chloride

Toxicity data was also located for calcium, magnesium, and potassium chloride salts. Tier 1 Risk Quotients (i.e., EEV/ENEV) are calculated for these salts although they are not commonly used for road deicing. Calcium chloride is, however, used as a dust suppressant and is occasionally mixed with sodium chloride. As previously noted, laboratory studies suggest that potassium chloride in the most toxic of the four salts considered, followed by magnesium. The same EEV value is used for these chloride salts as sodium chloride (i.e., 10,000 mg/L Cl^-)

7.2.5 Bacteria

Only one laboratory study was located for bacteria. Ito et al. (1977) reported that 50% of the RNA of the bacteria, *Escherichia coli*, was degraded when exposed for 2 hours to an 8,767 mg/L NaCl solution (5,321 mg/L Cl^-). Such a response could impact bacterial numbers, growth rates, and important enzyme functions. Because *E. coli*, is not a representative aquatic bacteria, being more commonly found in the intestinal tracks of humans and other mammals, it is not being used for the Tier 1 and other tier assessments.

7.2.6 Fungi

Only one experiment testing the toxicity of sodium chloride to fungi was located. Sridhar and Barlocher (1997) report that there was an increase in the sporulation of aquatic fungi at a salinity of 659 mg/L NaCl or 400 mg/L Cl^- when compared to a control population. The experiment was run over 48 hours. The increase in the rate of sporulation was not reported. An application factor of 100 is used because no information was provided on the test species, the short duration of the test and because the data is semi-quantitative.

For sodium chloride, the Tier 1 Risk Quotient for fungi is 10,000 mg/L Cl^- divided by 40 mg/L Cl^- or 2,500. This quotient is greater than 1 and so a Tier 2 assessment should be conducted for fungi.

Sridhar and Barlocher (1997) reported an increase in the sporulation of aquatic fungi when exposed to a 554 mg/L calcium chloride solution (354 mg/L Cl^-) over a 48 hour exposure time. An application factor of 100 also is being applied to this data set. For calcium chloride, the Tier 1 Risk Quotient is 10,000 mg/L Cl^-/35.4 mg/L Cl^- or 2,825, a value that is not appreciably different from that for sodium chloride.

Rantamaki et al. (1992) reported that the fungus *Aphanamyces astaci* did not produce spores when exposed to 1,904 mg/L magnesium chloride (1,418 mg/L Cl^-) over 120 hours. An application factor of 100 is used because the data is semi-quantitative (i.e., data is not reported as EC's).

For magnesium chloride, the ENEV is 14.18 mg/L Cl^- and the Tier 1 Risk Quotient is 77.5.

7.2.7 Protozoans

Three experiments were located which investigated the sodium chloride toxicity to protozoans. One experiment (Fuji and Hellebust 1994) was based on the culture of a salt tolerant species, *Bvoekelovia hooglandi*. The second experiment tested *Euglina gracilis*, a photosynthetic flagellate which experienced a 16% reduction in cell number when cultured in the light and a 38% reduction in cells when cultured in the dark at a salinity of 5,845 mg/L NaCl or 3,548 mg/L Cl^- (Gonzalez-Moreno et al. 1997). The test was run for seven days. The most sensitive species was *Paramecium tetrourelia* which experienced a 17% reduction in cell division when exposed to 577 mg/L NaCl (350 mg/L Cl^-) for five days (Cronkite et al. 1985). No application factor is being used because the study is based on an EC_{17}.

Using *Paramecium tertourelia* as a "representative" protozoan, the EEV/ENEV chloride quotient for protozoans is 10,000 mg/L Cl^- divided by 350 mg/L Cl^- or 28.6. This quotient is above 1 and so a Tier 2 assessment should be conducted for protozoans.

Table 7-1: Summary of Tier 1 Calculations. CTV is the Critical Toxicity Value, EEV is the maximum estimated concentration (10,000 mg/L) of chloride in the environment (from highway runoff) and EEV, is the Estimated No effects value (CTV/AF). See text for further explanation.

Endpoint Organism	Exposure Time/Endpoint	CTV NaCl (and Cl^-) (mg/L)	Application Factor	EEV/ENEV	Toxicity Source
1. Fungi					
Unknown aquatic fungi	increased sporulation, 48	659 (400)	100	2,500	Sridhar and Barlocher
2. Protozoans					
Paramecium tetrourelia	17% reduction of cells cultured in light, 57-day test	577 (350)	1	28.6	Cronkite et al. 1985
3. Phytoplankton					
Nitzschia linearis	50% reduction in number of cells, 120 hours	2,430 (1,475)	100	678	Patrick et al. 1968
4. Macrophytes					
Sphagnum fimbriatum	43% reduction in growth, 45	2,471 (1,500)	100	667	Wilcox 1984
5. Zooplankton					
Ceriodaphnia dubia	50% mortality, 7 days	2,019 (1,225)	100	816	Cowgill and Milazzo 1990
6. Benthic invertebrates					
Nais variabilis	LC_{25} 48 hours	2,000 (1,214)	10	82.4	Hamilton et al. 1975
7. Amphibians					
Xenopus leavis	EC_{50} 7 days	2,510 (1,524)	100	656	Beak 1999
8. Fish					
Oncorhynchus mykiss	EC_{25} 7 days, egg/embryo	1,630 (989)	10	101	Beak 1999

7.2.8 Phytoplankton

Only two studies were located testing phytoplankton to sodium chloride responses. The more sensitive organism was the diatom, *Nitzschia linearis*. Patrick et al. (1968) reported a 50% reduction in the number of cells of this diatom when exposed to 2,430 mg/L sodium chloride (1,475 mg/L Cl^-) solution for 120 hours. An application factor of 100 is used because this test is based on an EC_{50} and in keeping with the hyperconservative approach of Tier 1. This gives an ENEV of 14.8 mg/L chloride and an EEV/ENEV of 678. This quotient is above 1 and so a Tier 2 assessment should be conducted for phytoplankton.

Patrick et al. (1968) also reported a 50% reduction in *N. linearis* cell numbers when exposed to 3,130 mg/L calcium chloride (2,000 mg/L Cl^-) and to 1,337 mg/L potassium chloride (636 mg/L Cl^{-1}). Using an application factor of 100 gives a Tier 1 Risk Quotient of 500 for chloride as calcium chloride and 1,572 for chloride as potassium chloride.

7.2.9 Macrophytes

Only two quantitative studies were located which investigated the response of macrophytes to road salt. Stanley (1974) determined that the European millfoil, *Myriophyllum spicatum*, experienced a 50% reduction in dry weight following a 32 day exposure to 2,196-8,178 mg/L NaCl (1,333-4,964 mg/L Cl^{-1}). Wilcox (1984) reported that *Sphagnum fimbriatum* (bog moss) experienced a 99% reduction in mean length when grown in bog water with a chloride concentration of 5,000 mg/L for 75 days. *S. recurvum* experienced a 43% reduction in mean length when grown in bog water with a chloride concentration of 1,500 mg/L for 45 days. The *S. recurvum* test results will be used in the Tier 1 assessment. An application factor of 100 is used because this test is based on an EC_{43} and in keeping with the hyperconservative approach of Tier 1. This gives an ENEV of 15 mg/L Cl^- and a Tier 1 Risk Quotient of 667. This quotient is above 1 and so a Tier 2 assessment should be conducted for macrophytes.

No studies were located investigating the tolerances of macrophytes to potassium, magnesium, and calcium chloride.

7.2.10 Zooplankton

A number of older studies were located investigating the toxicity of sodium chloride to the cladoceran, *Daphnia magna*. One older study was located investigating the copepod *Cyclops serrulatus*. Most of these studies were of limited value with the exposure time unknown or, in one instance, involved testing in distilled water. Several other studies reported threshold toxicity as occurring between 3,170-5,093 mg/L sodium chloride (cited in McKee and Wolf 1963). This data also lacked precision in the response factor (i.e. data was reported as thresholds).

Arambasic et al. (1995) reported an LC_{50} for *D. magna* when exposed to a 4,746 mg/L sodium chloride (2,881 mg/L Cl^-) for 48 hours. Anderson et al. (1948) reported that 50% of the *D. magna* were immobilized when exposed to 3,680 mg/L NaCl for 64 hours in Lake Erie water at 25 ^oC.

Cowgill and Milazzo (1990) reported a LC_{50} for *Ceriodaphnia dubia* of 2,019 mg/L NaCl (1,225 mg/L Cl^-). The test was conducted over 7 days. Birge et al. (1985) reported a LC_{50} of 1,470 mg/L chloride for the same species over 2 days. The Cowgill and Milazzo (1990) 7-day experimental results are used. An application factor of 100 is selected because the study is based on an LC_{50} and in keeping with the hyperconservative nature of Tier 1. This gives an ENEV of 14.7 mg/L Cl^- and an EEV/ENEV of 816. This quotient is above 1 and so a Tier 2 assessment should be conducted for zooplankton.

Anderson et al (1948) investigated the immobilization of *Daphnia magna* in various chloride salts. The EC_{50} was 432 mg/L for potassium chloride (205.4 mg/L Cl^-), 740 mg/L for magnesium chloride (551 mg/L Cl^-) and 920 calcium chloride (587 mg/L Cl^-) and, as already mentioned, 3,680 mg/L for sodium chloride (2,233 mg/L Cl^-). If an application factor of 100 is again used, this gives a Risk Quotient of 4,869, 1,815, 1,703 and 44 respectively.

7.2.11 Benthic Invertebrates

A number of papers were located investigating the tolerance of benthos to sodium chloride. Mackie (1978) reported that the clam, *Musculium securis*, produced no viable offspring at a sodium chloride concentration of 1,000 mg/L versus 54.3 young per rearing dish in the controls. However, clams were reared in distilled or deionized water to which had been added air-dried soil and willow or elm trees and the experiment was conducted over 60-80 days. This experiment is not used in this study because of its duration and the fact that deionized or distilled water was used as a culturing medium.

Birge et al. (1985) reported that the snail, *Physa gyrina*, had an LC_{50} of 2,540 mg/L Cl^-. The test was conducted over 96-hours in reconstituted water. A more sensitive response was exhibited by the oligochaete worm, *Nais variabilis*, which experienced 100% mortality when exposed to a 3,735 mg/L sodium chloride (2,267 mg/L Cl^-) solution in filtered lake water over 48 hours (Hamilton et al. 1975). Mortality data was presented in graphical form. The LC_{25} was extrapolated by eye from the figure and estimated at 2,000 mg/L NaCl (1,214 mg/L chloride). An application factor of 10 is being used because the study was short in duration (i.e., 48 hours) and because of the hyperconservative nature of Tier 1. This gives an ENEV of 12.1 mg/L chloride and an EEV/ENEV of 82.4. This quotient is above 1 and so a Tier 2 assessment should be conducted for benthos when exposed to sodium chloride.

Hamilton et al. (1975) reported a 100% mortality of, *N. variabilis*, when exposed to 204 mg/L KCl (130.4 mg/L Cl^-). Patrick et al. (1968) reported an LC_{50} for the freshwater snail *Physa heterostropha* when exposed to a 940 mg/l KCl (600 mg/L Cl^-) for 48 hours. Using the Hamilton et al. (1975) study and an application factor of 100 (because of the severity of the response) gives an ENEV of 1.3 mg/L chloride and a Risk Quotient of 7,692 for chloride as potassium chloride.

7.2.12 Amphibians

Beak (1999) conducted a 7-day survival test using eggs from the amphibian, *Xenopus leavis*. This frog is the African clawed frog, a common laboratory test species. The EC_{50} was 2,510 mg/L sodium chloride (1,524 mg/L Cl^-). *X. leavis* is not a native species to North America, although it has been introduced into southern California, and apparently is salt tolerant since it is able to acclimate to hypersaline media (Romspert 1976). Therefore, an application factor of 100 is employed because of the hardy nature of this species, particularly to salt. This gives an ENEV of 15.2 mg/L Cl^- and an EEV/ENEV of 656. This quotient is above 1 and so a Tier 2 assessment should be conducted for amphibians when exposed to sodium chloride.

Beak (1999) used potassium chloride as a reference toxicant in its amphibian tests. The 7-day EC_{50} was 2,230 mg/L KCl (1,425 mg/L Cl^-) which is slightly lower than that for sodium chloride. The ENEV is 14.3 mg/L Cl^- and the EEV/ENEV is 701.

7.2.13 Fish

A large number of toxicity tests were located for fish. However, many were very short in duration (i.e., one day) in comparison to fish longevity. Most tests were based on adults who may be more tolerant of exposures to high salinities than eggs and larvae. Accordingly, a series of tests were conducted by Beak (1999). For the fathead minnow, *Pimephales promelas*, the LC_{50} for larvae exposed to a series of chloride solutions for a 7-day exposure period was 5,490 mg/L sodium chloride (3,332 mg/L Cl^-) while the EC_{50} for the more sensitive embryo was 1,440 mg/L sodium chloride (874 mg/L Cl^-). The EC_{25} for rainbow trout, *Oncorhynchus mykiss*, eggs/embryos was 1,630 mg/L sodium chloride (989 mg/L Cl^-) while the EC_{25} for embryo/alvins was 1,830 mg/L sodium chloride (1,111 mg/L Cl^-). Thus, the most sensitive response (EC_{25}) was exhibited by rainbow trout egg/embryo survivorship. An application factor of 10 is being applied to this data because both rainbow trout and fathead minnows are hardy species readily reared in the laboratory, with a broad geographic range (Scott and Crossman 1973), often including saline habitats (Rawson and Moore 1944), and in keeping with the hyperconservative nature of Tier 1.

The EEV/ENEV is 101. This quotient is above 1 and so a Tier 2 assessment should be conducted for fish when exposed to sodium chloride.

A number of tests were located in the literature investigating the toxicity of calcium chloride to fish. Many tests were less than one day in duration and hence of little value. Some tests were located which were of longer duration, but many are from the older literature and thus of questionable value. Tests involving magnesium chloride also were conducted more than 60 years ago, were short in duration, and involved high exposures. Nothing more can be inferred about the toxicity of calcium and magnesium chloride solutions with these data sets.

A number of short-term (i.e., one day) tests have been conducted exposing fish to potassium chloride solutions. A few longer-term tests were located in the literature with bluegill sunfish (*Lepomis macrochirus*) experiencing 50% mortality when exposed to a 2,010 mg/L potassium chloride (1,284 mg/L Cl^-) solution for 4 days (Patrick et al. 1968). Beak (1999) used potassium chloride as a reference toxicant in its sodium chloride tests. The LC_{50} for the 7-day fathead minnow test was 861 mg/L KCl (550 mg/L Cl^-) versus 5,490 mg/L for sodium chloride. If an application factor of 10 is used, the EEV/ENEV is 116 for potassium chloride.

7.2.14 Conclusions for Tier 1 Assessment

1) Toxicity data is limited for aquatic fungi. Based on the one study located, chloride at a concentration of 10,000 mg/L may adversely affect the sporulation of these fungi for exposure times as short as 48 hours. Therefore, a Tier 2 assessment should be conducted for aquatic fungi. Aquatic fungi may be more sensitive to chloride than bacteria. Fungi may experience similar toxicity when the chloride is bound with calcium, but even higher toxicity when exposed to magnesium chloride.Toxicity data is also limited for protozoans. The Tier 1 assessment for the most sensitive species, *Paramecium tetrourelia*, suggested that chloride at a concentration of 10,000 mg/L may adversely affect cell growth over exposure times of 5 days. Therefore a Tier 2 assessment should be conducted for protozoans.

2) Limited toxicity data was located for phytoplankton. The Tier 1 assessment suggested that a chloride concentration of 10,000 mg/L would have an adverse impact on diatom growth over exposure times of 120 hours. Therefore, a Tier 2 assessment should be conducted for phytoplankton. Diatoms possibly are more sensitive to chloride than flagellates. Diatoms may have a greater tolerance for chloride as calcium chloride and a lower tolerance as potassium chloride than for chloride bound with sodium.

3) The limited toxicity data found for macrophytes suggested that a chloride concentration of 10,000 mg/L would adversely affect the long-term growth of two species of *Sphagnum*. Therefore, a Tier 2 assessment should be conducted for macrophytes.

4) Zooplankton toxicity data was limited to cladocerans and focused primarily on the cladocerans *Daphnia* and *Ceriodaphnia*. Toxicity data suggested that exposure to 10,000 mg/L chloride would be toxic to some zooplankton in a matter of days and a Tier 2 assessment should be conducted. Zooplankton may be most sensitive to potassium chloride, followed by magnesium chloride, calcium chloride and then sodium chloride.

5) A number of benthic studies were found. Toxicity data suggests that a 10,000 mg/L chloride concentration would be toxic to some benthic species and that a Tier 2 assessment should be conducted. Benthos may be more sensitive to potassium than sodium chloride.

6) Amphibians also were sensitive to chloride at concentrations of 10,000 mg/L with significant mortality occurring after a matter of days. Accordingly, a Tier 2 assessment should be conducted for amphibians. Amphibians may be more sensitive to potassium than sodium chloride.

7) A number of fish studies were located. The lowest toxicity values were found for fathead minnow eggs that would experience significant toxicity at 10,000 mg/L chloride. Accordingly, a Tier 2 assessment should be conducted for fish. Fish may be most sensitive to potassium chloride, followed by magnesium chloride, sodium chloride, and then calcium chloride.

7.3 Tier 2 Assessment

Tier 2 environmental assessments involves a further analysis of exposure and/or effects to calculate a quotient that is still conservative, but more "realistic" than the hyperconservative quotient calculated in Tier 1 (Environment Canada 1997c). The EEV also could be lowered by several means including the use of data from more recent studies. ENEVs can be lowered by decreasing the magnitude of the application factor. This can be done if the substance is not persistent and does not bioaccumulate.

For Tier 2 assessments, a more realistic estimate for the EEV is required for chloride concentrations which, while realistic, remained conservative. Road salt has been shown to have its greatest impact on the chloride levels in urban creeks and rivers. In Canada, the best-documented impacts of road salt on chloride levels in aquatic systems have been for the Toronto area. Hence, this data set is examined to find a more realistic estimate of the Tier 2 EEV.

As previously noted, chloride concentrations were measured in several streams in the Toronto Remedial Action Watershed over 1990-1996 (see section 4). Data was collected seasonally. Most stations were sampled less than 50 times during the multi-year study (see Table 4.1). Minimum chloride concentrations ranged from 1-51 mg/L Cl^- while maximum concentrations were considerably higher. Three stations were examined at Etobicoke Creek and had maximum reported chloride concentrations of 2,140-3,780 mg/L Cl^-. Maximum chloride concentration at Mimico Creek was 3,470 mg/L Cl^- and at the Black Creek station was 4,310 mg/L Cl^-. Five stations monitored on the Humber River had maximum chloride concentrations ranging from 96-4,310 mg/L Cl^-. Three stations monitored on the Don River had maximum concentrations of 960-2,610 mg/L Cl^- while one station monitored at Highland Creek had a maximum concentration of 1,390 mg/L Cl^-. Seasonal plots of chloride variations in Highland Creek (see Figure 4.3) over 1990-1993 show several winter sampling periods in which chloride concentrations exceeded 1,000 mg/L Cl^-. Williams et al. (1999) report that an Ontario spring located near a highway and bridge has a mean chloride concentration of 1,092 mg/L Cl^- as a probable result of road salt contamination. Snowmelt can contain high concentrations of chloride. For example, Delisle and André (1995) reported that chloride in snow carried by Montreal trucks occurred in an average concentration of 3,851 mg/L. Chloride concentrations in snow storage sites may also reach 1,210 mg/L (Table 4.4).

While the maximum chloride value observed in creeks and rivers in the Toronto area which have been contaminated by road salt is in the range of 2,000-4,000 mg/L Cl^-, the frequency of such occurrences, even in winter months, is likely to be low. Moreover, the duration is likely to be short (hours to a couple of days). However, exposures to chloride concentrations in the 2,000-4,000 mg/L range of this magnitude would not be as short in duration for in slow-flowing aquatic environments impacted by leakage from salt storage depots or from snow dumping, e.g., ponds, marshes, wetlands. Considering these various factors, a more conservative estimate of 1,000 mg/L will be used as the EEV in the Tier 2 assessments. Chloride concentrations in this range have been commonly observed in Toronto area creeks and rivers, as already noted, in a contaminated spring in the Toronto area (Williams et al. 1997), and in a bog contaminated by a salt-storage depot (Wilcox 1982).

Chloride is persistent (i.e., conservative) in the environment and, in fact, can be used as a tracer for water movement. Some organisms, poikilosmotic regulators (see Figure 5-1) allow their internal fluids to increase in salinity with the increasing salinity of their environment. Most freshwater organisms allow their internal salt concentrations to conform to that of their environment. Thus, in a sense, these organisms "bioaccumulate" salt when their environment suddenly becomes more saline (e.g., when there is a pulse of road salt in their environment). Thus, because chloride is persistent and does "bioaccumulate", there is little rationale for lowering the Tier 1 Application Factors. While there is some data to allow for the estimation of the concentration at which no effects are expected to be observed with long-term exposure to elevated chloride concentrations, these estimates will be used in the Tier 3 assessment. Too little information was obtained on the effects of water temperature on toxicity to readjust ENEVs for late winter and early spring water temperatures. Considerations of these issues also are dealt with in the Tier 3 assessments along with considerations of water hardness. No further calculations are performed of Risk Quotients for chloride as calcium, potassium, and magnesium salts.

7.3.1 Bacteria

The Tier 1 EEV/ENEV chloride quotient for bacteria was 187.9. The Tier 1 Application Factor is reduced from 100 to 31.6 and the EEV from 10,000 mg/L Cl^- to 1,000 mg/L Cl^-. Under Tier 2, the Toxicity Risk Quotient is reduced by a factor of 31.7 or to 5.94. This quotient is greater than 1 and so a Tier 3 assessment should be conducted for bacteria.

7.3.2 Fungi

The Tier 1 EEV/ENEV chloride quotient for fungi was 2,500. The Tier 1 Application Factor unchanged at 100 but the 2,000-4,000 mg/L $^-$. Under Tier 2, the Risk Quotient is reduced to 250. This quotient is greater than 1 and so a Tier 3 assessment should be conducted for fungi.

7.3.3 Protozoans

The Tier 1 EEV/ENEV for the protozoan, *Paramecium tertourelia*, is 28.6. No application factor was used in this assessment. Under Tier 2, the EEV/ENEV quotient is reduced to 2.9. This quotient is above 1 and so a Tier 3 assessment should be conducted for protozoans.

7.3.4 Phytoplankton

The Tier 1 EEV/ENEV for the diatom, *Nitzschia linearis*, was 678. Under Tier 2, the EEV/ENEV quotient is reduced to 67.8. A Tier 3 assessment should be conducted for phytoplankton.

7.3.5 Macrophytes

The bog moss, *Sphagnum fimbriatum*, had a Tier 1 quotient of 69.5. The Tier 2 EEV/ENEV quotient is 66.7. Since this is greater than 1, a Tier 3 assessment should be conducted for macrophytes.

7.3.6 Zooplankton

Ceriodaphnia dubia had a Tier 1 EEV/ENEV of 816. The Application Factor remains the same at 100 and the Tier 2 EEV/ENEV is reduced to 81.6 which is above 1. A Tier 3 assessment should be conducted for zooplankton.

7.3.7 Benthic Invertebrates

The oligochaete worm, *Nais variabilis*, had a Tier 1 EEV/ENEV of 82.4. The Tier 2 Risk Quotient is 8.2. A Tier 3 assessment should be conducted for benthos when exposed to sodium chloride.

Table 7-2: Summary of the Tier 2 Calculations. CTV is the Critical Toxicity Value, EEV is the maximum estimated concentration (1,000 mg/L) of chloride in the environment (from highway runoff) and EEV, is the Estimated No effects value (CTV/AF). See text for further explanation.

Endpoint Organism	Exposure Time/Endpoint	CTV NaCl (and Cl) (mg/L)	Application Factor	EEV/ENEV	Toxicity Source
1. Fungi					
Unknown aquatic fungi	sporulation, 48 hours	659 (400)	100	250	Sridhar and Barlocher
2. Protozoans					
Paramecium tetrourelia	17% reduction of cells cultured in light, 57-day test	577 (350)	1	2.9	Cronkite et al. 1985
3. Phytoplankton					
Nitzschia linearis	50% reduction in number of cells, 120 hours	2,430 (1,475)	100	67.8	Patrick et al. 1968
4. Macrophytes					
Sphagnum fimbriatum	43% reduction in growth, 45 days	2,471 (1,500)	100	66.7	Wilcox 1984
5. Zooplankton					
Ceriodaphnia dubia	50% mortality, 7 days	2,019 (1,225)	100	81.6	Cowgill and Milazzo 1990
6. Benthic invertebrates					
Nais variabilis	LC_{25}, 48 hours	2,000 (1,214)	10	8.2	Hamilton et al. 1975
7. Amphibians					
Xenopus leavis	EC_{50}, 7 days	2,510 (1,524)	100	65.6	Beak 1999
8. Fish					
Oncorhynchus mykiss	EC_{25}, 7 days, egg/embryo.	1,630 (989)	10	10.1	Beak 1999

7.3.8 Amphibians

The amphibian *Xenopus leavis* had a Tier 1 EEV/ENEV of 656. The Application Factor is 100 and the chloride concentration is reduced to 1,000 mg/L, the Tier 2 Risk Quotient is reduced to 65.6. This quotient is above 1 and so a Tier 3 assessment should be conducted for amphibians when exposed to sodium chloride.

7.3.9 Fish

A number of tests were found investigating sodium chloride toxicity to various species of fish. The most sensitive endpoint organism was rainbow trout, *Oncorhynchus mykiss*, eggs/embryos. The EEV/ENEV for Tier 1 was 101. When the Application Factor is 10 and the chloride concentration is 1,000 mg/L, the Risk Quotient is reduced to 10.1. This quotient is above 1 and so a Tier 3 assessment should be conducted for fish when exposed to sodium chloride.

7.3.10 Conclusions for Tier 2 Assessment

1) Tier 2 risk assessments were conducted by reducing the EEV from 10,000 mg/L Cl^- to 1,000 mg/L Cl^-. All EEV/ENEV quotients were greater than 1. Quotients 250 for fungi, 2.9 for protozoans, 67.8 for phytoplankton, 66.7 for macrophytes, 81.6 for zooplankton, 8.2 for benthos, 65.6 for amphibians, and 10.1 for fish.

2) Tier 2 risk assessments were made using an EEV of 1,000 mg/L chloride. Even if this value were reduced to 500 mg/L chloride, based on less conservative exposure estimates, all quotients would continue to exceed 1.

3) Application Factors were 100 for fungi, phytoplankton, macrophytes, zooplankton, and amphibians. Reducing the Application Factor to 1 would result in a Tier 2 Risk Quotient would still result in all quotients exceeding 1.

4) An application factor of 10 was used for benthos and fish. Reducing the Application Factor to 1 would result in a Tier 2 Risk Quotient of <1 for benthos and a Risk Quotient of 1.01 for fish.

5) Tier 3 assessments should be conducted for all groups of organisms considered in Tiers 1 and 2.

7.4 Tier 3 Assessment

Tier 3 assessments provide for the analysis of the likelihood that the substance under consideration will have a harmful impact on the environment. It does so by considering the distributions of exposures and/or effects (Environment Canada 1997c).

7.4.1 Tier 3 Assessment of Road Salt Concentrations in the Aquatic Environment

A review of the existing literature on chloride concentrations in the Canadian environment, particularly chloride that could be associated with road salt, yielded surprisingly few studies. As noted in Section 4, most of the Canadian studies were located in the Maritimes, Quebec, and Ontario, regions of the heaviest road salt application (Fig. 4-1). Highest chloride concentrations tended to be associated with highway runoff and small creeks and rivers in the Toronto area. Considerably lower concentrations were observed in the less densely populated areas in Ontario, Quebec, and the Maritime region. This literature review, and that of other ERG members, has identified geographic regions and watersheds in which road salt is most heavily applied (Morin and Perchanok 2000). Moreover, through modeling approaches, ERG members have identified regions which, on the basis of climatic considerations, especially precipitation, and soil characteristics (i.e., bedrock versus silts, clays, etc.), surface water concentrations of chloride are most likely to be relatively high (Mayer et al. 1999; Morin et al. 2000). The approach followed is described below for Mayer et al. (1999).

Mayer et al. (1999), with the ERG Work Group, modeled chloride concentrations (g/L) in the Canadian environment based on road salt application rates, precipitation, and watershed size. Average watershed chloride concentration (mg/L) was estimated by dividing the unit area road salt loading by a runoff coefficient, where:

1) The unit area loading ($g/m^2/yr$) for the watershed was calculated by dividing the road salt application for a given watershed (g/y) by the area of the watershed (m^2). This approach, while useful, "dilutes" the localized application rate on the highway.

2) The runoff coefficient (m/yr) was estimated by dividing the total annual volumetric runoff (m^3/yr) for that watershed by the area of the watershed (m^2). Volumetric runoff, in turn, was an estimate of the proportion of total annual precipitation in the watershed that reached stream channels. However, this approach, while useful on an annual watershed basis, does not provide detailed information on pulse events and localized areas of impacts.

Mayer et al.'s (1999) study estimated that regions of highest chloride loadings per watershed (>20 g Cl^-/m^2 watershed) occurred in southern Ontario and Quebec in a narrow band along the northern Lake Erie, lake Ontario, and both sides of the St. Lawrence River. Observed average chloride concentrations in these watersheds ranged from 10-25 mg/L with road salt apparently the primary source of this chloride. The next highest loadings (up to 1 g Cl^-/m^2 watershed) on a watershed basis were in southern Ontario and Quebec, and in the Maritimes, including southern Newfoundland. Average watershed chloride concentrations in these regions were 2-10 mg/L with road salt again a major contributor to total chloride concentration. Mayer et al. also estimated that road salt was a major contributor to chloride levels in watersheds in central British Columbia and central Alberta although average watershed chloride concentrations tended to low (<5 mg/L).

These modeling studies, while not environmentally precise, identify regions in which road salt is being heavily applied (i.e., southern Ontario and Quebec, the Maritimes, including southern Newfoundland)

and areas in which road salt may be significantly increasing average surface water chloride concentrations. Unfortunately, no studies have been conducted in Canada to determine spatial patterns and long-term trends in chloride concentrations in streams and lakes in areas of dense population growth and the possible contribution of road salt to these enhanced chloride levels. This is in contrast to the United Sates where several such studies have been conducted (e.g., Peters and Turk 1981, Mattson and Godfrey 1994, Herlihy et al. 1998, Eilers and Selle 1991, Siver et al. 1996, and Dixit et al. 1999). Moreover, while Mayer et al.'s (1999) modeling studies provide for the identification of watersheds that are vulnerable to road salt impacts, they do not provide site-specific and time-specific estimates of chloride concentrations in these watersheds. The actual concentrations that would be observed in these areas would vary as noted in Section 2.

Accordingly, to continue with the Tier 3 assessment, a scenario type approach will be used to describe areas of probable or possible impact. Scenarios will begin with brief exposures to highly elevated chloride concentration (i.e., in runoff and in streams draining extensive highway areas) and move to extended exposures to low elevations in chloride concentrations (i.e., large rivers, lakes).

7.4.2 Tier 3 Assessments of Chloride Toxicity

Tier 3 assessments include considerations of the distribution of exposures and/or effects in the environment (Environment Canada 1997c). In order to proceed with this approach, the toxicity of road salt in the environment is investigated by examining all the accrued data sets to assess the likelihood that toxicity responses would be observed. Exposure period is very important. Accordingly, the data is divided into four time periods: less than one day, one day, four days, and one week. In addition, chronic toxicity (long-term) values are estimated as described later in this section.

i. Less Than One Day Exposure

Rapid snowmelt events following road salt application can deliver pulses of concentrated chloride solutions to aquatic ecosystems. Highly elevated concentrations may last less than one day with concentrations declining through the melt event. Therefore, it is instructive to examine toxicity data to assess levels at which exposures of less than one day may have lethal effects.

Only a few studies investigated the toxicity response of various taxa to short-term exposures to sodium chloride (i.e., exposures ranging from 15 minutes to 12 hours). For the <1 day data sets (Table 7-3), LC_{50} values ranged from 8,767-50,000 mg/L NaCl or 5,318-30,330 mg/L chloride. The geometric mean is 12,826 mg/L chloride. Salt concentrations at the upper end of this range have been associated with direct road runoff in urban areas and other heavily trafficked areas (see Section 4). Concentrations at the lower end of the range have been associated with storm sewers and small urban streams during periods of high road salt application rate (Bubeck et al. 1971; Hawkins and Judd 1972; Cherkauer 1975) and with ditches (Champagne 1978). Concentrations of this magnitude also have been noted in wetland areas, which have been adversely impacted by leakage from road salt depots (Ohno 1990), and in groundwater and leachates from road salt storage depots (Morin 2000; Arp 2001).

Table 7-3: Toxicity responses of organisms to sodium chloride for exposures less than one day.

SPECIES	TAXON	NACL (MG/L)	CL (MG/L)	RESPONSE	TIME (HR)	SOURCE
Salvelinus fontinalis	brook trout	50,000	30,330	LC_{50}	0.25	Phillips 1944
Lepomis macrochirus	Bluegill	20,000	12,132	LC_{47}	6	Waller et al. 1996
Oncorhynchus mykiss	Rainbow trout	20,000	12,132	LC_{40}	6	Waller et al. 1996
Chironomus attenatus	Chironomid	9,995	6,063	LC_{50}	12	Thorton and Sauer 1972

The evidence suggests that direct road salt runoff can be toxic to a wide variety of organisms over relatively short exposures times. The environments in which is this is most likely to occur are those in close proximity to multilane highways and salt storage depots where relatively little dilution of the road salt has occurred. Toxicity from highway runoff may be aggravated by other contaminants carried in the runoff including metals and PAHs. Highway agencies, with information on highway width and length, road salt application rates, weather events, and the major features of urban ditches, roadside wetlands, and small creeks, should be able to identify potentially vulnerable areas. In situ monitoring of transient changes in conductivity (ideally chloride) as in Whitfield and Wade (1992) should help determine the nature and significance of such events. Water quality monitoring studies conducted around road salt storage depots will help identify potentially sensitive wetland areas and creeks.

ii. One Day Exposure

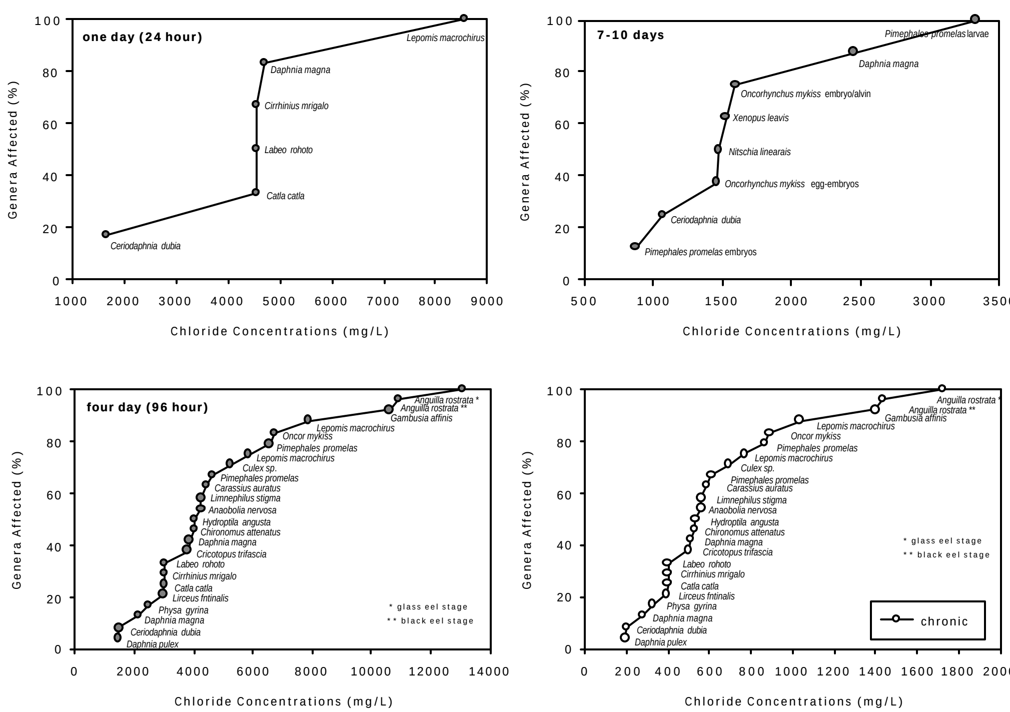
Road salt, on entering the environment, may persist in a relatively concentrated form for some period of time. For example, during melting events, chloride concentrations may increase in ditches and streams and remain elevated for a day or more. Stored chloride may be flushed from stream banks during spring, summer, and autumn storms (Scott 1980b). Accumulated chloride at the bottom of urban lakes may be flushed from the lake during spring and summer overturn (Cherkauer and Ostenso 1976). All of these events may result in brief (e.g., 1 day) exposures to elevated chloride concentrations. Some organisms, such as fish, may be able to swim away from such chloride pulses unless the stream flow velocity exceeds swimming speed or they are trapped in stationary waters (e.g., ditches, wetlands).

Six laboratory studies were located which determined 24-hour LC_{50}s (Figure 7-1; Table 7-4). This data is graphed as a cumulative frequency distribution of chloride toxicity as a sodium chloride salt (Fig. 7-1). Concentrations ranged from 2,724 mg/L NaCl (1,652 mg/L Cl^-) for the cladoceran, *Ceriodaphnia dubia*, to 14,100 mg/L NaCl (8,553 mg/L Cl^-) for bluegill sunfish. The geometric mean chloride concentration is 3,746 mg/L.

Chloride concentrations at the upper end of this range have been observed in road runoff in heavily salted areas. Concentrations at the lower end of the range, 2,000 mg/L Cl^-, have been observed for urban streams in dense population areas, such as the Metropolitan Toronto area (Table 4-1, 4-2; Figure 4-1, 4-2). In such situations, brief, undiluted pulses of road salt could have adverse impacts in aquatic ecosystems. Creeks and roadside waterways near heavily-salted, multiple lane highways also are vulnerable to such impacts.

Chloride concentrations in the 1,600-8,500 mg/L range have been observed in impacted areas near salt depots where there has been significant leakage from the salt piles (Arp 2001). Because such contaminated waters may move slowly, exposure times are likely to be longer than 1 day. Chloride concentrations in the 1,600-8,500 mg/L range also have been observed in snow removed from salted roads (Delisle and Andre 1995). If large amounts of this snow were dumped into a relatively small and slowly flowing water body (e.g., a stream or marsh) some organisms could experience toxic impacts.

Figure 7-1: Predicted chronic and actual acute chloride toxicity levels for aquatic taxa.



Environments in which one-day toxicity is most likely to be significant are heavily urbanized areas in southern Ontario, particularly immediately north of lakes Erie and Ontario and along both sides of the St. Lawrence River, especially in the Montreal-Quebec City corridor. Toxicity events may also occur with some frequency in densely populated urban areas in the Maritimes. Toxicity events may also occur near road salt storage depots and yards and from melting snow dumps. In situ monitoring of transient changes in conductivity (ideally chloride) would confirm whether such areas are, in fact, experiencing highly elevated chloride concentrations with runoff events. Monitoring of chloride concentrations in aquatic ecosystems near road salt storage facilities and near snow dumps will help resolve this issue.

Table 7-4: Toxicity responses of organisms to sodium chloride for exposures of one day.

Species	Taxon	NaCl (mg/L)	Cl (mg/L)	Response	Sources
Lepomis macrochirus	bluegill	14,194	8,616	LC_{50}	Dowden and Bennett 1965
Lepomis macrochirus	bluegill	14,100	8,553	LC_{50}	Doudoroff and Katz 1953
Carassius auratus	goldfish	13,480	8,388	LC_{50}	Dowden and Bennett 1965
Daphnia	cladoceran	7,754	4,704	LC_{50}	Cowgill and Milazzo 1990
Cirrhinius mrigalo	Indian carp fry	7,500	4,550	LC_{50}	Gosh and Pal 1969
Labeo rohoto	Indian carp fry	7,500	4,550	LC_{50}	Gosh and Pal 1969
Catla catla	Indian carp fry	7,500	4,550	LC_{50}	Gosh and Pal 1969
Salmo	Rainbow trout	5,496	3,336	LC_{50}	Kostecki and Jones 1983
Daphnia	cladoceran	2,724	1,652	LC_{50}	Cowgill and Milazzo 1990
Ceriodaphnia dubia	cladoceran	2,724	1,652	LC_{50}	Cowgill and Milazzo 1990

iii. Four-Day Exposure

Acute (or short-term) toxicity studies typically are conducted over periods of up to 96-hours (4 days) and measure mortality (Lee et al. 1997). For short-lived organisms, such as algae, a reduction in cell growth relative to the control population may be measured. In terms of the algal population, these 4-day studies then can be considered chronic studies. However, at the community level, longer exposure times would be required to measure changes in algal species succession and composition.

Short-term (in the order of four days) exposure scenarios are most applicable to stream and river situations where aquatic organisms are exposed to brief pulses of elevated chloride concentration.

Accordingly, toxicity data for relatively long-lived organisms (e.g., zooplankton, benthos, fish, and amphibians) was examined to determine the concentrations at which chloride was toxic to various components of the aquatic ecosystem.

The 4-day toxicity data set was increased by including 3-day toxicity data. In order to do this, 3-day toxicity estimates were converted to 4-day toxicity estimates through the use of a conversion factor. Conversion factors were based on Cowgill and Milazzo (1990) who investigated LD_{50} responses of two species of cladocerans to sodium chloride at daily intervals over a 7-day period. The LD_{50} for *Daphnia magna* was 7,754 mg/L NaCl at day 2 and day 3 and 6,031 mg/L NaCl at day 4 (i.e., *D. magna* was 1.27 times more sensitive to NaCl at four days of exposure than at two or three days of exposure). For *Ceriodaphnia dubia*, these values were 2,308 mg/L NaCl, 2,169 mg/L NaCl and 2,077 mg/L NaCl respectively (i.e., *C. dubia* was 1.1 times more sensitive at day 4 than at day 2 of exposure and 1.04 times more sensitive than at day 3 of exposure). The mean of these increased sensitivities was used to convert 48-hour (factor of 1.19) and 72-hour (factor of 1.16) mortality data to 96-hour data as shown below; all data is presented as LC_{50} toxicity (Figure 7-1; Table 7-5). The geometric mean was calculated for *Daphnia magna* was calculated from the four available 4-day studies.

The data is graphed as a cumulative frequency distribution of toxicity to chloride as a sodium chloride salt. Toxicity data for *C. dubia* and *D. pulex* is presented by the mean values reported in Cowgill and Milazzo (1990), Wisconsin State Laboratory of Hygiene (WI SLOH) (1995) and Birge et al. (1985). Similarly, a mean is calculated for fathead minnows based on Adelman et al. (1976) and WI SLOH (1995), but not Birge et al. (1985). These results are reported separately because it is not certain whether differences are due to experimental variations or genetic and/or phenotypic differences in the ability of the organisms tested to withstand salinity stresses.

For a 4-day exposure, chloride becomes lethally toxic to half the exposed population at concentrations ranging from 1,400 mg/L for the cladoceran, *Ceriodaphnia dubia*, to 13,085 mg/L for the black eel stage of the American eel. The median chloride toxicity was ca. 4,026 mg/L^{-}, similar to the geometric mean of 4,031 mg/L. There was a large overlap in the 1-day and 4-day toxicity curve, primarily due to the low number of 24-hour studies located in the search of the literature.

Concentrations of 1,400-3,000 mg/L Cl^{-} typically have been associated only with urban creeks and rivers in relatively densely populated areas, such as the Metropolitan Toronto area. However, concentrations in this range appear to occur infrequently, even during winter (see Figures 4.2 and 4.3). Schroeder and Solomon (1999) conducted a formal investigation of chloride concentrations and toxicity at three stations on the Don River. Chloride data collected from November to April 1995 (or 1996) was plotted as cumulative frequency graphs. Chloride concentrations reached up to ca. 950 mg/L at site 1, up to ca. 2,600 mg/L at site 2, and up to ca. 1,150 mg/L at site 3. Next acute LC_{50} data for 13 fish and 7 invertebrate taxa was plotted on the same graph. Experiments were at least 24 hours in duration, but the total duration of each experiment was not given. LC_{50} values ranged from ca 1,000-30,000 mg/L Cl^{-} for most taxa. Thus, the toxicity range was greater than the 4-day toxicity data shown in Fig. 7.1. Schroeder and Solomon estimated that 90% of the fish species would be expected to be protected from acute effects of chloride toxicity at least 90% of the time at two of the sites and at least 85% of the time at the third site. Sensitive species had LC_{50}'s of ca 1,000 mg/L to ca. 2,700 mg/L Cl^- and would experience significant chloride stress some 10-15% of the time at these three sites in the Don River.

Maximum chloride concentrations in creeks in the Toronto Remedial Action Plan Watershed attain levels of 1,040 mg/L Cl^- to 4,310 mg/L Cl^- (see Tables 4.1 and 4.2), i.e., greater than values reported in Schroeder and Solomon 1999. Thus, based on Schroeder and Solomon's study, it is probable that aquatic organisms inhabiting these waters are acutely stressed by road-salt containing highway runoff more than 10% of the time during the winter period. This 10% value is likely to be an underestimate for a number of reasons. First, because sampling frequency in these rivers has not been intense, it is probable that even higher chloride levels do occur on occasion. Second, laboratory studies routinely use organisms that are ubiquitous in the environment and also are very hardy (i.e., they can be cultured and maintained as self-reproducing populations in the laboratory). Most organisms are considerably more sensitive and, as such, may have greater sensitivities to chloride and other related stresses. Laboratory studies are conducted on healthy, well-fed animals. In the real environment, animals generally experience significant periods of low food availability, thermal stress, and other stresses. Thus, they may be less capable of withstanding chloride stress than well-maintained laboratory specimens, particularly during winter, a period of low temperatures and limited food availability. Finally, road salt runoff can contain a variety of other contaminants that also may contribute to toxic stress.

As previously noted, chloride concentrations in the 1,000-3,000 mg/L range have been associated with ditches and with wetlands contaminated by road salt storage depots. It is probable that organisms inhabiting such saline stationary waters will experience significant stress in a matter of days. Chloride concentrations of 1,000-3,000 mg/L correspond to sodium chloride concentrations of 1,650-4,944 mg/L, (i.e., the upper part of the subsaline range and lower part of the hyposaline range as defined by Hammer 1986a). This is the range over which freshwater organisms decline rapidly in diversity.

Table 7-5: Four day LC_{50} of various taxa exposed to sodium chloride.

Species	Taxon	NaCl (mg/L)	Cl (mg/L)	Sources
Ceriodaphnia dubia	Cladoceran	2,308	1,400	Cowgill and Milazzo 1990
Daphnia pulex	Cladoceran	2,422	1,470	Birge et al. 1985
Ceriodaphnia dubia	Cladoceran	2,630	1,596	WI SLOH 1995
Daphnia magna	Cladoceran	3,054	1,853	Anderson 1948
Daphnia magna	Cladoceran	3,550	2155	Biesinger and Christensen 1972
Daphnia magna	Cladoceran	3,939	2,390	Arambasic et al. 1995
Physa gyrina	Snail	4,088	2,480	Birge et al. 1985
Lirceus fontinalis	Isopod	4,896	2,970	Birge et al. 1985
Chlamydomonas reinhardtii	Algae	4,965	3,014	Reynoso et al. 1982
Catla catla	Indian carp fry	4,980	3,021	Gosh and Pal 1969
Labeo rohoto	Indian carp fry	4,980	3,021	Gosh and Pal 1969
Cirrhinius mrigalo	Indian carp fry	4,980	3,021	Gosh and Pal 1969
Cricotopus trifascia	Chironomid	5,192	3,795	Hamilton et al. 1975
Hydroptila angusta	Caddisfly	5,526	4,039	Hamilton et al. 1975
Daphnia magna	Cladoceran	6,034	3,658	Cowgill and Milazzo 1990
Chironomus attenatus	Chironomid	6,637	4,026	Thorton and Sauer 1972
Daphnia magna	Cladoceran	6,709	4,071	WI SLOH 1995
Anaobolia nervosa	Caddisfly	7,014	4,255	Sutcliffe 1961b
Limnephilus stigma	Caddisfly	7,014	4,255	Sutcliffe 1961b
Carassius auratus	Goldfish	7,341	4,453	Adelman et al. 1976
Pimephales promelas	Fathead minnow	7,650	4,640	Adelman et al. 1976
Pimephales promelas	Fathead minnow	7,681	4,600	WI SLOH 1995
Culex sp.	Mosquito	8,614	5,229	Dowden and Bennett 1965
Lepomis macrochirus	Bluegill	9,627	5,840	Birge et al. 1985
Pimephales promelas	Fathead minnow	10,831	6,570	Birge et al. 1985
Oncorhynchus mykiss	Rainbow trout	11,112	6,743	Spehar 1987
Lepomis macrochirus	Bluegill	12,964	7,864	Trama 1954
Gambusia affinis	Mosquito fish	17,500	10,616	Wallen et al. 1957
Anguilla rostrata	American eel, glass eel stage	17,969	10,900	Hinton and Eversole 1978
Anguilla rostrata	American eel, black eel stage	21,571	13,085	Hinton and Eversole 1978

iv. Seven to Ten Day Exposures

Most acute toxicity studies are conducted over periods of 4 days or less. However, a number of studies have recently been developed which extend toxicity measurements from 7-10 days. These studies, in addition to considering mortality, may also investigate growth and, for rapidly reproducing organisms, such as cladocerans, fecundity.

Two studies (Cowgill and Milazzo 1990; Gonzales-Moreno et al. 1997) were located which investigated 7-10 day toxicity to chloride (Table 7-6). In addition Beak (1999) conducted a series of 7-day toxicity tests using fathead minnows, rainbow trout, and the African clawed frog. This data provides some insight to toxicity in waters with slow exchanges (e.g., ditches, wetlands, ponds, small lakes). Data was presented as LC_{50} and EC_{50}. Cowgill and Milazzo (1990) reported LC_{50}s and three (*Ceriodaphnia dubia*) to four (*Daphnia magna*) measures of EC_{50} (Table 7-6). For *Daphnia magna*, the 7-day LC_{50} was 6,034 mg/L NaCl (3,660 mg/L Cl^-); the 7-day EC_{50} for total progeny, mean number of broods, and mean brood size ranged from 4,040-5,777 mg/L NaCl (2,451-3,506 mg/L Cl^-). For *Ceriodaphnia dubia*, the 7-day LC_{50} was 2,019 mg/L NaCl (1,225 mg/L Cl^-); the EC_{50} ranged from 1,761-1,991 mg/L NaCl (1,068-1,208 mg/L Cl^-) for total progeny, mean number of broods, mean brood size, and dry weight.

Beak (1999) reported most toxicity data as EC_{50}, defined as the "median effective concentration that causes an adverse effect in 50% of the test organisms". Where mortality was the 'effect", the median was calculated by excluding test results from the higher salinities (i.e., 4,000 and 8,000 mg/L) if no test animals survived the experiment. The EC_{50} for fathead minnow growth was only 10% lower than that of survivorship.

This experimental data revealed that, as exposure time increases, nonlethal effects such as reduced growth and reproduction begin to be observed as animals (and diatoms) are stressed by higher salinities. The 7-day EC_{50} data is plotted on Figure 7-1, using the lowest value for a given species and life history stage from Table 7-6. This data shows that chloride becomes toxic at concentrations ranging from 874-3,330 mg/L for 7-day exposures with a geometric mean of 1,718 mg/L Cl^- versus 2,288 mg/L Cl^- for the 7-day LC_{50} data. Concentrations of this magnitude have been associated with ditches, urban creeks, urban rivers, and in small, urban lakes in areas such as Toronto. For example, the median December-February chloride concentration in Mimico Creek was ca. 840 mg/L Cl^- over the 1989-1995 period: creeks such as Etobicoke and Highland have elevated chloride concentrations in this range during some winter periods (Figures 4-2 and 4-3). Moreover, small, urban lakes that have chloride concentrations in this range may develop anoxic bottom waters, another source of stress to the aquatic community. Chloride concentrations in this range may occur in wetlands near road-salt storage depots also may be expected to result in toxic stress to aquatic communities.

7.4.3 Chronic Toxicity

Most toxicity studies were conducted over periods of one week or less, providing estimates of the vulnerability of the aquatic ecosystem to brief pulses of evaluated chloride concentrations (i.e., organisms living in streams, rivers, and ditches). However, organisms living in more stationary (lentic) waters, such as wetlands, lakes, and ponds, may experience elevated chloride concentrations for considerably longer periods of time. During this period, non-lethal effects may begin to manifest themselves in reduced growth and reproductive rates, increased metabolic rates, etc., some of which may lead to death. Unfortunately, few laboratory studies have investigated long-term, chronic effects.

Two studies were located which investigated chronic toxicity (Birge et al. 1985; Spehar 1987). These studies allow for the calculation of an acute:chronic toxicity ratio which can the be applied to the 4-day LC_{50} data.

Birge et al. (1985) determined the NOEC and LOEC toxicity of chloride to fathead minnow (32 day test) and *Daphnia pulex* (21 day test). The geometric mean of these two estimates is the chronic concentration. For *D. magna*¸ the NOEC for reproduction was 314 mg/L Cl^- while the LOEC (27% reduction in reproduction or number of offspring) was 441 mg/L Cl^-. The geometric mean of these two values is 372 mg/L Cl^-. The 48-hour LC_{50} for *D. pulex* was 1,470 mg/L Cl^-, giving an acute:chronic ratio of 3.95. It is important to note that Birge et al.'s (1985) studies were conducted using reconstituted water which gave twice the 48-hour LC_{50} for *D. pulex* as tests using natural waters. However, for this exercise, it is assumed that this does not affect acute:chronic ratios.

Table 7-6: Seven to ten day LC_{50} and EC_{50} of various taxa exposed to sodium chloride.

Species	Taxon	NaCl (mg/L)	Cl (mg/L)	Exposure	Response	Sources
Ceriodaphnia dubia	cladoceran	2,019	1,225	7 days	LC_{50}	Cowgill and Milazzo 1990
Xenopus leavis	frog embryo	2,940	1,784	7 days	LC_{50}	Beak 1999
Daphnia magna	cladoceran	6,031	3,660	7 days	LC_{50}	Cowgill and Milazzo 1990
Pimephales promelas	fathead minnow larvae,	5,490	3,330	7 days	LC_{50}	Beak 1999
Pimephales promela	fathead minnow embryos, normal	1,440	874	7 days	EC_{50}	Beak 1999
Ceriodaphnia dubia	cladoceran, mean brood size	1,761	1,068	9 days	EC_{50}	Cowgill and Milazzo 1990
Ceriodaphnia dubia	cladoceran, total progeny	1,761	1,088	9 days	EC_{50}	Cowgill and Milazzo 1990
Ceriodaphnia dubia	cladoceran, mean number of broods	1,991	1,208	9 days	EC_{50}	Cowgill and Milazzo 1990
Oncorhynchus mykiss	rainbow trout egg embryo, normal	2,400	1,456	27 days	EC_{50}	Beak 1999
Nitschia linearais	diatom, cell numbers	2,430	1,475	7 days	EC_{50}	Gonzales-Moreno et al.1997
Xenopus leavis	frog embryo, normal development	2,510	1,524	7 days	EC_{50}	Beak 1999
Oncorhynchus mykiss	rainbow trout embryo/alvin, normal development	2,630	1,595	27 days	EC_{50}	Beak 1999
Daphnia magna	cladoceran, mean brood size	4,040	2,451	10 days	EC_{50}	Cowgill and Milazzo 1990
Daphnia magna	cladoceran, total	4,282	2,599	10 days	EC_{50}	Cowgill and Milazzo 1990
Daphnia magna	cladoceran, dry weight	4,310	2,616	10 days	EC_{50}	Cowgill and Milazzo 1990
Daphnia magna	cladoceran, mean number of broods	5,777	3,506	10 days	EC_{50}	Cowgill and Milazzo 1990
Pimephales promelas	fathead minnow larvae, growth	4,990	3,029	7 days	LC_{50}	Beak 1999

Birge et al. (1985) reported that the NOEC for the 33-day early stage test for fathead minnow was 252 mg/L Cl^- and the LOEC (9% reduction in survivorship) as 352 mg/L Cl^- giving an estimated chronic value of 298 mg/L Cl^-. The 4-day acute toxicity was 6,570 mg/L Cl^- giving an acute:chronic ratio of 22.1. The USEPA (1988) included the results of the Birge et al. (1985) study in accessing chronic toxicity, but used the 353 mg/L Cl^- (9% reduced survival), and 533 mg/L Cl^- (15% reduction in survival) to calculate the chronic concentration (433 mg/L Cl^-) and acute:chronic ratio (15.2).

USEPA (1988) cited a study by Spehar (1987) which used early life-stages of rainbow trout. All trout were killed at a concentration of 2,740 mg/L Cl^- while survival was 54% at 1,324 mg/L Cl^- and 97% or higher at 643 mg/L Cl^- as well as at two lower concentrations and the control. No further details are provided on this acute study. USEPA estimated the chronic value at 922.7 mg/L Cl^- (the geometric mean of 1,324 and 2,740 mg/L Cl^-) and the acute:chronic ratio as 7.31.

The USEPA, which uses the Spehar (1987) study and a lower value for Birge et al.'s (1985) fathead minnow study estimated a geometric mean acute:chronic ratio of 7.6. If an estimated 4-day LC_{50} is used for *D. pulex*, as described above, this mean acute:chronic ratio becomes 6.98. Birge et at. (1985) and USEPA (1988) calculated the geometric mean of the various chronic ratios to estimate a Final Chronic Value (FCV). Birge et al.'s study provides a concentration estimate of 333 mg/L while the USEPA study provides an estimate of 226.5 mg/L. USEPA also estimated Final Acute Value (FAV) by calculating the geometric means of acute toxicity data first at the species and then at the genus level. The most sensitive genus was *Daphnia* providing a Final Acute Value of 1,720 mg/L Cl^-.

A mean acute:chronic ratio of 7.59 is used to predict the chronic values for the organisms shown in Figure 7-1 for the 4-day exposures to elevated chloride concentrations. This curve represents the range of values at which organisms may begin to respond to long-term exposure to elevated chloride concentrations. Concentrations range from 194-1,724 mg/L Cl^- with a geometric mean value of 570 mg/L Cl^-.

Concentrations at which chloride may begin to have chronic impacts on aquatic communities ranged from a low 194 mg/L Cl^- for sensitive species, such as *Ceriodaphnia dubia*, to 327 mg/L Cl^- for the snail, *Physa gyrina*, to 561 mg/L Cl^- for the caddisflies, *Anaobolia nervosa* and *Limnephilus stigma*, to 1,036 mg/L Cl^- for bluegill sunfish (Figure 7-1). Chloride concentrations is this range have again been observed for salt-impacted creeks, rivers, and lakes in urban areas and in lakes and wetlands contaminated by leakage from road salt depots.

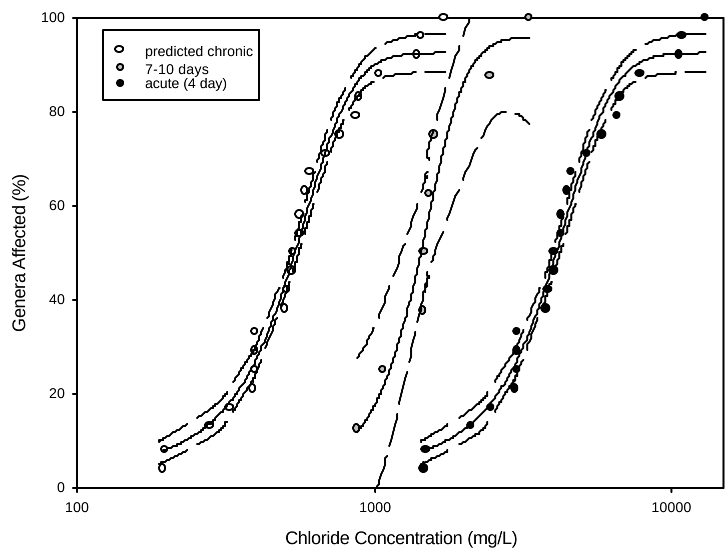
7.4.4 Short-term to Long-term Toxicity

A family of curves can be prepared based on chloride toxicity following 4-day, 7-day, and predicted long-term (chronic) exposures (Figure 7-2). Experiments of one day or less were excluded because they were few in number. For 7-10 day toxicity, the curve is based only on EC_{50} data. Data was plotted on a linear-log scale and SigmaPlot 5.0 was used to fit the data to a sigmoid model and calculate the 95% confidence intervals. This provided a similar curve fitting described in the guidance Manual

(Environment Canada 1997a). Curves follow the anticipated pattern with organisms becoming more and more sensitive to chloride following prolonged periods of exposure.

As previously noted, this toxicity data was generated using laboratory species that are relatively hardy and capable of being maintained in culture for extended periods of time. Moreover, they are well maintained with a steady and sufficient food supply and, except for the test conditions, assumed to be under ideal conditions. In the natural world, the vast majority of species are likely to be considerable more sensitive to toxic stress. Moreover, except during brief periods of time, they may be under food limitation and other physiological stresses. Highway runoff contains not only sodium chloride salts but other constituents of road salt, metals, PAHs, and other contaminants which may stress organisms in the receiving water body. Therefore, this family of curves should be viewed as conservative estimates of stress resulting from elevated chloride concentrations from road salt in the natural environment.

Figure 7-2: Predicted chronic and actual (four days and 7-10 days) chloride toxicity levels for aquatic organisms



Predicted chronic toxicity also can be shown in tabular form with confidence intervals (Table 7-7). Approximately 10% of tested species are predicted to experience acute effects as a result of long-term exposure to chloride concentrations as low as 240 mg/L; the upper 95% confidence interval is 295 mg/L while the lower interval is <194 mg/L. Fifty percent of the taxa are affected by a long-term exposure to chloride concentrations of 541 mg/L while 90% are affected at 1,018 mg/L. These effects are based on laboratory studies and do not take into account the more subtle interactions and effects which may occur in the natural world where there are a variety of stresses in the environment and species interactions.

Table 7-7: Predicted cumulative percentage of species affected at various chloride concentrations (mg/L). Also shown are the upper and the lower 95% confidence limits; data from Figure 7.2.

Cumulative % of Species Affected	Mean Chloride Concentration	Lower Confidence Interval	Upper Confidence Interval
10	240	<194	295
25	382	360	404
50	541	525	556
75	789	769	810
90	1,018	886	1,200

As previously noted, Schroeder and Solomon (1999) conducted a formal investigation of chloride concentrations and toxicity at three stations on the Don River. Chloride data collected from November to April 1995 (or 1996) was plotted as cumulative frequency graphs. Next acute LC_{50} data for 13 fish and 7 invertebrate taxa was plotted on the same graph. Experiments were at least 24 hours in duration, but the total duration of each experiment was not given. This approach can be refined using toxicity data categoried by time intervals as in Figure 7.2.

Using a similar approach as Schroeder and Solomon (1999), we plotted the November-April chloride data for the Don River at Pottery Road for the 1990-2000 period as a cumulative frequency distribution (Fig. 7-3). This site was selected because of the large number of observations. However, instead of a single toxicity curve, we plot the acute (4-day) and predicted chronic data on this curve. Chloride concentrations ranged from 11-2,610 mg/L and averaged 282 mg/L. Considering first the 4-day acute toxicity data, the two most sensitive species experience acute chloride toxicity at concentrations of ca. 1,480 mg/L. Chloride occurs in these winter concentrations for ca. 2% of the observations. Thus, acute chloride toxicity may be uncommon at the site although it must be remembered that the toxicity data probably underestimates community sensitivity. Considering next chronic toxicity, 10% of the taxa

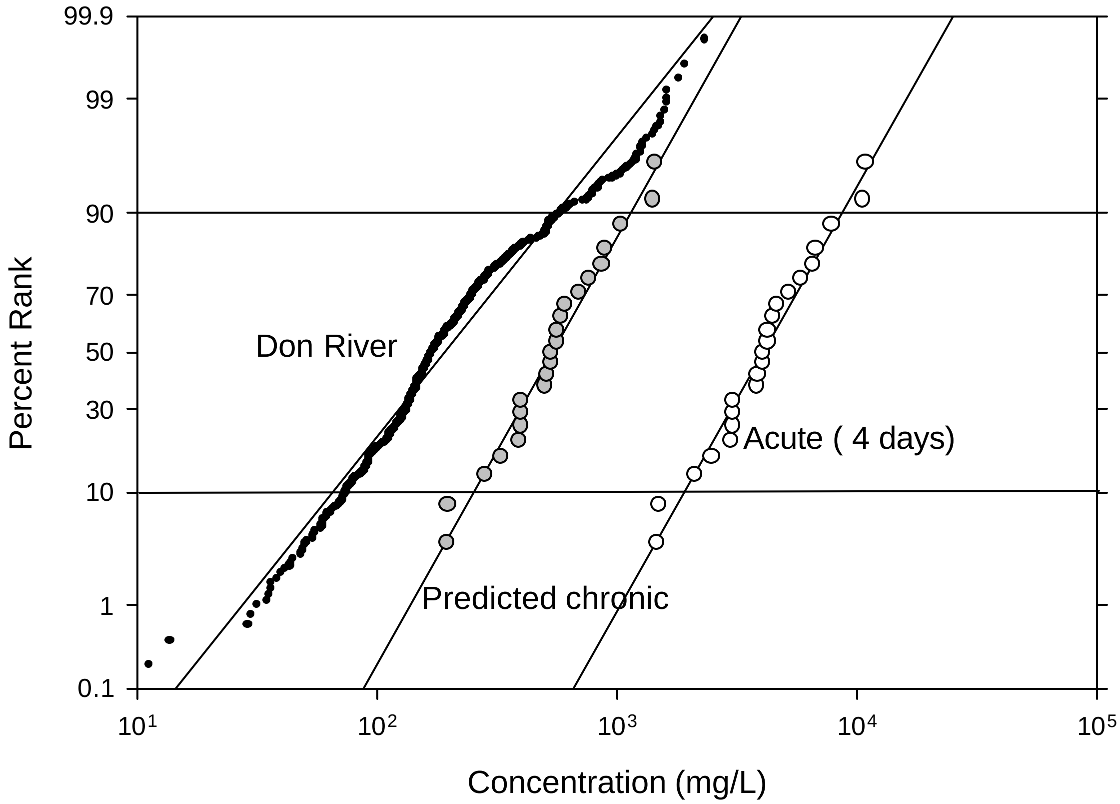


Figure 7.3: Cumulative frequency distribution of November-April, 1990-2000, chloride concentrations in the Don River at Pottery Road. Also shown is the predicted chronic and measured acute toxicity data from Figure 7-2. The 10% and 90% percentiles appear as solid lines.

Experience chronic toxicity at concentrations as low as 240 mg/L. Such concentrations occur approximately 31% of the time at the Don River Pottery Road site. While similar such calculations have not been made for other rivers in the Toronto area, with mean chloride concentrations of 135-220 mg/L (Table 4-1), such aquatic systems are likely to be chronically toxic to a significant fraction of the biota over major time intervals. Similarly, elevated chloride concentrations in contaminated springs (Williams et al. 1999), lakes such as Chocolate (Kelly et al. 1976), and ponds near highways (Watson 2000) are likely to exert chronic toxicity effects on aquatic communities.

It is instructive at the point in the review to consider various water quality criteria that have been developed for chloride and/or salinity. Such criteria provide for a further assessment of the concentrations at which chloride (or salinity) is likely to have adverse effects on the aquatic environment.

7.5 Water Quality Criteria for Chloride

Various agencies have developed water quality criteria for chloride in order to protect aquatic life from harm. These criteria also can be used in the assessment of the potential toxicity of chloride from road salts in the Canadian environment.

7.5.1 Birge et al. (1985)

Birge et al. (1985) recommended that, in order to protect aquatic life and its uses, for any consecutive 3-day period:

1) the average chloride concentration should not exceed 600 mg/L;

2) the maximum chloride concentration should not exceed 1,200 mg/L;

3) chloride concentrations may average between 600-1,200 mg/L for up to 48 hours.

The 1,200 mg/L value was determined from an investigation of benthic community structure and fish survivorship at 7 sites downstream of a salt seepage. Survivorship and diversity was lower at the 1,000 mg/L than the 100 mg/L site and further reduced at the 3,160 mg/L site. In the laboratory, toxicity studies determined a Final Acute Value of 760 mg/L chloride and a Final Chronic Value of 333 mg/L chloride.

7.5.2 United States Environmental Protection Agency (1988)

The USEPA (1988) has developed ambient water quality criteria for chloride. They concluded that except possibly where a locally important species is very sensitive, freshwater organisms and their uses should not be affected unacceptably if:

1) the 4-day average concentration of chloride, when associated with sodium, does not exceed 230 mg/L more than once every three years on average;

2) the 1-hour average chloride concentration does not exceed 860 mg/L more than once every three years on average.

They also noted:

1) these criteria will not be adequately protective when the chloride is associated with potassium, calcium, or magnesium;

2) because animals have a narrow range of acute sensitivities to chloride, excursions above this range might affect a substantial number of species.

The Criterion Maximum Concentration, 860 mg/L Cl^-, was obtained by dividing the Final Acute Value, 1,720 mg/L Cl^- by 2. The Final Chronic Value, 230 mg/L Cl^-, was obtained by dividing the Final Chronic Value by the Final Acute:Chronic Ratio, 7.594.

If these USEPA criteria were applied to Canadian waters, a number of situation urban creeks, rivers, and lakes would be above these recommended guidelines. Unfortunately, Canada has not developed a water quality guideline for the protection of aquatic life. Other guidelines have been developed as described below.

7.5.3 Canadian Water Quality Guidelines: Freshwaters

The Canadian Council of the Ministers of the Environment (1991) has developed a number of water quality guidelines, including for chloride, although none of these are for the protection of aquatic life. These guidelines are as follows:

1) For Canadian drinking water, chloride concentrations should not exceed 250 mg/L. This rationale is based on taste rather than human health considerations. Guidelines are identical in Quebec.

2) For irrigation waters, sensitive plants should not be irrigated with waters >100 mg/L Cl^- while tolerant plants can be irrigated with water up to 700 mg/L Cl^-. This guideline suggests that some sensitive wetland and aquatic plants would be adversely affected by growing in road salt contaminated waters at chloride concentrations as low as 100 mg/L. Quebec maintains identical guidelines.

3) Livestock can be safely watered with concentrations up to 1,000 mg/L Cl^-, however, at concentrations of 1,000-3,000 mg/L livestock health may become impaired. This guideline suggests that terrestrial animals obtaining their drinking water from streams, marshes, and ponds would have their health impaired at these chloride levels. Some animals may be even more sensitive to chloride (i.e., at concentrations <1,000 mg/L). There is no specified standard in Quebec.

4) In Quebec, aquatic life suffer acute toxicity at minimum chloride concentrations of 860 mg/L. Chronic toxicity occurs at chloride concentrations of 230 mg/L and any increases must not exceed 10 mg/L.

5) One of the goals of sustainable development is to manage natural resources so that they are not damaged for future and/or other uses. A number of industries, most notably the food and beverage industry, require the use of waters with chloride concentrations of <250 mg/L.

7.5.4 Canadian Water Quality Guidelines: Marine and Estuarine Life and Wildlife

The Canadian Council of the Ministers of the Environment (1991) has developed an interim water quality guideline for salinity (expressed as parts per thousand) for the protection of marine and estuarine life. Specifically:

> "Human activities should not cause the salinity (expressed as parts per thousand) of marine and estuarine waters to fluctuate by more than 10% of the natural salinity expected at that time and depth".

This guideline was developed from guidelines issued by other jurisdictions. Highlights of these are as follows:

1) For Alaska, the maximum allowable variation above natural salinity is 1 g/L if the natural salinity of the habitat lies between 0-3.5 g/L and 2 g/L if the natural salinity of the habitat lies between 3.5-13.5 g/L.

2) For estuarine waters in British Columbia, the maximum allowable salinity change is ±10% of the natural salinity. Furthermore, the maximum 24-hour change in salinity should not exceed 1 g/L if the habitat lies between 0-3.5 g/L.

3) The European Community requires that discharges affecting shellfish waters not cause the salinity of the receiving water body to be exceeded by more than 10%.

4) For the United States, no changes in channels, basin geometry of the area, or freshwater inflow should be made that would cause permanent changes in isohaline (salt concentration lines) of more than 10% of the natural variation. Furthermore, the maximum 24-hour change in salinity should not exceed 1 g/L if the natural salinity of the habitat lies between 0-3.5 g/L.

5) For Western Australia, water quality parameters should be maintained at pristine or ambient levels where applicable. For Class 1 (maximum protection) waters, unnatural influences should not change the seasonal mean salinity, measured preferably over 5 years, by more than 25% of the standard deviation, nor change the salinity beyond the range recorded over that period for waters requiring a minimum or high level of protection.

Seawater is predominately sodium and chloride. A 1 g/L change in salinity roughly corresponds to a chloride concentration of ca. 600 mg/L. Estuarine organisms are adapted to fairly wide variations in salinity. It is highly probable that freshwater organisms would have a considerably lower tolerance to elevated salinity (and chloride) levels.

7.6 General Conclusions

1) To date, there is limited evidence of road salt impacts on aquatic ecosystems in Canada. This is due primarily to the lack of study. Data which has been located suggests that areas in the Toronto, Ottawa, Montreal, Quebec corridor, parts of New Brunswick, Nova Scotia, central British Columbia, and Alberta may be particularly vulnerable to adverse impacts from road salt applications. Modeling studies by other ERG members support this conclusion.Documented chloride concentrations in snow melt, drainage ditches, storm sewers, and urban creeks in heavily developed urban areas reach concentrations that are toxic to aquatic life for exposures as short as one day. Concentrations from leaking road salt depots also reach toxic levels for exposures of 1 day or less.

2) Documented chloride concentrations in snow melt, drainage ditches, storm sewers, urban creeks and rivers in heavily developed urban areas, such as the Toronto area, can reach concentrations which will be toxic to some aquatic life for exposures of 4 days. While laboratory toxicity studies suggest that only 10-15% or less of creek populations may be affected, this is likely to be an underestimate because laboratory populations typically are considerably more tolerant that wild populations. Documented chloride concentrations from leaking road salt depots also reach these toxic levels.

3) Few studies have determined toxicity at 1-week exposures. LC_{50} values are lower (geometric mean 2,288 mg/L Cl^-) than at 4-days. Responses other than mortality begin to appear when cladocerans and early life history stages of fish are tested. EC_{50} values decline with a geometric mean of 1,718 mg/L Cl^-. Chloride concentrations in this toxicity range have been observed in urban creeks, although the duration is unknown. Concentrations in this range have been observed in wetlands near road salt depots and, in extreme examples, in shallow urban lakes.

4) Chronic toxicity was estimated from 4-day toxicity data and ranged from 200-1,875 mg/L Cl^- with a geometric mean of 578 mg/L Cl^-. Chloride concentrations in this range have been observed in urban creeks in the Toronto area; the duration is unknown. Concentrations in this range have been observed in wetlands near road salt depots and, in extreme examples, in shallow urban lakes. Changes in algal species composition have been associated with changes in chloride concentration in this range.

5) In the United States, a number of lakes in well-populated states have shown small increases in chloride concentrations, including lakes in rural areas. Recent studies involving phytoplankton, suggest that algal communities are responding to such increases that are small and below the levels considered toxic. Nevertheless, it is probable that small changes in salinity (and chloride) will exert

subtle impacts on aquatic ecosystem structure and productivity. A similar situation may be occurring in Canada, particularly in southern Ontario, Quebec, and the Maritimes.

6) Road salt applications are resulting in chloride concentrations in many Canadian aquatic environments that exceed the USEPA's criteria for chloride and the protection of aquatic life, many of Canada's water quality guidelines for chloride in freshwaters, and salinity guidelines established for the protection of estuarine and marine waters.

7.7 Conclusion

Based on the general conclusions outlined above and the earlier discussions in this report, road salt is viewed as CEPA toxic according to the definition of "having or that may have an immediate or long-term harmful effect on the environment". This conclusion is developed more fully in the last section of this report.

8.0 GENERAL DISCUSSION

8.1 A Brief Overview of the Roadway and the Human Environment

Today's driving population consists of a broad assemblage of drivers; from the high school student just learning how to operate a vehicle, to the elderly driver who is dependent on his/her vehicle for essentials, such as shopping and doctor's appointments; from the immigrant with limited driving experiences to the highly trained truck driver with hundreds of thousands of kilometers of driving experience; and from the vacation traveler enjoying a scenic drive through a mountain range, to the busy office worker driving on a multi-lane highway and conducting business on his/her cellular telephone.

It has been almost one hundred years since the commercially viable automobile was first designed. There has been a tremendous evolution in the automobile from a small, unreliable, low-speed vehicle initially driven by a small segment of our society, to a diversity of highly-reliable vehicles operated by all segments of society and frequently driven at speeds of 100 kilometers per hour and greater. The modern highway thus carries a vast multitude of drivers with different driving experiences, priorities, and reasons for being on the roadway. All must safely share these limited lanes of roadway space as they go about their travels.

The varieties of vehicles on the roadways also have increased immeasurably since Henry Ford's time. At any one time, the roadway is carrying passenger vehicles from compact to full-size sedans, all terrain vehicles, pickup trucks, and an increasing number of trucks towing two or three cargo containers. The nature of the cargo also has diversified. When the automobile was first designed, it was used primarily for carrying passengers and general goods, such as household effects, agricultural and forestry products. While the nature of products carried by the modern car has diversified, the greatest diversity has been with trucks: the modern truck now carries a myriad of manufactured equipment and products, including numerous items now considered dangerous goods.

As use of the automobile and related vehicles has increased, so has the safety of such usage. The earliest automobiles had relatively modest safety features by today's standards – a horn, lights, and an enclosed passenger compartment, to mention a few. However, as accidents, injuries, and deaths increased with automobile use, improved safety features were designed to minimize such mishaps. For example, vehicles were designed to include such features as collapsible steering columns, safety glass in windshields, improved braking systems, seatbelts, airbags, etc. Manufacturers continue to work towards improving the probability that drivers will avoid accidents and drivers and passengers will survive those that cannot be avoided. Safety features also have been designed into cargo containers to protect the public and environment from damage incurred by the accidental release of cargo onto the roadway and surrounding environment.

Governments work together in a variety of ways to protect the well-being of the driving public. The underpinnings of such protection are firmly based in legislation. Without such legislation, there will always be small segments of society who unknowingly or deliberately operate the vehicles in unsafe manners. Legislation also allows society to respond rapidly to new knowledge in ways in which to avoid societal and environmental harm. (One of the most notable examples is the speed with which airport security changed following a few highly publicized aircraft highjackings).

Laws are designed and enacted to regulate and enforce safe driving conditions. Roads and highways are designed to incorporate safe driving features, and legal speeding limits are posted: signs also are posted to warn of potential driving hazards ahead, such as curves or bumps in the road and lane mergers. Limits are proscribed on when cars can be passed and speed limits may be temporarily reduced during road construction.

The government also ensures that roadways are maintained in safe conditions. Damaged roadways are repaired, obstructions removed, and roadways upgraded as their carrying capacity is exceeded. Roads are ploughed to remove snow and abrasives applied to improve traction during cold, slippery conditions. In many regions of Canada, roads can require grooming from as early as October to as late as April. The actual nature and intensity of such grooming varies regionally, both on a provincial and a municipal scale. For example, road salt is applied more frequently to roads in Ontario than in Saskatchewan where abrasives are more widely used. Municipalities also give priority to roadways that are groomed.

Regulations are passed to govern how vehicles are to be operated. This is to protect the public and the environment from the adverse impacts of vehicle use. Driving regulations are well known to the public with a variety of prohibitions on driving speed, passing lanes, and direction and timing of traffic flow. The tolerance for driving under the influence of alcohol continues to be lowered. Every driver requires licensing with the license restricted to vehicle class. For professional truck drivers, there are limitations on the number of hours that can be driven daily, among other regulations. For many jurisdictions, automobiles must undergo safety inspections before the annual license plate can be renewed: commercially operated vehicles have even more stringent licensing inspection requirements. Inspections are designed to ensure that the vehicle can be operated safely, considering both human and environmental well being.

Regulations also have been developed to protect the environment against the adverse impacts of vehicle use. For example, over the last few decades, has been a shift from leaded to unleaded gas and catalytic converters are now a required feature of the modern vehicle. Regulations also have been passed on construction designs of fuel storage tanks. Fuel leaking from underground and surface storage tanks may contaminant the environment and pose other hazards. Recycling is now required for the disposal of waste oil, used tires, and other automobile parts. It is illegal for the homeowner to pour used oil down the sewer drain or on the country lane. The use of Freon gas in air conditioners is now highly regulated to prevent damage to the atmosphere. New regulations continue to be developed to protect the public and the environment from the adverse impacts of vehicle use, including roadway operations. Consideration is now being given to regulations based on the use of road salt under PSL2 of CEPA.

8.2 Road Salt Toxicity

The PSL2 assessment defines toxic as "having or that may have an immediate or long-term harmful effect on the environment". Road salt, specifically sodium chloride, but also including potassium, magnesium, and calcium chloride, is now being assessed in this supporting document for PSL2 toxicity. Ferrocyanide additives also are being considered under the road salt assessment, but are dealt with in a separate supporting document (Letts 2000 a, b).

In the course of preparing this literature review and assessment, many approaches were taken to determine the known extent of chloride contamination in the Canadian environment as a result of road salt application. Despite the known concerns with road salt application dating back to the 1970s, a surprisingly small number of research studies have been conducted on road salt in the environment, particularly in Canada. However, unpublished information is available from provincial and municipal agencies of road salt use. This information has been compiled and it has been determined that there are several areas in Canada where road salt is being applied relatively heavily (i.e., southern Ontario, Quebec, and the Maritimes). Other areas include Alberta and British Columbia.

Many agencies monitor water quality, including conductivity and chloride concentrations. The vast majority of this information has not been synthesized into formal publications. That information which has been synthesized (or obtained) portrays a broad characterization of elevated chloride levels in the Canadian environment which can be associated with road salt usage. Studies conducted in Quebec and the Maritimes tend to report relatively low elevations in chloride concentration (<500 mg/L Cl^-) in the aquatic environment. No studies were located reporting elevated chloride concentrations in Manitoba and Saskatchewan as a result of road salt usage; specific information on the impacts of road salt usage in Winnipeg was not located in preparing this supporting document. Similarly, Alberta has not reported significantly increased chloride concentrations in the environment as a result of road salt usage; concentrations in snowmelt settling ponds are monitored. Increases in chloride concentrations in major prairie rivers appear to be minor. In British Columbia, problems with road salt use appear to be limited to leakage from road salt storage sites and dumping road salt-contaminated snow into two lakes.

By far the greatest known impact of road salt usage on the aquatic environment occurs within streams, creeks, and small rivers in the broader Toronto area, including municipalities such as Waterloo and Burlington. It is not uncommon for streams to reach chloride concentrations of 1,000 mg/L and greater during winter when road salt is applied. As a consequence, concerns have been raised regarding the contamination of groundwater in this area. Given the rapidly expanding population along the northern shores of lakes Erie and Ontario and, to a lesser extent, along the St. Lawrence River and parts of the Maritimes, this situation can reasonably be expected to worsen over the upcoming decades.

In the course of this review, information was obtained on chloride concentrations in different types of environments and on the chronic and acute chloride toxicity in the environment. On the basis of this review, it was possible to develop a number of brief scenarios where the chlorides from road salt may constitute a clear and present hazard to the environment.

8.2.1 Road Salt Storage Sites

Chloride concentrations frequently are elevated in surface and groundwater near salt storage depots and patrol yards (Morin et al. 2000). Chloride concentration in leachate may reach 66,000 mg/L Cl^- and concentrations as high as 12,400 mg/L Cl^- have been reported for one bog contaminated by runoff from an improperly maintained depot. Chloride concentrations in this range can be acutely toxic to a wide variety of aquatic organisms within a matter of hours.

While the maintenance and design of road salt storage sites continues to be improved, it is also evident that the operation of such sites, unless they are closely regulated, does present a hazard to the environment. Leakage from such sites has severely contaminated groundwater and damaged bogs. In other instances, small lakes located near such storage sites have become meromictic. Although only the algal community has been studied in these case histories, it is highly likely that all components of the aquatic ecosystem were adversely impacted, including benthic invertebrates and fish. While only a few studies have been located documenting such instances, they are probably much more widespread. It is the lack of detailed environmental impact assessment studies around such facilities that limits our knowledge of such occurrences.

8.2.2 Snow Dump Melt

Snow collected from roadways to which road salt has been applied has reported chloride concentrations reaching up to 2,000-10,000 mg/L. Chloride concentrations in this range can be toxic to a variety of organisms at exposure levels of one day or less (section 7; Snodgrass et al. 2000). In order to protect the aquatic and terrestrial environment, safe practices must be used in disposing of contaminated snow. Many of the larger municipalities have developed procedures to minimize environmental damage from snow dumping. Edmonton, for example, has constructed specific containment areas to accumulate snowmelt and to allow for the sedimentation of particulates, including associated organic and inorganic contaminants. Montreal and Winnipeg also limit where snow can be dumped.

However, it is not known whether all jurisdictions are using safe practices in snow dumping. It seems likely that, without regulation, situations are or will be occurring where environmentally unsafe practices are being followed, most likely unknowingly. Smaller municipalities, with more limited resources, may be particularly vulnerable to such procedures. There is a need for regulations to limit the environments into which chloride-contaminated snow is dumped. For example, chloride-contaminated snow should not be dumped into wetland areas, on elevated areas over looking small streams, nor routinely dumped on areas overlooking small lakes with limited water exchange. Small, relatively deep lakes are especially vulnerable to chloride contamination and the formation of meromixis. Road salt contaminated snow also should not be dumped near groundwater recharge areas. Since roadway snow contains a variety of inorganic and organic contaminants, including particulates, regulations should be developed to apply to the dumping of all snow, regardless of whether it contains road salt.

8.2.3 Weakly Diluted Highway Runoff

Undiluted highway runoff, like snow dump runoff, can reach concentrations of 2,000-10,000 mg/L Cl^-. Such concentrations can be acutely toxic to a variety of organisms that live near these concentrated chloride sources. The actual area affected is a function of the volume of weakly diluted highway runoff relative to the size and dilution capacity of the terrestrial/aquatic environment.

To date, the only examples of chloride concentrations in the 2,000-10,000 mg/L range that were located, were confined to the greater Toronto area and to similar regions of dense populations and highways in the United States. However, because chloride levels in runoff have not been routinely monitored in the vast majority of jurisdictions, it is highly probable that chloride concentrations in highway runoff in the 2,000-10,000 mg/L are more commonly encountered than our brief review of the literature would suggest. Such elevated chloride concentrations are dependent upon the amount of road salt the operator applies to the road, the amount of snowmelt immediately following this application, and to meteorological conditions immediately following road salt dispersal.

While road salt running off the roadway surface enters an artificial environment inhabited by the most hardy of organisms and possibly of limited concern from a regulatory viewpoint, it also begins to transfer to the soil environment as snowmelt percolates downwards. In some senses, the roadway can be viewed as a semi-permeable wound in the terrestrial landscape, with the highway edge the opening through which “infections” (environmental damage) begin to gain a foothold. Sodium in the road salt replaces calcium in the anion exchange and the soil becomes depleted of calcium and magnesium (Jones and Jeffrey 1986). The soil then may become alkaline reaching pH values as high as 9 or 10. Inorganic and organic colloids become dispersed at these higher pH values and move down the soil profile. However, organic matter and salt can move towards the surface with water evaporation. The soil surface may be darkened by the organic matter or whitened by the salt. The occurrence of road salt-whitened soil (and vegetation) appears to be a common occurrence in the Maritimes and presumably elsewhere where road salt is applied. Road salt apparently occurs in sufficient concentrations in these environments to serve as an attractant to moose, deer, and other wildlife (see Section 4.11; Cain et al. 2000). During spring and summer rainfall, the upper layer of organic matter and salt may be leached from the soil, improving the quality of the roadside soil, but potentially damaging other environments.

Sodium chloride percolating through soils may enhance the mobility of trace elements, such as lead and cadmium. Amrhein and Strong (1990) reported enhanced mobility at 5,640 mg/L NaCl (3,545 mg/L Cl^-), but not at 564 mg/L NaCl (355 mg/L Cl^-), average runoff concentrations. They also noted that dilute solutions of sodium chloride and pure snowmelt are likely to mobilize metals through the process of organic matter solubilization and clay dispersion. In a later study, Amrhein et al. (1992) reported that increasing concentrations of sodium chloride increased the leaching of chromium, lead, nickel, iron, cadmium, and copper from road side soils that were contaminated by runoff from roads that were subject to heavy traffic and road salt application. Copper, nickel, and iron concentrations often exceeded U. S. criteria for the protection of freshwater aquatic life. While they argued that toxicity should be reduced by the complexation of these metals with organic matter, it is also possible this would be counterbalanced by synergistic reactions. In a later study, (Amrhein et al. 1993) concluded that roadside soils contaminated by sodium chloride from deicing operations could be contributing to the mobility of trace metals to groundwater through colloid-assisted transport.

While metals entering the environment from the roadway and from transformations within the road soil may have toxic effects at elevated concentrations, they also may have growth enhancing effects at lower concentrations. Iron and copper are essential minerals to the well being of most living organisms, including algae, at the base of the food web. Minerals such as calcium, magnesium, and potassium also are essential to the well being of most organisms. Highway runoff thus may promote the availability of minerals to downstream aquatic communities. For example, Pugh et al. (1996) reported an increase in the availability of these nutrients to a peat bog that was contaminated by road salt runoff. The bog ran parallel to the Interstate Highway 95 over a distance of ca. 1 km. Birch Stream was located immediately east of the bog and at an approximately 1-2 m lower elevation than the highway. Approximately 90% of the road salt applied to the highway is transported into this stream each spring.

Weakly diluted highway runoff can thus be viewed as a solution that is potentially toxic to a variety of organisms following exposure times of hours to a few days. Toxicity will be due not only to chloride salts, but to the myriad of other contaminants carried in the runoff – metals, PAHs, suspended solids, oils, and greases. Runoff percolating into roadside soil can initiate a variety of physical and chemical transformations, which can ultimately impact roadside vegetation as the pH and various chemical properties of the soil, are transformed. This is dealt with more fully in Morin et al. (2000) and Cain et al. (2000). In addition, aquatic and terrestrial ecosystems can be impacted as later runoff (e.g., spring and summer rains) transport metals, other ions, organics and clays into these downstream systems.

Agencies are developing procedures to reduce the amount of chloride salts applied to roadways. Lower concentrations of chloride salts are being applied now compared to 10 or 20 years ago. Nevertheless, various circumstances can occur which may result in highly concentrated chloride salts in runoff. This would include vegetation along the roadway, and small water-bodies with weak dilution capacity (e.g., ditches; small, weakly flowing creeks; and wetlands). There is a need to develop improved environmental regulations to protect such environments from the adverse constituents contained in roadway runoff, including road salt. Engineers who design highways must take into account features such as runoff and the carrying capacity of the landscape on either side of the roadway. This information, along with more detailed environmental information on features such as wetlands, small streams, bogs, groundwater recharge areas can be incorporated into Geographic Information Systems (GIS) to identify areas which are particularly sensitive to damage from highway runoff, including road salt. Mitigative actions can be taken, including channeling highway runoff into storm water collectors, or impermeable drains, for disposal into less vulnerable areas.

8.2.4 Moderately Diluted Roadway Runoff

Relatively few studies were located which reported elevated concentrations of chloride in the Canadian environment as a result of moderately diluted road salt runoff. This probably is due to the fact that highway runoff entering most creeks and streams is, in fact, rapidly diluted. It also is due to the fact that there have been few studies specifically investigating chloride concentrations in those streams, creeks, and small rivers which are vulnerable to significantly elevated chloride concentrations. Examples of such vulnerable ecosystems are creeks and wetlands in areas where road salt is being heavily applied and where the creek runs parallel to the highway for a significant distance. In addition, small creeks and wetlands at the base of a long stretch of highway can reasonably be expected to be impacted by highway runoff; such creeks and wetlands will have an extensive highway catchment area relatively to the entry point of the runoff into those aquatic ecosystems. Creeks, streams, and small rivers in urban areas also could be expected to receive significant quantities of moderately diluted roadway runoff.

Numerous studies conducted in creeks, streams, and small rivers in the Toronto area, have shown that chloride pulses of 1,000 g/L and greater commonly occur during the winter months. Concentrations at this level are likely to be stressful to aquatic organisms, particularly if such pulses last for two or three days. Unfortunately, there is little information on the duration of such pulses. However, based on the frequency at which elevated chloride concentrations have been observed in the Don River (Schroeder and Solomon 1999), chloride pulses may be acutely toxic to a significant fraction (10% or more) of organisms in Toronto area creeks during various periods of winter road salt application. These toxicity estimates are based on laboratory studies involving hardy organisms that are readily maintained in culture. As previously noted, these organisms are in excellent condition being maintained at optimum temperatures with a constant food supply. In the real world, organisms experience many stresses including food limitation, fluctuations in temperature, etc. In addition to sodium chloride and other chloride salts, highway runoff contains many potential toxicants, including metals, PAHs, oils and greases, and suspended solids. These toxicants may act synergistically, enhancing sodium chloride toxicity.

The impacts of elevated chloride concentrations on creeks, streams, and small rivers are poorly understood. This is because few studies have been conducted in these waters, particularly investigating the impact of road salt. The most likely impacts include some mortality for immobile or trapped organisms, movement of organisms away from such runoff through increased benthic drift and active swimming, and some changes in the periphyton and microbial communities. As previously noted, highway runoff contains a variety of toxicants including metals, organics, oils, greases, and suspended solids. Suspended solids eventually settle in depositional areas and have a more direct and permanent impact on the stream, wetland, and pond community. Such effects occur through the smothering and toxic effects of metals, organics, and other contaminants. Contaminants associated with fine silts and clays may become very concentrated in such depositional areas. While the toxicity of some metals may decrease with increasing salinity (Hall and Anderson 1995), increased sodium and chloride concentrations in runoff facilitates the entry of these metals into the aquatic environment.

Moderately dilute highway runoff containing elevated chloride concentrations has been shown to affect a number of urban lakes. The most common phenomenon has been the development of meromixis and the formation of oxygen-depleted bottom waters. Such conditions are stressful to bottom living organisms. Furthermore, such conditions can enhance the release of toxic metals from the sediments and algal-bloom stimulating nutrients, such as phosphorus. Such lakes also are undoubtedly impacted by increased loadings of organics, metals, and particulates. Chloride concentrations can increase rapidly in lakes that become incorporated into the urban landscape as the recent study of Bridgeman et al (2000) suggests.

In a recent literature review, Buckler and Granato (1999) assessed the known biological effects of highway runoff constituents for the U. S. Department of the Interior. They noted that, while highway runoff usually may not be toxic, tissue analyses and community assessments do indicate effects in sediments that have accumulated near highway discharge points, including low traffic sites. While this review contained relatively few studies of road salt runoff, it does support the contention that highway runoff is damaging to the environment and that road salt will enhance this damage through direct (chloride toxicity) and indirect (metal mobilization, synergistic reactions, chronic impacts in depositional areas) actions.

It may be argued that periodic pulses of chloride to streams, creeks and lakes in urban areas should be of little environmental concern because such waters already are perturbed ecosystems. However, this is a weak argument. It was not that long ago that London, England was blanketed in smog from coal burning and Toronto is now beginning to experience its own smog problems. Regulations have been and continue to be developed to restore and maintain air quality. Lakes, such as Erie and Ontario, were experiencing rapidly accelerating eutrophication because of the dense human population along the shores and the massive release of nutrients into these lakes from direct sources such as sewage and from diffuse sources such as agricultural lands. Regulations have and continue to be developed to restore and maintain water quality. Increased concentrations of total dissolved solids such as calcium, sulfate, and chloride (Beeton 1969) also were viewed with concern and action taken to reduce these inputs.

Many harbors in the Great Lakes were severely contaminated by metals, organics, and nutrients from industrial and urban activities that began in the mid- to late 1800s and continue to this day. These conditions were not accepted either from a human health or an environmental viewpoint and remedial actions were taken to protect and restore these systems. Many regions around the Great Lakes have remedial action plans for restoring and protecting these systems. Furthermore, regulations exist for the protection of urban creeks and streams: they are not extended parts of the sewer and storm water systems. The current issue is whether this protection is to be extended to include regulations on the use and subsequent release of road salt from the roadway surface into the terrestrial and aquatic environment.

Urban areas such as Toronto have included in their planning various provisions to protect urban creeks, streams, and lakes. Parkways often have been developed along these waterways to provide the public with a natural haven from the hectic and artificial features of the urban landscape. It is of vital importance to the emotional well being of urban society that such havens continue to be present and that they be healthy. There is a growing expectation that such creeks, streams, rivers and lakes be as similar as possible in their features as those found in the natural countryside, before these creeks, streams, rivers and lakes were incorporated into the urban landscape. Included in such features are waters free of toxics and nuisance algal growths and a healthy, self-maintaining population of invertebrates and fish. Other features include healthy avian, reptilian, and mammalian communities.

Artificially created urban lakes and storm water detention ponds are in a somewhat different category of aquatic ecosystems. However, the intent of an urban lake is to mimic the natural world, as much as possible, and most certainly with respect to ecosystem health. Stormwater detention ponds, while created to retain runoff surges, are often located in somewhat natural settings and are providing habitat for a variety of wildlife (Bishop et al. 2000a and b). These habitats are becoming of increasing importance as the urban landscape and agricultural industry continue to expand, removing ponds, wetlands, and creeks from the country landscape. It is essential that, as urban and agricultural areas expand, they embrace and not engulf the wetlands, creeks, rivers, and ponds in these areas. Thus, creeks, small rivers, streams, wetlands, and small lakes located near highways need to be protected from environmental degradation by runoff from the roadways and waterways already incorporated into the urban and agricultural landscape also merit protection against further degradation.

Healthy creeks, rivers and lakes in urban areas serve many important functions. For the human population, there are recreational, educational, and philosophical benefits. There also are clear economic benefits as witnessed by the conversion of contaminated industrial harbor areas into very popular recreational areas characterized by walkways along the water course, restaurants, shops, and opportunities for boating and even fishing. Increasingly, environmental education is being incorporated into such regions, informing the general public of the natural features and value of their environment.

Urban wetlands, creeks and streams serve other functions. Most fish spawn in shallow waters which tend to warm more quickly in the spring than the offshore regions of lakes and larger rivers. Food sources may be more abundant in such systems and fish fry grow at a more rapid rate than in colder, larger waters. Moreover, shallow waters provide a refuge from large fish that tend to be found in deeper waters. Some lake dwelling fish spawn in the nearshore or littoral zone of lakes, but many migrate into creeks and small rivers to spawn. Fish inhabiting larger rivers may also spawn in the smaller tributaries. In the marine world, East and West Coast salmonids migrate into rivers and then up into decreasingly smaller creeks to spawn. Eggs laid in the fall overwinter in a dormant stage, completing their development in late winter. Conversely, eels spawn in the marine environment; young eels, called the glass-eel stage, migrate into freshwaters to complete their development to adulthood.

As cities grow in size, incorporating more and more wetlands, streams, creeks, and lakes into their boundaries, it is of vital importance that the health of these ecosystems be protected. Without such protection, it is likely that important spawning creeks for lake and river fish populations will be lost. A recent study by Bradford and Irvine (2000) related major declines in coho salmon spawning in the Thompson River (British Columbia) watershed to a decline in salmonid productivity due to changing ocean conditions, overfishing and freshwater habitat alteration. Within the freshwater habitats, the decline in salmonid abundance was related to the proportion of agricultural land use ($r = -0.50$), road density ($r = -0.39$), and a qualitative measure of stream habitat status ($r = -0.44$), but not to logging ($r = 0.02$). While the use of road salts was not mentioned in this paper, the study does suggest that the increased presence of roads in an area can be one of many factors adversely affecting aquatic communities.

Based on these various considerations, it seems reasonable to conclude that moderately dilute highway runoff containing chloride at concentrations of ca. 1,000 mg/L from road salt running off highways, leaching from storage depots and patrol yards, and melting from snow dumps does have a harmful effect on the environment. However, this environment will be localized and the events may be periodic. The notable exceptions will be wetland areas and small lakes and ponds, particularly those with poor water exchange. Regulations need to be developed to ensure that the toxicity of highway runoff to the aquatic environment is minimized and that vulnerable habitats are given special protection.

8.2.5 Highly Diluted Roadway Runoff

The impacts of chloride concentrations on aquatic ecosystems at concentrations below 1,000 mg/L are not well understood. It is known that, in some instances, lakes may become meromictic. Deep-waters then may become anoxic and the release of metals and nutrients from the sediment-water interface is accelerated. Such occurrences have been limited to shallow, urban lakes or to small lakes located near road salt storage depots. Studies conducted in the United States are determining that chloride concentrations are gradually increasing in many lakes, particularly in lakes located near highways that are subject to road salt applications.

Chloride, potassium, magnesium, sodium, and calcium are essential elements to the well being of organisms. Such elements serve many important biochemical roles with sodium and chloride being of especial importance in the organism’s ability to maintain an ideal osmotic pressure within their cells. It is well known that some organisms are adapted to seawaters, other organisms to freshwaters, and still others to brackish waters or to saline lakes. Within freshwaters, it is highly probable that some species are adapted for extremely dilute waters while others require higher concentrations of total dissolved solids, including chloride salts. Certainly, within the algal community, species occurrences can be related to hard versus soft waters just as species occurrences can be related to pH and phosphorus concentrations. Phytoplankton ecologists are just now starting to develop estimates of chloride optima for algae just as previous researchers have developed nutrient and temperature optima. Furthermore, on the basis of such studies, they are able to infer changes in algal communities that appear to be related to increasing chloride concentrations, especially in lakes located near highways subject to road salt application. The actual mechanism of change has not been determined, but does not appear to be solely related to phosphorus. Mechanisms could be associated with chloride optima and/or to other factors associated with increased chloride (e.g., increased movement of trace elements and organics). Whatever the mechanism(s), it is probable that the many small creeks and streams feeding these lakes have been even more strongly impacted than the lakes themselves.

If “harm” is defined as a change from natural conditions and, if the results of the United States limnology studies are assumed to apply to similar regions in Canada, it is reasonable to conclude that road salt may have “… a long-term harmful effect on the environment”. That is, there are probably many aquatic environments in Canada where small increases in chloride concentrations from road salt are having measurable impacts aquatic community composition and productivity. Increased productivity may be driven by the sodium chloride itself and/or by the increased movement of nutrients and essential trace elements from road side and streamside soils into the system. As chloride concentrations increase, unknown optima will be reached in diversity and production. Further increases in salinity will result in the osmotic stress to freshwater organisms adapted to the low salinity environment of most inland waters. The concentration at which salinity begins to become stressful to most freshwater organisms is not well characterized, but is believed to occur somewhere between 500-1,000 mg/L salinity (303-606 mg/L Cl^-). More sensitive species may be affected at even lower concentrations.

8.3 Road Salt and Water Quality Criteria for Chloride

As noted in Section 7, various agencies have developed water quality criteria for chloride in order to protect aquatic life (from harm). Specifically, in the United States, Birge et al. (1985) recommended that, in order to protect aquatic life and its uses, for any consecutive 3-day period:

1) the average chloride concentration should not exceed 600 mg/L;

2) the maximum chloride concentration should not exceed 1,200 mg/L;

3) chloride concentrations may average between 600-1,200 mg/L for up to 48 hours.

In addition, the United States Environmental Protection Agency (1988) has developed ambient water quality criteria for chloride. They concluded that except possibly where a locally important species is very sensitive, freshwater organisms and their uses should not be affected unacceptably if:

1) the 4-day average concentration of chloride, when associated with sodium, does not exceed 230 mg/L more than once every three years on average;

2) the 1-hour average chloride concentration does not exceed 860 mg/L more than once every three years on average.

They also noted:

3) these criteria will not be adequately protective when the chloride is associated with potassium, calcium, or magnesium;

4) because animals have a narrow range of acute sensitivities to chloride, excursions above this range might affect a substantial number of species.

Moreover, the Canadian Council of the Ministers of the Environment (1991) has developed a number of water quality guidelines, including for chloride, although none of these are for the protection of aquatic life. These guidelines are as follows:

1) for Canadian drinking water, chloride concentrations should not exceed 250 mg/L;

2) for irrigation waters, sensitive plants should not be irrigated with waters > 100 mg/L while tolerant plants can be irrigated with water up to 700 mg/L;

3) live stock can be safely watered with chloride concentration up to 1,000 mg/L. However, at concentrations of 1,000-3,000 mg/L, livestock health may be impaired.

The Canadian Council of the Ministers of the Environment (1991) also has developed an interim water quality guideline for salinity (expressed as parts per thousand) for the protection of marine and estuarine life. This guideline was developed from guidelines issued by other jurisdictions that emphasize the concern with relatively small excursions in salinity above (or below) the natural salinity for that water body of concern. Specifically:

- human activities should not cause the salinity (expressed as parts per thousand) of marine and estuarine waters to fluctuate by more than 10% of the natural salinity expected at that time and depth.

As previously noted, a 1 g/L change in salinity roughly corresponds to a chloride concentration of ca. 600 mg/L. Estuarine organisms are adapted to fairly wide variations in salinity. It is highly probable that freshwater organisms would have a considerable lower tolerance to elevated salinity and chloride levels.

If the USEPA (1988) ambient water quality criteria for chloride and the CCME's interim salinity guideline for marine and estuarine waters were applied to Canada's aquatic environments, there are documented situations in which road salt storage and use results in chloride concentrations which exceed those guidelines. This is of some concern. In order to reduce the likelihood that road salt usage will result in elevated chloride concentrations in the environment, its use and release needs to be regulated.

8.4 Conclusion

Road salt applications are resulting in chloride concentrations in many Canadian aquatic environments that exceed the USEPA's criteria for chloride and the protection of aquatic life, many of Canada's water quality guidelines for chloride in freshwaters, and salinity guidelines established for the protection of estuarine and marine waters. Therefore, according to this statement and earlier discussions in this report, road salt is viewed as CEPA toxic according to the definition of "having or that may have an immediate or long-term harmful effect on the environment".

Road salt has an obvious benefit to the safety of the motorist preventing roadway accidents. Human life and well-being are protected, as is the environment, from accidental releases of toxic compounds carried by trucks and other vehicles. Nevertheless, like other aspects of roadway use, there is a growing requirement for improved human and environmental protection.

Manufacturers are continuing to design safer and safer vehicles and legislation continues to be passed regulating the use of these vehicles, including their waste products. Highway departments continue to build safer roads. While road salt undoubtedly will continue to be used for many years, and possibly for decades to come, it can be used more safely.

Highway departments are continuing to develop improved methods for applying road salt and to reduce the amount of salt that is being used. Storm-water and snow-dump detention ponds serve many important functions including reducing the rates and amounts at which toxic materials are released into the environment. Highway engineers routinely consider features such as highway runoff and the surrounding landscape in designing highways that will carry traffic. Roadways must drain readily of rainfall and snowmelt and must have some permanency (i.e., the road bed must be stable). The technology also exists through GIS applications to incorporate environmental concerns into highway construction and maintenance. With respect to highway runoff, environmentally sensitive environments can be identified and special structures designed to protect them highway runoff.

While there are many reasons why road salt and their additives are required on highways, one central reason is that the various segments of the driving population expect to drive at, or above, the speed limit during virtually all weather conditions. Thus, safety features may not be as protective as initially expected. For example, some drivers of vehicles equipped with an antilock braking systems believe that there no-longer is a need to slow down in poor driving conditions. People driving in fog or other conditions of poor visibility often have a hard time judging their speed of travel and tend to drive faster. Thus, while highway agencies routinely use road salt to provide for a safe driving surface during snowfall, these agencies and legislators may wish to consider an alternative, such as the use of variable speed limits for given roadways which can be changed in a matter of minutes. For example, while the posted speed on the QEW in Toronto is 100 km/h, the posting of a variable speed limit would immediately allow the highway/police department to reduce the legal speed limit to 50 km/h (or lower) should road conditions deteriorate. Speed limits could be reduced should visibility suddenly decline, a snowstorm become eminent, etc. Airports routinely adjust landing schedules with changes in weather in the interest of public safety.

Meteorological agencies continue to provide increasing spatially detailed coverage of local weather conditions. This information could be submitted to highway control. In addition, monitoring stations could be established along the roadway at various locations to collect information on road condition, visibility, and traffic. All of this information could be combined with the appropriate software to identify stretches of the roadway where it is no longer possible for the average motorist and cargo-towing truck to safely drive the maximum speed limit. The speed limit could then be immediately lowered through the use of electronic speed limit and other warning signs. On the short term, this would provide further protection to the public from accidents occurring from motorists exceeding the safe speed limit for that roadway under those conditions. On the long-term, it could reduce some of the use of road salt, providing further protection to the environment.

There are additional environmental requirements with respect road salt and the environment. First, existing water quality monitoring data needs to be synthesized to better characterize the impact of road salt on chloride levels in various compartments of the aquatic environment. Second, studies need to be conducted to assess the environmental status of streams and other aquatic environments that are subject to significant highway runoff, particularly those contaminated by road salt. As part of such studies, factors affecting the aquatic communities need to be assessed (e.g., chlorides, metals, PAHs, particulates). There also is the need to assess lakes in regions where highway development is significant to determine whether chloride levels are increasing. Finally, mesocosm studies should be conducted to assess the impact of road salt of aquatic communities, both for short pulses of highly elevated runoff and more gradual, but persistent low-level additions.

LITERATURE CITED

Abegg, R. 1949. Some implications from the changes in the mucous covering of fish subjected to salt solutions. Proc. Louisiana Acad. Sci. 12: 7-10. (In Doudoroff and Katz 1953).

Abegg, R. 1950. Some effects of inorganic salts on the blood specific gravity and tissue fluids of the bluegill, *Lepomis macrochirus* Raf. Physiol. Zool. 23: 124-134.

Adelman, I.R., L.L.J. Smith, and G.D. Siesennop. 1976. Acute toxicity of sodium chloride, pentachlorophenol, guthion, and hexavalent chromium to fathead minnows (*Pimephales promelas*) and goldfish (*Carassius auratus*). J. Fish. Res. Board. Can. 33: 203-208.

Amrhein, C., and J.E. Strong. 1990. The effect of deicing salts on trace metal mobility in roadside soils. J. Environ. Qual. 19:765-772.

Amrhein, C., J.E. Strong, and P.A. Mosher. 1992. Effect of deicing salts on metal and organic matter mobilization in roadside soils. Environ. Sci. Technol. 26:703-709.

Amrhein, C., P.A. Mosher, and J.E. Strong. 1993. Colloid-assisted transport of trace metals in roadside soils receiving deicing salts. Soil Sci. Soc. Am. J. 57:1212-1217.

Anderson, B.G. 1946. The toxicity thresholds of various sodium salts determined by the use of *Daphnia magna*. Sewage Works 18: 82-87.

Anderson, B.G. 1948. The apparent thresholds of toxicity to *Daphnia magna* for chlorides of various metals when added to Lake Erie Water. Trans. Amer. Fish. Soc. 78: 96-113.

Anderson, B.G., D.C. Chandler, T.F. Andrews. and W.J. Jahoda. 1948. The evaluation of aquatic invertebrates as assay organisms for the determination of the toxicity of industrial wastes. Amer. Petroleum Inst. Project Final Report 51. (In McKee and Wolf 1963).

Anonymous. 1960. The sensitivity of aquatic life to certain chemicals commonly found in industrial wastes. Acad. of Natural Sciences, Philadelphia. (In McKee and Wolf 1963).

Arambasic, M.B., S. Bjelic and G. Subakov. 1995. Acute toxicity of heavy metals (copper, lead, zinc), phenol and sodium on *Allium cepa* L., *Lepidium sativum* L. and *Daphnia magna* St.: comparative investigations and the practical applications. Water Research 29: 497-503.

Arp, P.A. 2001. Electrical conductivity and chloride ontents of stream, pond and ditchwater locations near a salt depot: a case study. Faculty of Forestry and Environmental Management, University of New Brunswick, Fredericton, New Brunswick. 14pp.

Arsenault, R.A., T.L. Pollock, S. Ratnam, and D. Hansen. 1985. Surface water quality study in the Waterford River Basin. Technical Report No. UHS-WRB1.4 Urban Hydrology Study of the Waterford River Basin. 91 pp.

August, L.L., and T.A. Graupensperger. 1989. Impacts of highway deicing programs on groundwater and surface water in Maryland. pp. 37-394. In, Groundwater issues and solutions in the Potomac River Basin, Chesapeake Bay Region. National Water Well Association Conference, May 14-16, 1989, Washington, D.C.

Beak International Inc. 1999. Ecotoxicity test results. Unpublished report for M. S. Evans.

Beeton, A.M. 1969. Changes in the environment and the biota of the Great Lakes. Pages 150-187 in Eutrophication: Causes, Consequences, Correctives. Printing and Publishing Office National Academy of Sciences, Washington, D.C.

Biesinger, K.E. and G.M. Christensen. 1972. Effects of various metals on survival, growth, reproduction, and metabolism of *Daphnia magna*. J. Fish. Res. Board Can. 29: 1691-1700.

Birge, W.J., J.A. Black, A.G. Westerman, T.M. Short, S.B. Taylor, D.M. Bruser, and E.D. Wallingford. 1985. Recommendations on numerical values for regulating iron and chloride concentrations for the purpose of protecting warmwater species of aquatic life in the Commonwealth of Kentucky. Memorandum of Agreement No. 5429. Kentucky Natural Resources and Environmental Protection Cabinet.

Birkeland, P.W. and E.E. Larson. 1978. Putnam's Geology. Oxford University Press, New York 659 pp.

Bishop, C.A., J. Struger, D.R. Barton, L.J. Shirose, L. Dunn, A.L. Lang, and D. Shepherd. 2000a. Contamination and wildlife communities in stormwater detention ponds in Guelph and the Greater Toronto Area, Ontario (1997, 1998). Part 1. Wildlife Communities. Unpublished manuscript. Canadian Wildlife Service, Burlington.

Bishop, C. A., J. Struger, L.J. Shirose, L. Dunn, G.D. Campbell. 2000b. Contamination and wildlife communities in stormwater detention ponds in Guelph and the Greater Toronto Area, Ontario (1997, 1998). Part 1. Contamination and biological effects of contamination. Unpublished manuscript. Canadian Wildlife Service, Burlington.

Black, H.H. 1950. Federal industrial pollution studies. Sewage and Industrial Wastes 22: 1049-1053. (In Hanes et al. 1970).

Blinn, D.W., M. Hurley, and L. Brokaw. 1981. The effects of saline seeps and restricted light upon the seasonal dynamics of phytoplankton communities within a southwestern (USA) desert canyon stream. Archiv. fur Hydrobiol. 92: 287-305.

Borgmann, U. 1996. Systematic analysis of aqueous ion requirements of *Hyallela azteca* - A standard artificial medium including the essential bromide ion. Arch. Environ. Contam. Toxicol. 30: 356 - 363.

Boucher, R.J. 1982. Penjajawoc Stream Salt Study. Maine Dept. of Transportation, Materials and Research Division Technical Paper 82-18. 15 pp.

Bowser, C.J. 1992. Groundwater pathways for chloride pollution of lakes. pp. 283-301. In F.M. D'Itri, ed. Chemical Deicers and the Environment. Lewis Publishers, Florida.

Bradford, M.J., and J.R. Irvine. 2000. Land use, fishing, climate change, and the decline of Thompson River, British Columbia, coho salmon. Can. J. Fish. Aquat. Sci. 57:13-16.

Bridgeman, T.B., C.D. Wallace, G.S. Carter, R. Carvajal, L.C. Schiesari, S. Aslam, E. Cloyd, D. Elder, A. Field, K.L. Schulz, P.M. Yurista and G.W. Kling. 2000. A limnological survey of Third Sister Lake, Michigan with historical comparisons. Lake and Reserv. Manage. 16(4):253-267.

Bubeck, R.C. and R.S. Burton. 1987. Changes in chloride concentrations, mixing patterns and stratification characteristics of Irondequoit Bay, Monroe County, New York, after decreased use of road-deicing salt, 1974-1984. New York Dept. of the Interior, U.S. Geological Survey 87-4223. 52 pp.

Bubeck, R.C., W.H. Diment, B.L. Deck, A.L. Baldwin, and S.D. Lipton. 1971. Runoff of deicing salt: effect on Irondequoit Bay, Rochester, New York. Science 172: 1128-1132.

Bubeck, R.C., W.W. Staubitz, A.D. Wediemann, and L.P. Spittal. 1995. Physical and chemical limnology of Ides Cove near Rochester, New York, 1970-1982. U. S. Geol. Surv. Water Resources Investigations Report 92-4052.

Buckler, D.R. and G.E. Granato. 1999. Assessing biological effects from highway-runoff constituents. U. S. Department of the Interior, U. S. Geological Survey. Northborough, Mass. 45 pp.

Cain, N.P., B. Hale, E. Berkelaar, and D. Morin. 2000. Critical review of effects of NaCl and other road salts on terrestrial vegetation in Canada. Report submitted to the Environment Canada CEPA Priority Substances List Environmental Resource Group on Road Salts. March 2000. Environment Canada, Commercial Chemicals Evaluation Branch, Ottawa, Ontario.

Cairns, J., Jr. and A. Scheier. 1958. The effects of temperature and hardness of water upon the toxicity of zinc to the pond snail, *Physa heterostropha* (Say). Notulae Naturae Acad. Nat. Sci. No. 308: 1-11.

Cairns, J., Jr. and A. Scheier. 1959. The relationship of bluegill sunfish body size to tolerance for some common chemicals. Proc. Ind. Waste Conf. Purdue Univ. 13: 243-252. (In USEPA 1988).

Camargo, J.A. and J.V. Tarazona. 1991. Short-term toxicity of fluoride ion (F-) in soft water to rainbow trout and brown trout. Chemosphere 22 (5-6): 605-611.

Canadian Council of the Ministers of the Environment (CCME). 1991. Canadian Water Quality Guidelines. Environment Canada, Ottawa.

Champagne, D.M. 1978. Impact of highway deicing salts on rural stream water quality. Transportation Research Record 647: 47-51.

Chan, C.H. and A.H. Clignett. 1978. Short-term variation of the chemical composition of the Niagara River. Environment Canada Inland Waters Directorate, Ontario Region. 30 pp.

CH2M Hill Engineering Ltd. 1993. A study of dust-suppressants in Ontario. Ontario Ministry of Environment and Energy. Volume 3. 25 pp.

Cherkauer, D.S. 1975. Urbanization impact on water quality during a flood in small watersheds. Water Resources Bulletin 11: 987-998.

Cherkauer, D.S. and N.A. Ostenso. 1976. The effect of salt on small artificial lakes. Water Res. Bull. 12(6): 1259–1266.

Cherkauer, D.S. 1977. Effects of urban lakes on quantity and quality of baseflow. Water Resources Bull. 13: 1119-1130.

Chételat, J., F.R. Pick, A. Morin, and P.B. Hamilton. 1999. Periphyton biomass and community composition in rivers of different nutrient status. Can. J. Fish. Aquat. Sci. 56: 560-569.

Chow-Fraser, P. 1991. Use of morphoedaphic index to predict nutrient status and algal biomass in some Canadian Lakes. Can. J. Fish. Aquat. Sci. 48:1909-1918.

Colburn, E.A. 1988. Factors influencing species diversity in saline waters of Death Valley, USA. Hydrobiol. 158: 215-226.

Cowgill, U.M. and D.P. Milazzo. 1990. The sensitivity of two cladocerans to water quality variables: salinity and hardness. Arch. Hydrobiol. 120(2): 185-196.

Cronkite, D.L., A.N. Gustafson and B.F. Bauer. 1985. Role of protein synthesis and ninhydrin-positive substances in acclimation of *Paramecium tetraurelia* to high NaCl. J. Exp. Zool. 233-21-28. (In USEPA 1988).

Crowther, R.A. and H.B.N. Hynes. 1977. The effect of road deicing salt on the drift of stream benthos. Environ Pollut. 14: 113-126.

Cumming, B.F. and J.P. Smol. 1993. Development of diatom-based salinity models for paleoclimatic research from lakes in British Columbia (Canada). Hydrobiol. 269/270: 179-196.

De Jong, L.E.D. 1965. Tolerance of *Chlorella vulgaris* for metallic and non-metallic ions. Antonie Leeuwenhoek J. Microbiol. 31: 301-313.

Delisle, C.E., and L. Deriger. 1999 (draft). Disposition des sels de deglacage et des abats-poussiere. Report submitted to the ERG.

Delisle, C.E. and P. André. 1995. The Montreal, Quebec experience in used snow disposal and treatment. Proceedings of the 35^{th} Annual Meeting of the Canadian Society of Environmental Biologists.

Delisle, C.E., J.-F. Paradis and P. André. 1995. The Montréal, Quebec experience in used snow disposal and treatment. *In*: Ecological reclamation in Canada at century's turn. Proceedings of the 35th Annual Meeting of the Canadian Society of Environmental Biologists. Canadian Plains Research Center, Regina, Saskatchewan. pp. 10–18.

Demers, C.L. 1992. Effects of road deicing salt on aquatic invertebrates in four Adirondack streams. pp. 245-252. In F. M. D'Itri, ed. Chemical Deicers and the Environment. Lewis Publishers, Florida.

Demers, C.L., and R.W. Sage, Jr. 1990. Effects of road deicing salt on chloride levels in four Adirondack streams. Water, Air, and Soil Pollution 49: 369-373.

Dickman, M.D. and M.B. Gochnauer. 1978. Impact of sodium chloride on the microbiota of a small stream. Environ. Pollut. 17: 109-126.

Dixit, S.S., J.P. Smol, D.F. Charles, R.M. Hughes, S.G. Paulsen and G.B. Collins. 1999. Assessing water quality changes in the lakes of the northeastern United States using sediment diatoms. Can. J. Fish. Aquat. Sci. 56: 131-152.

Dixit, S.S., A.S. Dixit, J.P. Smol, R.M. Hughes, and S.G. Paulsen. 2000. Water quality changes from human activites in three northeastern U.S.A. lakes. Lake and Reserv. Manage. 16(4): 305-321.

Dixit, S. S., A. S. Dixit, and J. P. Smol. 1992. Diatoms: powerful indicators of environmental change. Envir. Sci. Technol. 26:22-33.

Dowden, B.F. 1961. Cumulative toxicities of some inorganic salts to *Daphnia magna* as determined by median tolerance limits. Proc. La. Acad. Sci. 23: 77-85. (In USEPA 1988).

Dowden, B.F. and H.J. Bennett. 1965. Toxicity of selected chemicals to certain animals. J. Water Pollut. Control Fed. 37: 1308-1316. (In USEPA 1988).

Droste, R.L., and J.C. Johnston. 1993. Urban snow dump quality and pollutant reduction in snowmelt by sedimentation. Can. J. Civ. Eng. 20:9-21.

Doudoroff, P. and M. Katz. 1953. Critical review of literature on the toxicity of industrial wastes and their components to fish. Sewage and Industrial Wastes 25: 802-839.

Downing, J. A., C. Plante, and S. Lalonde. 1990. Fish production correlated with primary productivity, not the morphoedaphic index. Can. J. Fish. Aquat. Sci. 47:1929-1936.

Dussart, G.B.J. 1984. Effects on motorway run-off on the ecology of stream algae. Water Pollution Control 83: 409-415.

Edmister, J.O. and J.W. Gray. 1948. The toxicity thresholds for three chlorides and three acids to the fry of whitefish (*Coregonus clupeaformis*) and yellow pickerel (*Stizostedion v. vitreum*). Prog. Fish-Culturist 10: 105-106.

Eilers, J.M. and A.R. Selle. 1991. Changes in the conductivity, alkalinity, calcium, and pH during a 50-year period in selected Wisconsin lakes. Can. J. Fish. Aquat. Sci. 46: 1929–1944.

Ellis, M.M. 1937. Detection and measurement of stream pollution (related principally to fish life). U.S. Dept of Commerce, Bur. of Fisheries Bull. 22. (In McKee and Wolf 1963); Bull. U.S. Bur. Fish. 48: 363-437. (In Anderson 1948).

Environment Canada. 1997a (draft). Canadian Environmental Protection Act - Problem formulation for the environmental assessment of the priority substance road salts: version 1.0. Government of Canada.

Environment Canada. 1997b (draft). Canadian Environmental Protection Act - Problem formulation for the environmental assessment of the priority substance road salts: background document. Government of Canada.

Environment Canada. 1997c. Environmental assessments of priority substances under the Canadian Environmental Protection Act: Guidance Manual version 1.0. Government of Canada.

Environment Canada. 1998. Biological test method: toxicity tests using early life stages of Salmonid fish (rainbow trout). Government of Canada.

Fairchild, E.J. 1955. Low dissolved oxygen effect upon the toxicity of certain inorganic salts to the aquatic invertebrate *Daphnia magna*. Louisiana State Univ. Engineering Experiment Station Bull No. 51. 95 pp. (In McKee and Wolf 1963).

Field, R. and M.L. O'Shea. 1992. The USEPA research program on the environmental impacts and control of highway deicing salt pollution. pp. 117-133. In F. M. D'Itri, ed. Chemical Deicers and the Environment. Lewis Publishers, Florida.

Field, C.K., P.A. Silver, and A.M. Lott. 1996. Estimating the effects of changing land use patterns on Connecticut lakes. J. Environ. Qual. 25: 325-333.

Foster, A.C., and M.A. Maun. 1978. Concentration of highway deicing agents along roadsides near London. Can. J. Bot. 56:1081-1085.

Fowler, J.R. 1931. The relations of numbers of animals to survival in toxic concentrations of electrolytes. Physiol. Zool. 4: 212-245. (As cited in Anderson 1948).

Fraser, A.S. 1981. Salt in the Great Lakes. Environment Canada, National Water Research Institute.

Free, B.M. and G.G. Mulamoottil. 1983. The limnology of Lake Wabukayne, a storm-water impoundment. Water Res. Bull. 19: 821-827.

Fritz, S.C., S. Juggins, and R.W. Battarbee. 1993. Diatom assemblages and ionic characterization of lakes of the northern Great Plains, North America: a tool for reconstructing past salinity and climatic fluctuations. Can. J. Fish. Aquat. Sci. 50: 1844-1856.

Fuji, S. and J.A. Hellebust. 1994. Growth and osmoregulation of *Boekelovia hooglandii* in relation to salinity. Can. J. Bot. 72: 823-828.

Galat, D.L., M. Coleman, and R. Robinson. 1988. Experimental effects of elevated salinity on three benthic invertebrates in Pyramid Lake, Nevada. Hydrobiol. 158: 133-144.

Garrey, W.E. 1916. The resistance of freshwater fish to changes of osmotic and chemical conditions. Am. J. Physiol. 39: 313-329. (In Hammer 1977).

Gibbs, R.J. 1970. Mechanisms controlling water chemistry. Science 170: 1088-1090.

Goldman, C.R. and Horne, A.J. 1983. Limnology. McGraw-Hill, Inc., Toronto.

Gonzalez-Moreno, S., J. Gomez-Barrera, H. Perales, and R. Moreno-Sanchez. 1997. Multiple effects of salinity on photosynthesis of the protist *Euglena gracilis*. Physiologia Plantarum 101: 777-786.

Gosh, A.K. and R.N. Pal. 1969. Toxicity of four therapeutic compounds to fry of Indian major carps. Fish. Technol. 6: 120-123. (In Hammer 1977).

Gosz, J.R. 1977. Effects of ski area development and use on stream water quality of the Santa Fe Basin, New Mexico. Forest Science 23: 167-179.

Government of Quebec. 1997. Gestion des neiges usées, Guide pour l'aménagement des lieux d'élimination de neige et mise en oeuvre du règlement sur les lieux d'élimination de neige. Quebec Department of the Environment. 56 pp.

Grizzle, J.M. and A.C. Mauldin II. 1995. Effect of calcium on acute toxicity of sodium chloride to larval and newly transformed juvenile striped bass. Journal of Aquatic Animal Health 7: 298-303.

Grundy, J.E. and K.B. Storey. 1994. Urea and salt effects on enzymes from estivating and non-estivating amphibians. Molecular and Cellular Biochemistry 131: 9-17.

Hall, L.W., and R.D. Anderson. 1995. The influence of salinity on the toxicity of various classes of chemicals to aquatic biota. Critical Reviews in Toxicology 25:281-346.

Hamilton, R.W., J.K. Buttner, and R.G. Brunetti. 1975. Lethal levels of sodium chloride and potassium chloride for an oligochaete, a chironomid midge, and a caddisfly of Lake Michigan. Environmental Entomology 4: 1003-1006.

Hammer, U.T. 1977. The Effects of alkali halides on fish and other aquatic fauna. pp. 99-115. In National Research Council Environmental Secretariate. The Effects of Alkali Halides in the Canadian Environment 15019.

Hammer, U.T. 1984. The Saline Lakes of Canada. pp. 521-540. In Taub, F. B., ed. Lakes and Reservoirs. Elsevier Science Publishers, Amsterdam.

Hammer, U.T. 1986a. Saline Lake Resources of the Canadian Prairies. Revue Canadienne des Resources en Eau 11: 43-57.

Hammer, U.T. 1986b. Saline Lake Ecosystems of the World. Dr. W. Junk Publishers, Boston.

Hammer, U.T. and L. Forro. 1992. Zooplankton distribution and abundance in saline lakes of British Columbia, Canada. Int. J. Salt Lake Res. 1: 65-80.

Hammer, U.T. and R.C. Haynes. 1978. The saline lakes of Saskatchewan. II. Locale, hydrography and other physical aspects. Int. Revue ges. Hydrobiol. 63(2): 179-203.

Hammer, U.T. and R.C. Parker. 1984. Limnology of a perturbed highly saline Canadian lake. Arch. Hydrobiol. 102(1): 31-42.

Hanes, R.E., L.W. Zelazny, and R.E. Blaser. 1970. Effects of Deicing Salts on Water Quality and Biota: Literature Review and Recommended Research. Washington, D.C. Highway Research Board, National Academy of Sciences - National Academy of Engineering. 91. 70 pp.

Hanke, H. and C. Levin. 1990. Influence of winter road maintenance on traffic safety and transport efficiency. Technische Hochschule Darmstadt, Darmstadt, Germany. (In Hanneman 1992).

Hanneman, R.L. 1992. Benefits and costs in the use of salt to deice highways. pp. 431-439. In F. M. D'Itri, ed. Chemical Deicers and the Environment. Lewis Publishers, Florida.

Hanson, J.M. and W.C. Leggett. 1982. Empirical predictions of fish biomass and yield. Can. J. Fish. Aquat. Sci. 39: 257-263.

Harned, D.A. 1988. Effects of highway runoff on streamflow and water quality in the Sevenmile Creek Basin, a rural area in the Piedmont province of North Carolina, July 1981 to July 1982. U. S. geological Survey Water-Supply Paper 2329. 33 pp.

Hart, W.C. 1985. Effects on the chemistry of water supply lakes due to chloride inputs. Centre for Water Resources Research, Tech. Univ. N. S., Halifax, N. S. 16 pp. In, Nova Scotia Department of the Environment (1989). The Environmental Implications of Road Salting in Nova Scotia. Unpublished report. 40 pp. (jointed released with the Nova Scotia Department of Transportation and Communication.

Hart, W.C. 1988. Chloride measured in the vicinity of Second Chain Lake, 1987-1988. Centre for Water Resources Research, Tech. Univ. N. S., Halifax, N. S. 6 pp. In, Nova Scotia Department of the Environment (1989). The Environmental Implications of Road Salting in Nova Scotia. Unpublished report. 40 pp. (jointed released with the Nova Scotia Department of Transportation and Communication.

Hart, B.T., P. Bailey, R. Edwards, K. Hortle, K. James, A. McMahon, C. Meredith, and K. Swadling. 1990. Effects of salinity on river, stream, and wetland ecosystems in Victoria, Australia. Water Res. 24: 1103-1107.

Hawkins, R.H. and J.H. Judd. 1972. Water pollution as affected by street salting. Water Res. Bull. 8: 1246-1252.

Herbst, D.B. 1988. Comparative population ecology of *Ephydra hians* Say (Diptera: Ephydridae) at Mono Lake (California) and Albert Lake (Oregon). Hydrobiol. 158: 145-166.

Herlihy, A.T., J.L. Stoddard, and C.B. Johnson. 1998. The relationship between stream chemistry and watershed land cover data in the mid-Atlantic region, U.S. Water, Air, and Soil Pollution 105: 377-386.

Hinton, M.J. and A.G. Eversole. 1978. Toxicity of ten commonly used chemicals to American eels. Agricultural Experiment Station Technical Contribution 1595.

Hodgson, E.S. 1951. Reaction thresholds of an aquatic beetle, *Lacciphilus maculosis,* to salts and alcohols. Physiol. Zool. 24: 131. (In McKee and Wolf 1963).

Hoffman, R.W., C.R. Goldman, S. Paulson, and G.R. Winters. 1981. Aquatic impacts of deicing salts in the central Sierra Nevada Mountains, California. Water Res. Bull. 17: 280-285.

Hosiaisluoma, V. 1976. Effect of HCl and NaCl on the growth of *Netrium digitus* (Desmidiales). Ann. Bot. Fenn. 13: 107-113. (In USEPA 1988).

Hough, J.L. 1958. Geology of the Great Lakes. University of Illinois Press, Urbana. 313 pp.

Howard, K.W.F., and P.J. Beck. 1993. Hydrogeochemical implications of groundwater contamination by road de-icing chemicals. J. Contam. Hydrology 12: 245-268.

Howard, K.W.F., and J. Haynes. 1997. Contamination of urban groundwater by road de-icing chemicals. In, N. Eyles (ed.). Environmental Geology of Urban Areas. Geological Association of Canada Geotext.

Hughes, M.R., T. Zenteno-Savin, and D. Kojwang. 1991. Effects of saline acclimation and cecal ligation on body water and water flux in male and female Pekin ducks. Can. J. Zool. 69: 771-775.

Hutchinson, G.E. 1933. Experimental studies in ecologie. I. The magnesium tolerance of Daphniidae and its ecological significance. Rev d ges Hydrobiol u Hydrog Bd., 28: 90-108.

Hutchinson, G.E. 1957. A Treatise on Limnology. John Wiley and Sons, New York.

I.D. Systems Ltd. 1989. A Report on Road Dust Suppression in Manitoba. Environment Canada, Winnipeg.

Isabelle, P.S., L.J. Fooks, P.A. Keddy, and S.D. Wilson. 1987. Effects of roadside snowmelt on wetland vegetation: an experimental study. J. Environ. Manage. 25: 57-60.

Ito, K., K. Nakamura, K. Izaki, and H. Takahashi. 1977. Degradation of RNA in *Escherichia coli* induced by sodium chloride. Agric. Biol. Chem. 41: 257-263.

Iwao, T. 1936. Comparative investigation of the toxicity of various metals. Jiken Yakubutsugaku Zassi (Jour. Exp. Pharmacol.) (Japanese) 10: 357-380. (In Doudoroff and Katz 1953).

Jones, J.R.E. 1940. A further study of the relation between toxicity and solution pressure, with *Polycelis nigra* as test animal. Journal of Experimental Biology 17: 408-415.

Jones, J.R.E. 1941. A study of the relative toxicity of anions, with *Polycelis nigra* as test animal. Journal of Experimental Biology 18: 170-181.

Jones, P.H., and B.A. Jeffrey. 1986. Environmental impact of road salting – state of the art. Research and Development Branch, Ontario Ministry of Transportation and Communications. 47 pp.

Judd, J.H. 1969. Lake Stratification Caused by Runoff from Street Deicing, Milwaukee, Wisconsin. Center for Great Lakes Studies. 25 pp.

Judd, J.H. and J.W. Steggall. 1982. Reevaluation of the effects of deicing salt runoff on a small urban lake, Springfield, VA. National Technical Information Service A-114-MICH. 28 pp.

Kalinkina, L.G. 1979. Growth and biomass accumulation in marine and freshwater algae of the genus *Chlorella* as a function of NaCl concentration in the medium. Plant Physiol. 26: 320-325. (In USEPA 1988).

Kalinkina, L.G. and B.P. Strogonov. 1980. Excretion of glycolic acid by marine and freshwater algae under the influence of different NaCl concentrations. Plant Physiol. 27: 44-50. (In USEPA 1988).

Kalinkina, L.G., K.S. Spektorov and V.O. Bogoslovskaya. 1978. Effects of different concentrations of NaCl on growth and development of *Chlorella pyrenoidosa*. Plant Physiol. 25: 16-20. (In USEPA 1988).

Kanygina, A.V. & M.P. Lebedeva. 1957. The influence of industrial wastes containing sodium chloride on a (river) water system. Vodosnabzhnie I Sanit. Tekh 1: 15 (1957); Chem Abs. 51: 9045 (1957); Pub. Health Engineering Abs 37: 21 (Dec. 1957). (In McKee and Wolf 1963).

Keizer, P.D., D.C. Gordon, Jr., T.W. Rowell, R. McCurdy, D. Borgal, T.A. Clair, D. Taylor, J.G. Ogden, III, and G.E.M. Hall. 1993. Synoptic water quality survey of Halifax/Dartmouth metro area lakes on April 16, 1991. Canadian Data Report of Fisheries and Aquatic Sciences. 914

Kelly, D., J. Underwood, and J. Hansen. 1976. Influence of highway deicing salts upon water quality in Chocolate Lake, Halifax, N.S. Nova Scotia Dept. of the Environment, Halifax, N.S. 118 pp. (In N.S. Dept. of Env. 1989).

Kerekes, J.J. 1974. Limnological conditions in five small oligotrophic lakes in Terra Nova National Park, Newfoundland. J. Fish. Res. Board Can. 31: 555-583.

Kersey, K. 1981. Laboratory and field studies on the effects of road de-icing salt on stream invertebrates SIC-9. R. J. Mackay, ed. University of Toronto, Toronto.

Khanna, N., C.P. Cressman III, C.P. Tatara, and P.L. Williams. 1997. Tolerance of the nematode Caenorhabditis elegans to pH, salinity, and hardness in aquatic media. Arch. Environ. Contam. Toxicol. 32: 110-114.

Kieley, K. 1994. An assessment of the use of dust suppressant chemicals in the Atlantic Region. Canada Environmental Protection Directorate, Atlantic Region. 20 pp.

Kirchner, J.W., P.J. Dillon, and B.D. LaZerte. 1992. Predicted response of stream chemistry to acid loading tested in Canadian catchments. Nature 358: 478-482.

Kostecki, P.T. and J.J. Jones. 1983. The effect of osmotic and ion-osmotic stresses on the mortality of rainbow trout (*Salmo gairdneri*). Comp. Biochem. Physiol. 74A: 773-775. (In USEPA 1988).

Kostick, D.S. 1993. The Material Flow of Salt. Washington, D.C. U.S. Dept. of Interior - Bureau of Mines. 16 pp.

Kszos, L.A., J.D. Winter, and T.A. Storch. 1990. Toxicity of Chautauqua Lake bridge runoff to young-of-the-year sunfish (*Lepomis macrochirus*). Bull. Environ. Contam. Toxicol. 45: 923-930.

Kuemmel, D.A., and R.M. Hanbali. 1992 (preprint). Accident analysis of ice control operations. Third International Symposium on Snow Removal and Ice Control Technology, Minnesota.

LeClerc, E. 1960. The self-purification of streams and the relationship between chemical and biological tests. Proc. of the second symposium on the treatment of water waters, 281. Pergamon Press, London. (In McKee and Wolf 1963).

LeClerc, E. and F. Devlaminck. 1950 (estimated). Toxicological experiments with some products utilized in tanneries. Reprint from Centre Belge d'Etude et de Documentdes Eaux 541. (In McKee and Wolf 1963).

LeClerc, E. and F. Devlaminck. 1955. Fish toxicity tests and water quality. Bull. of Centre Belge Document Eaux 28: 11.

Leduc, A, and C.E. Delisle. 1990. Quality of used snow from city streets and sidewalks in Montreal (Quebec), Canada. International Conference on Urban Hydrology Under Wintry Conditions, Norway.

Lee, S.-III, E.-J. Na, Y.-O. Cho, B. Koopman, and G. Bitton. 1997. Short-term toxicity test based on algal uptake by *Ceriodaphnia dubia*. Water Environmental Research 69(7): 1207-1210.

Leland, H.V., and S.V. Fend. 1998. Benthic invertebrate distributions in the San Joaquin River, California, in relation to physical and chemical factors. Can. J. Fish. Aquat. Sci. 55: 1051-1067.

Letts, A.D. 2000a. Effects of sodium ferrocyanide derived from road salting on the ecosystem. Morton Salt, Morton International Inc., Chicago, Illinois, prepared for the. Environmental Resource Group on Road Salts. 165 pp.

Letts, A.D. 2000b. A model to predict chloride concentrations in meltwater runoff from road salt. Morton Salt, Morton International Inc., Chicago, Illinois.

Likens, G.E. 1985. An ecosystem approach to aquatic ecology: Mirror Lake and its environment. Springer-Verlag, New York. 516 pp.

Lowell, R.B., J.M. Culp, and F.J. Wrona. 1995. Toxicity testing with artificial streams: effects of differences in current velocity. Envir. Tox. Chem. 14: 1209-1217.

Mackie, G.L. 1978. Effects of pollutants on natality of *Musculium securis* (Bivalvia: Pisidiidae). The Nautilus 92: 25-33.

MacLeod, C.L., M.P. Borcsik and P.R. Jaffe. 1996. Effect of infiltrating solutions on the desorption of mercury from aquifer sediments. Environ. Technol. 17: 465-475.

Maltby, L., D.M. Farrow, A.B.A. Boxall, P. Calow and C.I. Betton. 1995a. The effects of motorway runoff on freshwater ecosystems: 1. Field study. Environ. Toxicol. Chem. 14: 1079-1092.

Maltby, L., A.B.A. Boxall, D.M. Farrow, P. Calow and C.I. Betton. 1995b. The effects of motorway runoff on freshwater ecosystems: 2. Identifying major toxicants. Environ. Toxicol. Chem. 14: 1093-1101.

Manahan, S.E. 1990. Toxicological Chemistry - A Guide to Toxic Substances in Chemistry. Lewis Publishers, Inc., Michigan.

Marsalek, J., Q. Rochfort, T. Mayer, M. Servos, B. Dutka and B. Brownlee. 1999a. Toxicity testing for controlling urban wet-weather pollution: advantages and limitations. Urban Water 1: 91-103.

Marsalek, J., Q. Rochfort, B. Brownlee, T. Mayer and M. Servos. 1999b. An exploratory study of urban runoff toxicity. Wat. Sci. Tech. 39: 33-39.

Mattson, M.D., and P.J. Godfrey. 1994. Identification of road salt contamination using multiple regression and GIS. Journal of Environmental Management 18: 767-773.

Matuszek, J.E. 1978. Empirical predictions of fish yields of large North American lakes. Trans. Am. Fish. Soc. 107: 385-394.

Mayer, T., J. Marsalek, and E. Delos Reyes. 1996. Nutrients and metal contaminants status of urban stormwater ponds. J. lake and Reservoir Management 12:348-363

Mayer , T., Q. Rochfort, J. Marsalek, M. Servos, A. Jurkovic, and R. McInnis. 1998 (draft). Effect of deicing salts on the chemistry and toxicity of highway runoff. Report submitted to the Road Salt Environmental Resources Working Group, Chemical Evaluation Branch, Environmental Protection, Hull. 8 pp.

Mayer, T., W.J. Snodgrass, and D. Morin. 1999. Spatial characterization of the occurrence of road salts and their environmental concentrations as chlorides in Canadian surface waters and benthic sediments. Water Qual. Res. J. Canada 34(4): 545-574.

McBean, E. and S. Al-Nassri. 1987. Migration pattern of de-icing salts from roads. Journal of Environmental Management 25: 231-238.

McKee, J.E. and H.W. Wolf. 1963. Water Quality Criteria, 2nd ed. California State Water Resources Control Board. 1-542 pp.

McKillop, W.B., R.T. Patterson, L.D. Delorme, and T. Nogrady. 1992. The origin, physico-chemistry and biotics of sodium chloride dominated saline waters on the western shore of Lake Winnipegosis, Manitoba. Canadian Field-Naturalist 106: 454-473.

Melack, J.M. 1988. Primary producer dynamics associated with evaporative concentration in a shallow, equatorial soda lake (Lake Elmenteita, Kenya). Hydrobiol. 158:1-14.

Mohammed, A.A. and A.A. Shafea. 1992. Growth and some metabolic activities of *Scenedesmus obliquus* cultivated under different NaCl concentrations. Biologia Plantarum, 34: 423-430.

Moll, R.A., R. Rossmann, J.A. Barres, and F.J. Horvath. 1992. Historical trends of chlorides in the Great Lakes. Pages 303-322 in F. M. D'Itri, ed. Chemical Deicers and the Environment. Lewis Publishers, Florida.

Molles, M.C. and J.R. Gosz. 1980. Effects of a ski area on the water quality and invertebrates of a mountain stream. Water, Air and Soil Pollution 14: 187-205.

Morin, D. 2000. Environmental impacts from the spreading and storage of road salts: a review of case studies. Environment Canada's Environmental Resource Group.

Morin, D. and M. Perchanok. 2000. Road salt loadings in Canada. Document subimitted to the Environmental Resource Group for Road Salts, Comercial Chemicals Evaluation Branch, Environment Canada. May 2000. 85 pp.

Morin, D., W. Snodgrass, J. Brown and P.A. Arp. 2000 (draft). Impacts evaluation of road salt loads on soils and surface water. Environment Canada's Environmental Resource Group.

Moser, K.A., J.P. Smol, D. R.S. Lean, and G.M. MacDonald. 1998. Physical and chemical limnology of northern boreal lakes, Wood Buffalo National Park, northern Alberta and the Northwest Territories, Canada. Hydrobiol. 377: 25-43.

Murphy, A. 1990. Hydrogeological and geochemical characteristics of a shallow overburden aquifer affected by the application of road salt in the Windsor Lake area of St. John's, Newfoundland. Hons. Diss. Memorial University of Newfoundland. 138pp.

Naumann, E. 1934. Uber die Beeinflussung von *Daphnia magna* durch einege Salze und Salzmischungen. K. fysiogr. Sallsk. Lund Forh., Bd. 4:11-12. (In Anderson 1948).

New Brunswick Department of the Environment - Water Resources Branch. 1978. An investigation of the quantity and quality of leachate from a highway salt treated sand pile under normal operating conditions. 57 pp.

Nova Scotia Department of the Environment. 1989. The Environmental Implications of Road Salting in Nova Scotia. Unpublished report. 40 pp. (jointed released with the Nova Scotia Department of Transportation and Communication.

Novotny, V., D. Muehring, D.H. Zitomer, D.W. Smith and R. Facey. 1998. Cyanide and metal pollution by urban snowmelt: impact of deicing compounds. IAWQ 19th biennial international conference preprint, book 8, pp. 219-226.

OECD Scientific Expert Group. 1989. Curtailing Usage of De-icing Agents in Winter Maintenance. Paris, OECD.

Oglesby, R.T. 1977. Phytoplankton summer standing crop and annual productivity as functions of phosphorus loading and various physical factors. J. Fish. Res. Board Can. 34: 2255-2270.

Ohno, T. 1990. Levels of total cyanide and NaCl in surface waters adjacent to road salt storage facilities. Environmental Pollution, 67: 123-132.

Oliver, B.J., J.B. Milne, and N. LaBarre. 1974. Chloride and lead in urban snow. J. Wat. Poll. Cont. Fed. 46: 766-771.

Ontario Ministry of the Environment - Environmental Monitoring and Reporting Branch. 1999. Surface water monitoring and assessment: 1997 Lake Ontario report featuring a summaryof tributary and nearshore conditions and trends for the Lake Ontario Basin. 49pp.

Oshima, S. 1931. On the toxic action of dissolved salts and their ions upon young eels (*Anguilla japonica*). Jour. Imperial Fisheries Exp. Sta. (Japanese). 2: 139-193. (In Doudoroff and Katz 1953).

Ouellet, M., M. Dickman, M. Bisson, and P. Page. 1989. Physico-chemical characteristics and origin of hypersaline meromictic Lake Garrow in the Canadian High arctic. Hydrobiologia. 172: 215-234.

Padhye, A.D. and H.V. Ghate. 1992. Sodium chloride and potassium chloride tolerance of different stages of the frog, *Microhyla ornata.* Herpetological Journal 2: 18-23.

Paine, R.L. 1979. Chlorides in the Don River watershed resulting from de-icing salt. Institute for Environmental Studies, University of Toronto, Toronto, Ontario. 23 pp.

Patrick, R., J.J. Cairns, and A. Scheier. 1968. The relative sensitivity of diatoms, snails and fish to twenty common constituents of industrial wastes. The Progressive Fish-Culturist 30: 137-140.

Perchanok, M.S., D.G. Manning, and J.J. Armstrong. 1991. Highway De-Icers: Standards, Practice, and Research in the Province of Ontario. Research and Development Branch, Ministry of Transportation.

Peters, N.E., and J.T. Turk. 1981. Increases in sodium and chloride in the Mohawk River, New York, from the 1950's to the 1970's attributed to road salt. Water Res. Bull. 17:586-598.

Phillips, A.M.J. 1944. The physiological effect of sodium chloride upon brook trout. Trans. Amer. Fish. Soc. 74: 297-309.

Pickering, Q.H., J.M. Lazorchak, and K.L. Winks. 1996. Subchronic sensitivity of one-, four- and seven-day-old fathead minnow (*Pimephales promelas*) larvae to five toxicants. Environ. Tox. and Chem. 15: 353-359.

Pientz, R., J.P. Smol, and D.R.S. Lean. 1997. Physical and chemical limnology of 24 lakes located between Yellowknife and Contwoyto Lake, Northwest Territories. Can. J. Fish. Aquat. Sci. 54: 347-358.

Pionke, H.B. and J.B. Urban. 1985. Effects of agricultural land use on ground-water quality in a small Pennsylvania watershed. Ground Water 23: 68-80.

Pringle, C.M., D.S. White, C.P. Rice, and M.L. Tuchman. 1981. The Biological Effects of Chloride and Sulfate with Special Emphasis on the Laurentian Great Lakes. University of Michigan, Ann Arbor. 51 pp.

Powers, E.B. 1917. The goldfish (*Carassius carassius*) as a test animal in the study of toxicity. ILL Biol. Monogr. 4(2): 1-73.

Prowse, C.W. 1987. The impact of urbanization on major ion flux through catchments: a case study in southern England. Water, Air, and Soil Pollution 32: 277-292.

Pruter, A.T. and D.L. Alverson. 1972. The Columbia River Estuary and Adjacent Ocean Waters - Bioenvironmental Studies. University of Washington Press, Seattle. 868 pp.

Pugh IV, A L., S.A. Norton, M. Schauffler, G.L. Jacobson Jr., J.S. Kahl, W.F. Brutsaert, and C.F. Mason. 1996. Interactions between peat and salt-contaminated runoff in Alton Bog, Maine, USA. J. Hydrology 182:83-104.

Ralston, J.G., and F. Hamilton. 1978. Chloride inputs to Lake Ontario and the significance of deicing salts. Unpublished report. Water Resources Branch, Ontario Ministry of the Environment. 25 pp.

Ramult, M. 1925. Development and resisting power of Cladocera embryos in solutions of certain inorganic salts. Bull. l'Acad. Polon. Sci. Series B 135-194. (In Anderson 1948).

Rantamaki, J., L. Cerenius, and K. Soderhall. 1992. Prevention of transmission of the crayfish plague fungus (*Aphanomyces astaci*) to the freshwater crayfish Astacus astacus by treatment with $MgCl_2$. Aquaculture 104: 11-18.

Rawson, D.S. 1953. The bottom fauna of Great Slave Lake. J. Fish. Res. Bd. Can. 10(8): 486-520.

Rawson, D.S. 1956. The net plankton of Great Slave Lake. J. Fish. Res. Bd. Can. 13(1): 53-127.

Rawson, D.S. 1960. A limnological comparison of twelve large lakes in northern Saskatchewan. Limnol. Oceanogr. 5(2): 195-211.

Rawson, D.S. and J.E. Moore. 1944. The saline lakes of Saskatchewan. Can. J. Res. 22: 141-201.

Raymont, J.E. 1967. Plankton and productivity in the oceans. Pergamon Press, Toronto.

Reynoso, G.T., B. A. de Gamboa and S. R. Mendoza. 1982. Induction of resistance to salinity in the freshwater alga *Chlamydomonas reinhardtii*. Environ. Sci. Res. 23: 531-534.

Ring, R.A. 1991. The insect fauna and some other characteristics of natural salt springs on Saltspring Island, British Columbia. Memoirs of the Entomological Society of Canada 155: 51-61.

Rokosh, et al. 1997. Toxicity of highway runoff samples. Ontario Ministry of the Environment Report. MAT-98-XX. 50 pp. In, Snodgrass et al., 2000. Entry characterization of road salt: dynamics of snowmelt runoff chloride concentrations and relationships to laboratory measurements of toxicity. Report submitted to the ERG.

Romspert, A.P. 1976. Osmoregulation of the African clawed frog, *Xenopus laevis*, in hypersaline media. Comp. Biochem. Physiol. 54A:207-210.

Rott, E., H.C. Duthie, and E. Pipp. 1998. Monitoring organic pollution and eutrophication in the Grand River, Ontario, by means of diatoms. Can. J. Fish. Aquat. Sci. 55: 1443-1453.

Rowe, C.L. and Dunson, W.A. 1993, Relationships among abiotic parameters and breeding effort by three amphibians in temporary wetlands of central Pennsylvania: Wetlands. 13: 237-246.

Ryder, R.A., S.R. Kerr, K.H. Loftus, and H.A. Regier. 1974. The morphoedaphic index, a fish yield estimator – review and evaluation. J. Fish. Res. Board. Can. 31: 663-668.

Scanton, D. 1999. Personal communication (letter dated May 19, 1999). Ontario Ministry of Transportation, Toronto, Ontario.

Schnabel, R.R., J.B. Urban, and W.J. Gburek. 1993. Hydrological controls in nitrate, sulfate, and chloride concentrations. J. Environ. Qual. 22: 589-596.

Schroeder, J.E. and K.R. Solomon. 1999. Impact of road salts on an urban watershed. Annual meeting of the Society of Environmental Toxicology and Chemistry (SETAC), November, Charlotte, N.C. Poster.

Scott, W.S. 1980a. De-icing salt levels in Toronto stream banks and roadside soils. Bull. Environm. Contam. Toxicol. 25: 208-214.

Scott, W.S. 1980b. Road salt movement into two Toronto streams. Journal of the Environmental Engineering Division EE3: 547-560.

Scott, W.S. 1981. An analysis of factors influencing de-icing salt levels in streams. Journal of Environmental Management 13: 269-287.

Scott, W.B. and E.J. Crossman. 1973. Freshwater Fisheries of Canada. Fish. Res. Bd. Canada Bull. 184. 966 pp.

Setter, T.L., H. Greenway and J. Kuo. 1982. Inhibition of cell division by high external NaCl concentrations in synchronized cultures of *Chlorella emersonii*. Aust. J. Plant Physiol.9: 179-196.

Shitole, M.G. and G.V. Joshi. 1984. Effect of sodium chloride on the balance between C_3 and C_4 carbon fixation pathways and growth. Photosynthetica 18: 377-384. (In USEPA 1988)

Short, T.M., J.A. Black, and W.J. Birge. 1991. Ecology of a saline stream: community responses to spatial gradients of environmental conditions. Hydrobiol. 226: 167-178.

Siver, P.A., R.W. Canavan IV, C.K. Field, L.J. Marsicano, and A. Lott. 1996. Historical changes in Connecticut lakes over a 55-year period. J. Environ. Qual. 25: 334-345.

Smith, M.E. and J.L. Kaster. 1983. Effect of rural highway runoff on stream benthic macroinvertebrates. Environmental Pollution Series A 32: 157-170.

Smith, R.A., R.B. Alexander, and M.G. Wolman. 1987. Water-quality trends in the nation's rivers. Science 235: 1607-1615.

Smol, J.P., Brown, S.R., and McNeely, R.N. 1983. Cultural disturbances and trophic history of a small meromictic lake from Central Canada. Hydrobiol. 103: 125-130.

Snodgrass, W.J., D. Rokosh, Q. Rochfort, T. Mayer, D. Morin, and J. Marsalek. 2000. Entry characterization of road salt: dynamics of snowmelt runoff chloride concentrations and relationships to laboratory measurements of toxicity. Report submitted to the Environment Canada CEPA Priority Substances List Environmental Resource Group on Road Salts. March, 2000. Environment Canada, Commercial Chemicals Evaluation Branch, Ottawa, Ontario.

Spehar, R.L. 1987. U.S. EPA, Duluth, MN. (Memorandum to C. Stephan. U.S. EPA, Duluth, MN. June 24). (In U. S. EPA 1988)

Sridhar, K.R. and F. Barlocher. 1997. Water chemistry and sporulation by aquatic hyphomycetes. Mycological Research 101(5): 591-596.

Stanley, R.A. 1974. Toxicity of heavy metals and salts to Eurasian watermilfoil (*Myriophyllum spicatum*). Arch. Environ. Contam. Toxicol. 2: 331-341. (In USEPA 1988).

Stemberger, R.S., and J.M. Lazorchak. 1994. Zooplankton assemblage responses to disturbance gradients. Can. J. Fish. Aquat. Sci. 51: 2435-2447.

Sullivan, T.J., C.D. Driscoll, J.M. Eilers, and D.H. Landers. 1988. Evaluation of the role of sea salt inputs in the long-term acidification of coastal New England lakes. Environ. Sci. Technol. 22: 185-190.

Sutcliffe, D.W. 1961a. Studies on salt and water balance in caddis larvae (Trichoptera): I. osmotic and ionic regulation of body fluids in Limnephilus affinis Curtis. J. Exp. Biol. 38: 501-519.

Sutcliffe, D.W. 1961b. Studies on salt and water balance in caddis larvae (Trichoptera): II. osmotic and ionic regulation of body fluids in Limnephilus stigma Curtis and Anabolia nervosa Leach. J. Exp. Biol. 38: 521-530.

Teeter, J.W. 1965. Effects of sodium chloride on the sago pondweed. J. Wildl. Manage. 29: 838-845. (In USEPA 1988).

Thirumurthi, D. and K.-H. Tan. 1978. Establishing baseline data for management of Halifax, N.S., Lake Watersheds. Journal of the American Water Works Association 70(8): 436-445.

Thornton, K.W. and J.R. Sauer. 1972. Physiological effects of NaCl on Chironomus attenuatus (Diptera: Chironomidae). Annals of the Entomological Society of America 65: 872-875.

Threader, R.W. and A.H. Houston. 1983. Use of NaCl as a reference toxicant for goldfish, *Carassius auratus*. Can. J. Fish. Aquat. Sci. 40: 89-92.

Trama, F.B. 1954. The acute toxicity of some common salts of sodium, potassium, and calcium to the common bluegill (*Lepomis macrochirus*). Proc. Acad. Nat. Sci. Philid. 196: 185. (In McKee and Wolf 1963).

Transport Association of Canada. 1999. Road Salt Management Guide. 295 pp.

Tuchman, M.L., E.F. Stoermer, and H.J. Carney. 1984. Effects of increased salinity on the diatom assemblage in Fonda Lake, Michigan. Hydrobiol. 109: 179-188.

Underwood, J.K., J.G. Ogden, and D.L. Smith. 1986. Contemporary chemistry of Nova Scotian lakes. Water, Air, and Soil Pollution 30: 991-999.

Underwood, J.K., and D.M. Josselyn. 1979. The extent of road salt and nutrient stressing of Williams Lake, Halifax, N. S., N. S. Department of the Environment, Halifax, N. S. 88 pp. In, Nova Scotia Department of the Environment (1989). The Environmental Implications of Road Salting in Nova Scotia. Unpublished report. 40 pp. (jointed released with the Nova Scotia Department of Transportation and Communication.

United States Environmental Protection Agency (USEPA). 1976. Quality criteria for water. USEPA, Washington, D.C.

United States Environmental Protection Agency (USEPA). 1988. Ambient water quality Criteria for chloride. USEPA, Washington, D.C.

Vaa, T., K. Sakshaug, and O.P. Resen-Fellie. 1996. Road salting and traffic safety - the effect of road salting on accidents and driving speed. Four International Symposium on Snow Removal and Ice Control Technology, Nevada.

Vallentyne, J.R. 1972. Freshwater supplies and pollution: effects of the demophoric explosion on water and man. Pages 181-211 in N. Polunin, ed. The Environmental Future. MacMillan Press, London. (As cited in Hammer 1986b).

Van Horn, W.M., J.B. Anderson, and M. Katz. 1949. The effect of kraft pulp mill wastes on some aquatic organisms. Trans. Amer. Fish. Soc. 79: 55-63.

Waller, D.L., Fisher, S.W., and H. Drabrowska. 1996. Prevention of zebra mussel infestation and dispersal during aquaculture operations. The Progressive Fish Culturist 58(2): 77-84.

Wallen, I.E., W.C. Greer, R. Lasater. 1957. Toxicity to *Gambusia affinis* of certain pure chemicals in turbid waters. The Environmental Future 29: 695. (In McKee and Wolf 1963).

Waller, D.L., S.W. Fisher, and H. Dabrowska. 1996. Prevention of zebra mussel infestation and dispersal during aquaculture operations. The Progressive Fish Culturist 58(2): 77 - 84.

Wang, J.S., P.M. Huang, W.K. Liaw, and U.T. Hammer. 1991. Kinetics of the desorption of mercury from selected freshwater sediments as influenced by chloride. Water, Air and Soil Pollution 56: 533.

Warren, E. 1899. On the reaction of *Daphnia magna* to certain changes in its environment. Quart. Jour. Micr. Sci. 43: 199-557. (In Anderson 1948).

Warren, L.A. and A.P. Zimmerman. 1994. The influence of temperature and NaCl on cadmium, copper and zinc partitioning among suspended particulate and dissolved phases in an urban river. Water Research 28(9): 1931.

Warrington, P. and C. Phelan. 1998 (draft). Road salt and Winter Maintenance - A Best Management Practices Guide to Protecting Water Quality. Water Management Section, Ministry of Environment, Lands and Parks, Victoria, BC.

Watson T. 2000. [Data on chloride concentrations in ponds adjacent to roadways in Southern Ontario]. Personal communication from T. Watson (University of Toronto, Dept. of Biology, University of Toronto) to D. Morin (Environment Canada, Existing Substances Branch, Hull), Communication between October, 2000, and January, 2001.

Wetzel, R. G. 1983. Limnology. Saunders College Publishing, Toronto.

Whitfield, P. H. and N. L. Wade. 1992. Monitoring transient water quality events electronically. Water Res. Bull. 28(4): 703-711.

Wiebe, A.H., J.G Burr, and H.E. Faubion. 1934. The problem of stream pollution in Texas with special reference to salt water from oil fields. Trans. Amer. Fish. Soc. 64: 81-86.

Wilcox, D.A. 1982. The Effects of Deicing Salts on Water Chemistry and Vegetation in Pinhook Bog, Indiana Dunes National Lakeshore. Purdue University. 134 pp.

Wilcox, D.A. 1984. The effects of NaCl deicing salts on *Sphagnum recurvum* P. Beauv. Environ. and Exp. Bot. 24: 295-304.

Wilcox, D.A. 1986. The effects of deicing salts on vegetation in Pinhook Bog, Indiana. Canadian Journal of Botany 64: 865-874.

Wilcox, D.A. and R.E. Andrus. 1987. The role of *Sphagnum fimbriatum* in secondary succession in a road salt impacted bog. Can. J. Bot. 65: 2270-2275.

Williams, W.D. 1978. Limnology of Victorian salt lakes, Australia. Verh. Int. Ver. Limnol. 20: 1165-1174.

Williams, W.D. 1987. Salinization of rivers and streams: an important environmental hazard. Ambio. 16: 180-185.

Williams, W.D., A.J. Boulton, and R.G. Taffe. 1990. Salinity as a determinant of salt lake fauna: a question of scale. Hydrobiol. 197: 257-266.

Williams, W.D., R.G. Taffe, and A. Boulton. 1991. Longitudinal distribution of macroinvertebrates in two rivers subject to salinization. Hydrobiol. 210: 151-160.

Williams, D.D., N.E. Williams, and Y. Cao. 1997. Spatial differences in macroinvertebrate community structure in springs in southeastern Ontario in relation to their chemical and physical environment. Can. J. Zool. 75: 1404-1414.

Williams, D.D., N.E. Williams, and Y. Cao. 1999. Road salt contamination of groundwater in a major metropolitan area and development of a biological index to monitor its impact. Water Res. 34:127-138.

WI SLOH (Wisconsin State Laboratory of Hygiene). 1995. Unpublished data on chloride toxicity to aquatic species. From A. Letts,(Technical Manager, Morton International Inc. Chicago, Illinois) to M.S. Evans, (National Hydrology Research Institute, Environment Canada, Saskatoon) Aug.11, 1998.

Wood, R.B., and J.F. Talling. 1988. Chemical and algal relationships in a salinity series of Ethiopian inland lakes. Hydrobiol. 158: 29-67.

Young, R.T. 1923. Resistance of fish to salts and alkalinity. Amer. Jour. Physiol. 63: 373. (In Hanes et al. 1970).

Zeeb, B.A. and Smol, J.P. 1991. Paleolimnological investigation of the effects of road salt seepage on scaled chrysophytes in Fonda Lake, Michigan. Journal of Paleolimnology 5: 263-266.

APPENDIX A: Summary Of The Regional Effects Of Road Salts On Aquatic Ecosystems

Table A-1: Maritime regions of Canada (Newfoundland, Nova Scotia, New Brunswick, and Prince Edward Island) and the United States.

Ecological Characteristic or Species Affected	Specific Location	Response to Salt Loading	Duration of Response	Baseline or Upstream Concentrations	Value of Main Loading into Ecosystem	Source of Loading	New Concentration After Loading	Duration of New Concentration	Additional Notes	Reference
Streams [1]										
water quality	Waterford River Basin, Nfld.	increased chloride concentration	unknown	17 mg/L Cl^- (Ruby Line)	unknown	industrial activities, road salt and depots, sea spray	70 mg/L Cl^- (Donovans Station)	unknown	Ruby Line had no known industrial sources	Arsenault et al. 1985
water quality	Monks' Brook, Hampshire England	ionic flux (Cl^- and Na^+)	unknown	unknown	258 tonnes Cl^- (urban); 4 tonnes Cl^- (rural)	urban and rural runoff	unknown	unknown	sewage and other wastes also sources of chloride and sodium	Prowse 1987
water quality	162 Massachusetts streams, USA	increased chloride concentration	unknown	unknown	4,380-22,500 kg/km NaCl	urban road runoff	≥50 mg/L	unknown	highest concentrations found in urban areas	Mattson and Godfrey 1994
water quality	Androscoggin and Kennebec Rivers, ME, USA	increased sodium and chloride concentration	unknown	<1 mg/L	unknown	highway deicing salt	15-18 mg/L	unknown	both rivers flow through areas of high road density	Hanes et al. 1970
water quality	Penobscot, Machias and Narraguagus Rivers, ME, USA	increased sodium and chloride concentration	unknown	<1 mg/L	unknown	highway deicing salt	6 to 8 mg/L	unknown	rivers located in areas of low road density	Hanes et al. 1970
water quality	Penjajwoc Stream, Bangor, ME, USA	increased chloride concentration	see Duration of New Concentration	5-15 mg/L Cl^- 5-12 mg/L Na^+ (yearly average at upstream control)	unknown	urban road runoff	10-50 mg/L Cl^- 15-30 mg/L Na^+ (yearly average)	highest concentrations (621 mg/L Cl^- 407 mg/L Na^+) occurred after	length of stream affected unknown	Boucher 1982

								runoff events		

Table A-1: Continued

Ecological Characteristic or Species Affected	Specific Location	Response to Salt Loading	Duration of Response	Baseline or Upstream Concentrations	Value of Main Loading into Ecosystem	Source of Loading	New Concentration After Loading	Duration of New Concentration	Additional Notes	Reference
water quality	Washington, D.C. and Baltimore, MD, USA	increased sodium and chloride concentration	see Duration of New Concentration	< 40 mg/L Cl^-, < 20 mg/L Na^+	~30-130 mg/L Cl^-, ~10-55 mg/L Na^+	highway deicing salt runoff	maximum 128 mg/L Cl^-, 54.5 mg/L Na^+	increase from November to peak in March, decrease in June and July	despite high loadings, little or no impact noted due to high level of dilution	August and Graupen-sperger, 1989
stream flow and water quality	Sevenmile Creek Basin, NC, USA	increased ionic concentration, salinity and alkalinity	see Duration of New Concentration	control sites: 3.1-3.7 mg/L Na^+, 2.9-7.8 mg/L Cl-, 44-69 µS/cm specific conductivity, 14-28 mg/L ($CaCO_3$) alkalinity	total dissolved solids 240-1800 tons per square mile	highway deicing salt runoff	25-510 mg/L Na^+, 70-1100 mg/L Cl^-, 256-591 µS/cm specific conductivity, 27-54 mg/L ($CaCO_3$) alkalinity	highway sites had higher concentrations than control sites throughout the year	Sevenmile drainage basin drains 4.8 mile section of I-85; adjacent drainage basins (Rocky Run and Cane Creek used as controls)	Harned 1988
Lakes[2]										
water quality	Pine Hill Pond, Terra Nova National Park, Nfld.	increased chloride concentration	unknown	14.0 mg/L Cl^- (average of monthly samples April to Dec. 1969)	unknown	highway deicing salt runoff	94.0 mg/L Cl^- (March, 1969)	unknown	area=2.05 ha; max. depth =5.5m; mean depth=1.06 m; vol.=$0.45 \cdot 10^6$ m^3	Kerekes 1974
water quality	234 Nova Scotia lakes	increased chloride concentration	unknown	8.1 mg/L	unknown	sea spray, highway deicing salt runoff	28.1 mg/L	unknown	samples collected in summers of 1981-1984	Underwood et al. 1986
incomplete vertical mixing in spring due to salt-induced density gradient	Chocolate Lake, NS	increased chloride concentration in bottom waters	unknown	unknown	120 tons/year of salt into the lake	highway deicing salt runoff	average = 120 mg/L Na^+ & 207 mg/L Cl^-; in deeper regions = 206 mg/L Na^+ & 330 mg/L Cl^- (maximum)	unknown	lake morphometry unknown	Kelly et al. 1976 (in NS Dept. of Env. 1989)

Table A-1: Continued

Ecological Characteristic or Species Affected	Specific Location	Response to Salt Loading	Duration of Response	Baseline or Upstream Concentrations	Value of Main Loading into Ecosystem	Source of Loading	New Concentration After Loading	Duration of New Concentration	Additional Notes	Reference
water quality	Chain Lakes, near Halifax, NS	increased chloride concentration	see Duration of New Concentration	~19 mg/L Cl^- (June to October 1975)	unknown	salt application to roads, such as St. Margaret's Bay Road, in winter and spring	43 mg/L Cl^- (May 1976; maximum for year of sampling)	increase from November to peak in May with rapid decrease to end of June, 1976	First Chain: surface area = 20ha; maximum depth = 11m; mean depth = 4.1m; volume = 0.81×10^6 m^3. Second Chain: surface area = 16ha; maximum depth = 11m; mean depth = 3.2m; volume = 0.33×10^6 m^3	Thirumurthi and Tan 1978
water quality	five ponds in Waterford River Basin, Nfld.	increased spring chloride concentration	unknown	6 mg/L Cl^- (Bremigens Pond)	unknown	industrial and road runoff	14 mg/L Cl^- (Barazil Pond); 18 mg/L Cl^- (Branscombe Pond); 24 mg/L Cl^- (District Pond)	unknown	sampling conducted in spring 1994	Arsenault et al. 1985
water quality	10 farm ponds, ME, USA	increased sodium and chloride concentration	see Duration of New Concentration	unknown	unknown	road salt	1.4-115 mg/L Na^+, <1-210 mg/L Cl^-	sodium and chloride concentrations increased from July 1965 to April 1966	ponds located near roads; samples collected in 1967 suggest increased chloride concentration (1.4-221 mg/L)	Hanes et al. 1970

Table A-1: Continued

Ecological Characteristic or Species Affected	Specific Location	Response to Salt Loading	Duration of Response	Baseline or Upstream Concentrations	Value of Main Loading into Ecosystem	Source of Loading	New Concentration After Loading	Duration of New Concentration	Additional Notes	Reference
Wetlands										
water quality	Mount Pearl, Waterford River Basin, Nfld.	increased chloride and sodium concentration	unknown	unknown	unknown	salt depot	7.8-178 mg/L Cl^- (mean 42.8 mg/L Cl^-); 6.1-130 mg/L Na^+ (mean 25.8 mg/L Na^+)	unknown	when depot moved, nearby well increased from 13-14 mg/L to 43-180 mg/L Cl^- and 9.8-16 mg/L to 55-1,360 mg/L Na^+	Arsenault et al. 1985
water quality	wetlands near sand-salt storage site in Alton, ME, USA	increased Na^+, Cl^- and CN^- concentration	see Duration of New Concentration	6 mg/L Na^+; 8 mg/L Cl^-; <10 µg/L total CN^- (from upslope location)	unknown	sand-salt storage piles	16-8663 mg/L Na^+; 15-12,463 mg/L Cl^-; <10-200 µg/L total CN^- (March to November 1988 with maximum in September)	higher concentrations in June, July, Sept. and Oct.; spring conc. may be result of water evap. from wetlands; fall conc. result of fresh salt application	wetland morphometry unknown; wetlands within 30 m of sand-salt stockpile	Ohno 1990
water quality	wetlands near sand-salt storage site in Kenduskeag, ME, USA	increased concentration of Na^+, Cl^- and CN^-	see Duration of New Concentration	3 mg/L Na^+; 4 mg/L Cl^-; <10 µg/L total CN^- (from upslope location)	unknown	sand-salt storage piles	807-8725 mg/L Na^+; 15-12,463 mg/L Cl^-; <10-36 µg/L total CN^- (March to Nov. 1988)	higher conc. in May, June, July, for Na^+ & Cl^-; for CN^-, higher conc. in Sept. & Oct.; (see Ohno 1990 above for possible reasons)	wetland morphometry unknown; wetlands within 30 m of sand-salt stockpile	Ohno 1990

Table A-1: Continued

Ecological Characteristic or Species Affected	Specific Location	Response to Salt Loading	Duration of Response	Baseline or Upstream Concentrations	Value of Main Loading into Ecosystem	Source of Loading	New Concentration After Loading	Duration of New Concentration	Additional Notes	Reference
water quality	wetlands near sand-salt storage site in Winterport, ME, USA	increased Na^+, Cl^- and CN^- concentration	see Duration of New Concentration	48 mg/L Na^+; 80 mg/L Cl^-; <10 µg/L total CN^- (from upslope location)	unknown	sand-salt storage piles	438-2950 mg/L Na^+; 788-4563 mg/L Cl^-; <10-117 µg/L total CN^- (March to Nov. 1988 with max. in Sept.)	higher conc. in June, July, and Sept. for Na^+ & Cl^-; for CN^-, higher conc. in Sept. & Nov.; (see Ohno 1990 above for possible reasons)	wetland morphometry unknown; wetlands within 30 m of sand-salt stockpile	Ohno 1990
water quality	wetlands near sand-salt storage site in Aurora, ME, USA	increased Na^+, Cl^- and CN^- concentration	see Duration of New Concentration	3 mg/L Na^+; 3 mg/L Cl^-; <10 µg/L total CN^- (from upslope location)	unknown	sand-salt storage piles	3750-9075 mg/L Na^+; 5550-13500 mg/L Cl^-; 15-103 µg/L total CN^- (March to Nov. 1988 with max. in Sept.)	higher conc. in June and Sept. for Na^+ & Cl^-; for CN^-, higher conc. in Sept.; (see Ohno 1990 above for possible reasons)	wetland morphometry unknown; wetlands within 30 m of sand-salt stockpile	Ohno 1990

[1] streams include streams, creeks, springs and rivers.
[2] lakes include lakes and ponds.

Table A-2: Central Canada (Ontario and Quebec) and the United States

Ecological Characteristic or Species Affected	Specific Location	Response to Salt Loading	Duration of Response	Baseline or Upstream Concentrations	Value of Main Loading into Ecosystem	Source of Loading	New Concentration After Loading	Duration of New Concentration	Additional Notes	Reference
Streams [1]										
water quality	Niagara River at Niagara-on-the-Lake, ON	chloride and sodium concentration greater in February than in August, November or May	see Duration of New Concentration	10.1-10.7 mg/L Na^+; 20.2-21.9 mg/L Cl^- (based on one week in each of August 1975, November 1975,and May, 1976)	unknown	winter use of road salt *suspected	11.2-12.4 mg/L Na^+; 22.0-23.7 mg/L Cl^- (February 11-18, 1976)	at least one week in February	length of stream affected unknown	Chan and Clignett 1978
water quality	Etobicoke Creek, ON	increased chloride concentration	see Duration of New Concentration	0-43 mg/L Cl^-	unknown	urban road runoff	2,140-3,780 mg/L Cl^- (max.); 278-392 mg/L Cl^- (mean)	January to March (highest concentration in February)	all 3 stations located in developed areas near roads	OMEE
water quality	Mimico Creek, ON	increased chloride concentration	see Duration of New Concentration	51 mg/L Cl^-	unknown	unknown	3,470 mg/L Cl^- (maximum); 553 mg/L Cl^- (mean)	January - March (highest conc. in December)	-	OMEE
water quality	Highland Creek, ON	increased chloride concentration	see Duration of New Concentration	22 mg/L Cl^-	unknown	unknown	1,390 mg/L Cl^- (maximum); 310 mg/L Cl^- (mean)	January - March (highest conc. in December)	-	OMEE
water quality	Black Creek, ON	increased chloride concentration	see Duration of New Concentration	20 mg/L Cl^-	unknown	unknown	4,310 mg/L Cl^- (maximum); 495 mg/L Cl^- (mean)	January - March (highest conc. in December)	-	OMEE
composition and diversity of stream invertebrates	Humber River, northwest Toronto, ON	no affect; road salting in area was lower than usual during time of study	unknown	13.8-24.1 mg/L Cl^- (Albion Hills; February and March, 1980	unknown	winter use of road salt	17.0-34.8 mg/L Cl^- (Cedar Mills; February and March, 1980)	unknown	-	Kersey 1981

Table A-2: Continued

Ecological Characteristic or Species Affected	Specific Location	Response to Salt Loading	Duration of Response	Baseline or Upstream Concentrations	Value of Main Loading into Ecosystem	Source of Loading	New Concentration After Loading	Duration of New Concentration	Additional Notes	Reference
water quality	Black Creek, ON (northern boundary of Metropolitan Toronto, ON)	increased concentration of chloride	see Duration of New Concentration	50-100 mg/L Cl^- (autumn)	590 tonnes Cl^- from street salting (November 1974 to April 1975)	deicing salts applied in the winter	maximum = ~250 mg/L Cl^- (June 1974; March, May and June 1975)	high concentrations associated with thaw periods in March, May and June; dilution in April with high spring discharges; concentrations stay elevated during summer, but not as high as in winter/spring	at least 3 km (approximated from map)	Scott 1980b
water quality	Don River, ON (northern boundary of Metropolitan Toronto, ON)	increased concentration of chloride	see Duration of New Concentration	100-150 mg/L Cl^- (autumn)	1,112 tonnes Cl^- from street salting (November 1974 to April 1975)	deicing salts applied in the winter	maximum = >1000 mg/L Cl^-	high concentrations associated with thaw periods from Dec. to June; dilution in April with high spring discharge; concentrations stay elevated during summer, but not as high as in winter/spring	at least 4 km (approximated from map)	Scott 1980b

Table A-2: Continued

Ecological Characteristic or Species Affected	Specific Location	Response to Salt Loading	Duration of Response	Baseline or Upstream Concentrations	Value of Main Loading into Ecosystem	Source of Loading	New Concentration After Loading	Duration of New Concentration	Additional Notes	Reference
drift of stream benthic invertebrates	Lutteral Creek, southern ON	increased drift after 1000 mg/L Cl^- in creek	6 hours; drift returned to normal levels after concentration fell below 800 mg/L Cl^-	<100 mg/L Cl^-	unknown	experi-mental application	increase up to 2165 mg/L Cl- followed by decline	approx. 20 hours	length of stream affected unknown	Crowther & Hynes 1977
water quality	Rideau River, ON	increased chloride concentration	see Duration of New Concentration	unknown	65,500 tonnes	road salt	8-57 mg/L Cl^- (mean = 19 mg/L Cl^-)	chloride concentrations were elevated during periods of thaw and heavy salting	samples taken from rooftops had almost no chloride, indicating that chloride was due to road salt	Oliver et al. 1974
water quality	Ottawa River, ON	increased chloride concentration	see Duration of New Concentration	unknown	65,500 tonnes	road salt	6-21 mg/L Cl^- (mean = 9 mg/L Cl^-)	see above	see above	Oliver et al. 1974
water quality	Don River Watershed, ON	increased chloride concentration	see Duration of New Concentration	unknown	54,760 tonnes	highway deicing salt runoff; snow dump; sewage treatment plants	49.5-1,100 mg/L Cl^-; 23,050 tonnes removed from watershed	November to April	sampling conducted November 1978 to April 1979; other sources of Cl^- removal occurred in Ashbridge's Bay S.T.P. (3540 tonnes) and snow dumping (330 tonnes)	Paine 1979

Table A-2: Continued

Ecological Characteristic or Species Affected	Specific Location	Response to Salt Loading	Duration of Response	Baseline or Upstream Concentrations	Value of Main Loading into Ecosystem	Source of Loading	New Concentration After Loading	Duration of New Concentration	Additional Notes	Reference
macro-invertebrate community structure	20 springs in southeastern ON	several taxa associated with high Cl^- levels (e.g., Tipulidae & Ceratopo-gonidae), whereas others (e.g., *Gammarus pseudo-limnaeus* & Turbellaria) found only in springs with low Cl^-	unknown	9.3 mg/L Cl^- (pristine spring at Glen Major Conservation Area)	unknown	ground-water believed to be contami-nated with Cl^- from road salt	8.1-1148.6 mg/L Cl^- (for 19 springs)	unknown	length of stream affected unknown; Cl^- concentrations correlated with level of urbanization	Williams et al. 1997; Williams et al. 1999
water quality	Rouge River, ON	increased chloride concentration	unknown	11-37 mg/L Cl^-	unknown	unknown	11-970 mg/L Cl^- (max.); 50-162 mg/L Cl^- (mean)	unknown	unknown	OMEE
water quality	Brown Deer Creek (urban) and Trinity Creek (rural), Milwaukee, WI, USA	increased load of chloride and sodium in urban versus rural creek	see Duration of New Concentration	0.6 kg Na^+/km^2 and 0.7 kg/km^2 Cl^- of drainage area (October 6-8, 1974; Trinity Creek -rural with very low road salt application)	unknown	road salt	25.7 kg/km^2 Na^+; 37.5 kg /km^2 Cl^- of drainage area (October 6 to 8, 1974; Brown Dear Creek - urban with much greater road salt application)	less than 1 day; however, last snow storm 7 months earlier indicating that heavy salt residues can still dominate chemistry of surface waters after extended	length of stream affected unknown	Cherkauer 1975

								periods of time		

Table A-2: Continued

Ecological Characteristic or Species Affected	Specific Location	Response to Salt Loading	Duration of Response	Baseline or Upstream Concentrations	Value of Main Loading into Ecosystem	Source of Loading	New Concentration After Loading	Duration of New Concentration	Additional Notes	Reference
water quality	ditch draining to a stream near Jamesville, NY, USA (south of Lake Ontario by Syracuse, NY)	chloride concentration increase, particularly with increases in rain and/or temperature	see Duration of New Concentration	~20 mg/L Cl^-	unknown, but higher concen-tration with increasing rain and/or temperature suggest higher loadings	highway runoff	mean = 448 mg/L Cl^-; maximum = 5500 mg/L Cl^-; highly variable over course of 1 day	chloride concentrations can vary significantly in a matter of hours	length of ditch affected unknown	Champagne 1978
water quality	Irondequoit Creek, Rochester, NY, USA	increased chloride concentration	see Duration of New Concentration	~15 mg/L Cl^-	77,000 metric tons applied to drainage basin during 1969-70	urban road runoff	360 mg/L Cl^-	unknown	approx. one half of the 77,000 metric tons applied to drainage basin stored in soil and groundwater	Bubeck et al. 1971
diversity of aquatic insects colonizing artificial substrates	4 streams by town of Newcomb in Adirondack region of northern NY, USA	decreased diversity in downstream versus upstream locations	unknown	0.61 mg/L Cl^- (overall mean)	unknown	winter use of road salt	5.23 mg/L Cl^- (overall mean)	unknown	smaller flow rate, higher downstream Cl^- concentration, and greater difference in diversity between upstream and downstream	Demers 1992
water quality	Flat Brook Creek, Newcomb, NY, USA	increased chloride concentration	see Duration of New Concentration	1.13 mg/L Cl^-	44 tonnes (1986-87); 41 tonnes (1987-88)	highway deicing salt runoff	1.70 mg/L (50 m downstream); 1.77 mg/L (100 m	chloride levels elevated until July 1987 and September 1988	stream flow rate 0.01-0.22 m^3/s; total length 0.6-3.2 km; Route 28N runs parallel to	Demers and Sage 1990

							downstream)		streams (2 km)	

Table A-2: Continued

Ecological Characteristic or Species Affected	Specific Location	Response to Salt Loading	Duration of Response	Baseline or Upstream Concentrations	Value of Main Loading into Ecosystem	Source of Loading	New Concentration After Loading	Duration of New Concentration	Additional Notes	Reference
water quality	No Name Creek, Newcomb, NY, USA	increased chloride concentration	see Duration of New Concentration	0.53 mg/L Cl^-	44 tonnes (1986-87); 41 tonnes (1987-88)	highway deicing salt runoff	17.05 mg/L (50 m downstream); 17.03 mg/L (100 m downstream)	chloride levels elevated until July 1987 and September 1988	stream flow rate 0.01-0.22 m^3/s; total length 0.6-3.2 km; Route 28N runs parallel to streams (2 km)	Demers and Sage 1990
water quality	Plantation Creek, Newcomb, NY, USA	increased chloride concentration	see Duration of New Concentration	0.53 mg/L Cl^-	44 tonnes (1986-87); 41 tonnes (1987-88)	highway deicing salt runoff	7.21 mg/L (50 m downstream); 7.54 mg/L (100 m downstream)	chloride levels elevated until July 1987 and September 1988	stream flow rate 0.01-0.22 m^3/s; total length 0.6-3.2 km; Route 28N runs parallel to streams (2 km)	Demers and Sage 1990
water quality	CCC Creek, Newcomb, NY, USA	increased chloride concentration	see Duration of New Concentration	0.51 mg/L Cl^-	44 tonnes (1986-87); 41 tonnes (1987-88)	highway deicing salt runoff	3.73 mg/L (50 m downstream); 3.58 mg/L (100 m downstream)	chloride levels elevated until July 1987 and September 1988	stream flow rate 0.01-0.22 m^3/s; total length 0.6-3.2 km; Route 28N runs parallel to streams (2 km)	Demers and Sage 1990
water quality	Mohawk River, NY	increased sodium and chloride concentration	unknown	7.9 mg/L Na^+; 8.3 mg/L Cl^-	4950 $(kg/km^2)yr$; 7450 $(kg/km^2)yr$	road salt, sewage and precipitation	13.6 mg/L Na^+; 20.4 mg/L Cl^-	unknown	data collected October 1951 to September 1953 and October 1970 to September 1974	Peters and Turk 1981
Lakes[2]										
water quality and vertical mixing	Little Round Lake, ON	incomplete vertical mixing	unknown	unknown	unknown	winter application of road salt	monolimnion concentration 58.4 mg/L Na^+, 103.7 mg/L Cl^-	approximately 30 years	surface area = 7.4 ha, maximum depth = 16.8 m	Smol et al. 1983

Table A-2: Continued

Ecological Characteristic or Species Affected	Specific Location	Response to Salt Loading	Duration of Response	Baseline or Upstream Concentrations	Value of Main Loading into Ecosystem	Source of Loading	New Concentration After Loading	Duration of New Concentration	Additional Notes	Reference
water quality	Lake Erie, ON	increased Cl^- concentration with increased loadings, but decreased Cl^- concentration with remediation	see Duration of New Concentration	10 mg/L Cl^- (year = 1910)	unknown	Detroit River and other tributaries that received road salt and urban sewage	>25 mg/L Cl^- (late 1960s); 20mg/L Cl^- (1990 - after remediation)	~50 years to increase from 10->25 mg/L Cl^-; ~20 years to decrease from >25-20 mg/L Cl^-; lake has short retention time (2.6 years)	-	Moll et al. 1992
water quality	Lake Ontario, ON	increased chloride concentration with increasing loadings	see Duration of New Concentration	<10 mg/L Cl^- around turn of the century	unknown	Niagara River receiving road salt and urban sewage	>25 mg/L Cl^- (late 1960s); no decline in recent years	~60 years to increase to >25 mg/L Cl^-, with no decline (as of 1992); this lake has longer retention time (6 years) than Lake Erie	-	Moll et al. 1992
water quality	Lake Huron, ON	increased chloride concentration with increased loadings, but decreased concentration with remediation	see Duration of New Concentrations	5 mg/L Cl^- (year = 1900)	unknown	Saginaw Bay tributaries	7 mg/L Cl- (late 1960s); 5 mg/L Cl- (~1990 - after remediation)	~50 years to increase from 5-7 mg/L Cl^-; ~20 years to decrease from 7-5 mg/L Cl^-	-	Moll et al. 1992

Table A-2: Continued

Ecological Characteristic or Species Affected	Specific Location	Response to Salt Loading	Duration of Response	Baseline or Upstream Concentrations	Value of Main Loading into Ecosystem	Source of Loading	New Concentration After Loading	Duration of New Concentration	Additional Notes	Reference
incomplete spring vertical mixing (salt-induced density gradient; changes in benthic invertebrate density	Lake Wabekayne, a storm-water impoundment for Mississauga, ON	anoxic conditions in bottom waters and decreased benthic invertebrate diversity	primarily in spring	50 mg/L Cl^- in August 1979	unknown	urban road salt application in winter	282 mg/L Cl^- in February 1979	concentrations elevated for approx. 3 months (February-April), especially on bottom	surface area = 1.9 ha; mean depth = 1.84 m; volume = 3.5×10^4 m^3	Free and Mulamootti 1983
water quality	8 lakes in the Humber River Watershed, ON	increased chloride concentration	unknown	unknown	unknown	suspected road salt	10.6-408.9 mg/L	unknown	Humber River Watershed is a major urban catchment basin located east of Metropolitan Toronto	D. Scanlon 1999 (letter)
water quality	4 Metropolitan Toronto detention ponds	increased nutrient, metal and ionic concentration	see Duration of New Concentration	unknown	unknown	urban runoff	22-345 mg/L Cl^- (Heritage Pond), 28-1,201 mg/L Cl^- (Unionville Pond), 36-617 mg/L Cl^- (S. Smith Pond), 59-216 mg/L Cl^- (Tapscott Pond)	highest concentrations observed during snow melt and spring storm runoff	runoff from industrial and commercial land and from the Queen Elizabeth Way	Mayer et al. 1996

Table A-2: Continued

Ecological Characteristic or Species Affected	Specific Location	Response to Salt Loading	Duration of Response	Baseline or Upstream Concentrations	Value of Main Loading into Ecosystem	Source of Loading	New Concentration After Loading	Duration of New Concentration	Additional Notes	Reference
water quality	3 Ontario highways: Skyway Bridge, Highway #2 and Plains Road	increased chloride concentration	see Duration of New Concentration	unknown	unknown	highway deicing salt runoff	45-10,960 mg/L Cl^- (Skyway Bridge), 28-1,480 mg/L Cl^- (Highway #2) 4-9,350 mg/L Cl^- (Plains Road)	highest concentrations observed between February and April	greatest concentrations observed in runoff associated with the busiest highway (Skyway Bridge)	Mayer et al. 1998
water quality	Lac a la Truit, PQ	increased chloride concentration	see Duration of New Concentration	12 mg/L Cl^-	unknown	road runoff from Highway 15 near Sainte-Agathe-des-Monts	150 mg/L Cl^- (max. in 1979); fell to 45 mg/L Cl^- in 1990	maximum concentration of 150 mg/L Cl^- occurred in 1979; declined during the 1980s reaching 45 mg/L Cl^- in 1990	surface area = 48.6 ha; mean depth = 21.5 m; volume = 486,000 m^3	Ministry of Transport Quebec
water quality	Mirror Lake, NH, USA	increased chloride and sodium concentration	see Duration of New Concentration	0.94 mg/L Cl^- 1.22 mg/L Na^+	unknown	road runoff and leaching from septic tanks	2.04 mg/L Cl^- 1.65 mg/L Na^+	increase occurred from 1975/76 to 1979/80	surface area = 11 ha; mean depth = 5.75 m; maximum depth = 11 m	Likens 1985
water quality	42 Connecticut lakes, USA	increased chloride and sodium concentration	unknown	unknown	unknown	road runoff	increase of 70 μeq/L Cl^- (mean increase 90 μeq/L Cl^-); increase 60 μeq/L Na^+	increase occurred primarily between 1970-1990	other contributing factors are conversion of forest to agricultural land, tilling and application of pesticides and fertilizers	Siver et al. 1996

Table A-2: Continued

Ecological Characteristic or Species Affected	Specific Location	Response to Salt Loading	Duration of Response	Baseline or Upstream Concentrations	Value of Main Loading into Ecosystem	Source of Loading	New Concentration After Loading	Duration of New Concentration	Additional Notes	Reference
water quality and vertical mixing	Northridge Lakes, WI, USA (shallow and artificial, Milwaukee, WI)	increased concentration of chloride in bottom of lakes, but vertical mixing occurred	see Duration of New Concentration	6 mg/L Cl^- (average for lake water in the area)	unknown	application of NaCl and $CaCl_2$ on paved roads	at 0.4m (just below ice) = ~250 mg/L Cl^-; at 3m (bottom) = ~2500 mg/L Cl^- (March 1975)	Cl^- gradient present from March 6-20, 1975; average concentrations still much greater than baseline in October 1975 (130 mg/L Cl^-), 7 months after last salt application.	surface area = 0.19 km^2; mean depth = 2 m and estimated total volume = 3.8×10^5 m^3	Cherkauer and Ostenso 1976
water quality	Sparkling Lake WI, USA (northern Wisconsin Lake District)	increased chloride concentration due to contamination from road salt laden groundwater	see Duration of New Concentration	0.3-0.5 mg/L Cl^- (background lake and groundwater chloride in the area)	1,200 kg/yr total chloride (based on average lake increase of 0.15 mg/L chloride per year)	ground-water contami-nated with road salt (initially applied to roads above the lake, road salt then leached into the ground-water before entering lake	2.61 mg/L Cl^- in 1982; 3.68 mg/L Cl^- in 1991	slow increase recorded from 1982-1991	lake morphometry unknown	Bowser 1992

Table A-2: Continued

Ecological Characteristic or Species Affected	Specific Location	Response to Salt Loading	Duration of Response	Baseline or Upstream Concentrations	Value of Main Loading into Ecosystem	Source of Loading	New Concentration After Loading	Duration of New Concentration	Additional Notes	Reference
incomplete vertical mixing in spring due to salt-induced density gradient	First Sister Lake, MI, USA	anoxic conditions in bottom waters of Lake; density gradient results from salt-laden runoff entering from a drainage pipe and going to bottom	spring of 1965 and 1967 - these two winters had heavy snow fall and heavy salt application; mixing occurred in 1966	0m = 84 mg/L Cl^-; 4m = 82 mg/L Cl^-; 7m = 89 mg/L Cl- (spring 1966 - when turnover occurred)	unknown	road salt application in winter; salt-laden runoff enters through storm sewer pipes	0m = 61 mg/L Cl^-; 4m = 91 mg/L Cl-; 7m = 148 mg/L Cl^- (spring 1967)	unknown, but salt concentration decreased enough by fall to allow for fall mixing of the Lake	maximum depth = ~7 m	Judd 1969
complete vertical mixing in spring even though concentrations higher than in Judd 1969 (above)	First Sister Lake, MI, USA	runoff entered through wetland instead of a pipe, resulting in diffuse entry of salt and no gradient	spring 1981	0m =93.9 mg/L Cl^-; 4m =97.8 mg/L Cl^-; 6m =111.8 mg/L Cl^- (April 1981)	unknown	road salt application in winter; salt-laden runoff enters through wetland	see Baseline Concentration	unknown	maximum depth = ~7 m	Judd and Steggall 1982
incomplete vertical mixing due to salt gradient	Fonda Lake, MI, USA	increased salt concentration	unknown	12 mg/L (Frains Lake); 15 mg/L (Portage Lake)	unknown	seepage from salt storage facility	235 mg/L	unknown	asphalt pad constructed in early 1970s, reducing salt input, but still remained elevated	Tuchman et al. 1984; Zeeb and Smol 1991

Table A-2: Continued

Ecological Characteristic or Species Affected	Specific Location	Response to Salt Loading	Duration of Response	Baseline or Upstream Concentrations	Value of Main Loading into Ecosystem	Source of Loading	New Concentration After Loading	Duration of New Concentration	Additional Notes	Reference
increased vertical mixing of Ides Cove waters in spring and fall due to reduced density gradient	Ides Cover, Irondequoit Bay, Rochester, NY, USA southern shore of Lake Ontario)	increased density gradient	see Duration of New Concentration	epilimnion concentrations 210-225 mg/L Cl^-; additional 80-160 mg/L in hypolimnion	225,000 kg/cm of snowfall (1969-71), decreased to 115,000 kg/cm (1979-82)	winter application of road salt	epilimnion concentrations 140-150 mg/L Cl^-; hypolimnion concentration decreased 0-90 mg/L	1970-82	surface area = 1.18 ha, maximum depth 8.8 m; separated from Bay by 50 m wide 1.5 m deep bedrock sill	Bubeck et al. 1995
incomplete vertical mixing of Irondequoit Bay waters in spring due to salt-induced density gradient	Irondequoit Bay, Rochester, NY, USA (southern shore of Lake Ontario)	anoxic conditions in bottom waters of Bay	1970-73	in 1910: Irondequoit Creek = ~14 mg/L Cl^- and Bay = ~12 mg/L Cl^- (surface concentration in Bay)	unknown, but in winter there was a maximum of 600 mg/L Cl^- from the Creek which flows into the Bay at a mean annual discharge of 3.7 m^3/s	highway runoff into Creek enters the Bay; primary source of water to the Bay is the Creek	from 1960-80: Creek = ~90 mg/L Cl^- and Bay = ~125 mg/L Cl^- (surface concentration in Bay); maximum for Bay = 152 mg/L Cl^- in 1971	at least 20 years (1960s to 1980s)	surface area = 6.78 km^2; maximum depth = 23.8 m; mean depth = 6.8 m; volume = 45.9 x 10^6 m^3; hydraulic retention time = 116 days	Bubeck and Burton 1987; Bubeck et al. 1995
water quality	various locations throughout the USA	increased chloride concentration	see Duration of New Concentration	unknown	unknown	primarily winter application of road salt	1130-25,100 mg/L Cl^- in winter runoff	1971-73	snow and ice deposits found to contain up to 10,000 mg/L NaCl	Field and O'Shea 1992

Table A-2: Continued

Ecological Characteristic or Species Affected	Specific Location	Response to Salt Loading	Duration of Response	Baseline or Upstream Concentrations	Value of Main Loading into Ecosystem	Source of Loading	New Concentration After Loading	Duration of New Concentration	Additional Notes	Reference
Wetlands										
alterations of plant species diversity	Pinhook Bog, LaPorte County, IN (in Indiana Dunes National Lakeshore)	absence of numerous native species such as *Sphagnum* spp. and *Larix laricina* and invasion of salt tolerant species such as *Typha angustifolia*	late 1960s to 1980 (when salt storage at the site was discontinued)	5-6 mg/L Cl^- (control sites; 1980-81)	total Cl^- inputs to bog over 10 year period: from salt pile = 2.3 million kg; from road salting = 0.4 million kg; from direct precipi-tation = 0.012 million kg	road salt storage pile, road salting of nearby highway, natural precipi-tation	max. single daily reading: 1468 mg/L Cl^- in 1979; 982 mg/L in 1980; 570 mg/L in 1981	late 1960s to 1980 (when salt storage at the site was discontinued)	area = 44 ha	Wilcox 1982

[1] streams include streams, creeks, springs, and rivers.
[2] lakes include lakes and ponds.

Table A-3: West Coast (British Columbia) and Rocky Mountains and the United States

Ecological Characteristic or Species Affected	Specific Location	Response to Salt Loading	Duration of Response	Baseline or Upstream Concentrations	Value of Main Loading into Ecosystem	Source of Loading	New Concentration After Loading	Duration of New Concentration	Additional Notes	Reference
Streams [1]										
water quality of mountain streams near ski developments	Rio en Medio Stream, Sante Fe Ski Basin, Sante Fe, NM, USA	increased concentration of sodium and chloride	see Duration of New Concentration	1.77-2.05 mg/L Na^+; 0.3 mg/L Cl^- (Area 1, upstream, 1971-75, yearly averages)	2100 kg/year Na^+; 3675 kg/year Cl^- (Area 3 = 4.9 ha, downstream of area 1; data for 1972-73)	salting of roads in winter	5.02-6.30 mg/L Na^+; 11.9-17.5 mg/L Cl^- (Area 3, downstream, 1972-75, yearly average)	highest concentrations in area 3 from February to April; yearly averages greater for Area 3 than Area 1	length of affected stream unknown	Gosz 1977
water quality of mountain streams	Billy Mack Creek in the Sierra Nevada Mountains, CA, USA	increased concentration of chloride	see Duration of New Concentration	<1 mg/L Cl^- (upstream of freeway crossing)	unknown	winter salting of freeway	~20-70 mg/L Cl^- (downstream of freeway crossing)	December to April; after salt applications stopped in spring, Cl^- conc. returned to normal levels in 1 month or less	at least 1.5 km	Hoffman et al. 1981
Lakes [2]										
chemocline in mountain lakes	Putt's Lake, Gold Run Pond, & Summit Pond, Sierra Nevada Mtn, CA, USA	high concentration of chloride on bottom of lake	see Duration of New Concentration	surface=8mg/L Cl^-; bottom = 142 mg/L Cl^- (Putt's Lake, May, 1975)	unknown	runoff	see Baseline Concentrations	early spring (April and part of May); normal spring mixing occurred	Putt's Lake: maximum depth = 4 m	Hoffman et al. 1981
Wetlands										
No data were found										

[1] streams include streams, creeks, springs, and rivers.
[2] lakes include lakes and ponds.

APPENDIX B: The Effects of Road Salt and their Additives on Aquatic Organisms

Table B-1: Sodium Chloride

Species of Concern	Toxic Response[2]	Exposure Concentration[1]	Exposure Time	Water Temperature (°C)	Chemical Composition of Test Water	Source of Test Organism	Reference
Bacteria, Protozoa, and Fungi							
aquatic hyphomycetes (fungi)	increased sporulation above controls	659 mg/L NaCl (400 mg/L Cl^-)	48 hours	20	soft water	Boss Brook, Fenwick, NS	Sridhar & Barlocher 1997
Boekelovia hooglandii (euryhaline, golden-brown microflagellate alga)	optimal growth	11,690 to 23,380 mg/L NaCl (0.2 to 0.4 M)	12 days (incubation time)	25	culture medium	W.R. Barclay, University of Denver	Fuji and Hellebust 1994
Boekelovia hooglandii (euryhaline, golden-brown microflagellate alga)	growth severely inhibited	0 mg/L or >58,450 mg/L NaCl (0 or >1M)	12 days (incubation time)	25	culture medium	W.R. Barclay, University of Denver	Fuji and Hellebust 1994
Escherichia coli (bacteria)	0% RNA degradation	8,767 mg/L NaCl + 6,019 mg/L $MgSO_4$ (0.15 M NaCl + 5 mM $MgSO_4$)	2 hours (120 minutes)	30	Tris Buffer[3]	Dr. T. Beppu, University of Tokyo	Ito et al. 1977
Escherichia coli (bacteria)	50% RNA degradation	8,767 mg/L NaCl (0.15 M)	2 hours (120 minutes)	30	Tris Buffer[3]	Dr. T. Beppu, University of Tokyo	Ito et al. 1977
Paramecium tetrourelia (paramecium)	17% reduction in cell division	562 mg/L NaCl (350 mg/L Cl^-)	5 days	unknown	unknown	unknown	Cronkite et al. 1985
Euglena gracilis (protozoan)	16% reduction in number of cells cultured in light;	5,845 mg/L NaCl (0.1 M)	168 hours (7 days)	27 - 28	acidic organotrophic medium with glutamate + malate	stock cultures of wild-type	Gonzalez-Moreno et al. 1997

	38% reduction in number of cells in dark						

Table B-1: Continued

Species of Concern	Toxic Response[2]	Exposure Concentration[1]	Exposure Time	Water Temperature (°C)	Chemical Composition of Test Water	Source of Test Organism	Reference
Euglena gracilis (protozoan)	decrease in O_2 production, conc. Mg^{2+}&Ca^{2+}, and ratio of chlorophyll *a* to *b*; increase in respiration, cell volume, conc. K^+& Na^+, and Chl *a* & *b*	11,690 mg/L NaCl (0.2 M)	168 hours (7 days)	27-28	acidic organotrophic medium with glutamate + malate	stock cultures of wild-type	Gonzalez-Moreno et al. 1997
Euglena gracilis (protozoan)	80% reduction in number of cells cultured in light or dark	11,690 mg/L NaCl (0.2 M)	168 hours (7 days)	27-28	acidic organotrophic medium with glutamate + malate	stock cultures of wild-type	Gonzalez-Moreno et al. 1997
Phytoplankton							
Nitzschia linearis (diatom)	50% reduction in number of cells	2,430 mg/L NaCl	120 hours	unknown	soft dilution water	unknown	Patrick et al. 1968
Scendesmus obliquus (freshwater green alga)	decrease in dry matter, photosynthetic pigment and O_2 production; increases in respiration, soluble saccharides and proteins, as well as lipoid and proline content	7,014 and 11,690 mg/L NaCl (120 and 200 mM)	168 hours (7 days)	25±1	modified Beijerinck medium	Nile River, Egypt	Mohammed and Shaffea 1992
Scendesmus obliquus (freshwater green	decrease in cell number to	11,690 mg/L NaCl (200 mMol)	168 hours (7 days)	25±1	modified Beijerinck medium	Nile River, Egypt	Mohammed and Shaffea 1992

alga)	43.1 % of control						

Table B-1: Continued

Species of Concern	Toxic Response[2]	Exposure Concentration[1]	Exposure Time	Water Temperature (°C)	Chemical Composition of Test Water	Source of Test Organism	Reference
Chlamydomonas reinhardtii (alga)	49% growth inhibition	4,965 mg/L NaCl (3,014 mg/L Cl^-)	3-6 days	unknown	unknown	unknown	Reynoso et al. 1982
Chlorella emersonii (alga)	growth inhibition	11,249 mg/L NaCl (7,000 mg/L Cl^-)	8-14 days	unknown	unknown	unknown	Setter et al. 1982
Periphyton							
No data were found							
Macrophytes							
Myriophyllum spicatum (Eurasian millfoil)	50% reduction in dry weight	5,813-7,977 mg/L NaCl (3,617-4,964 mg/L Cl^-)	32 days	unknown	unknown	unknown	Stanley 1974
Potamogeton pectinatus (pondweed) seed	reduced germination	2,925 mg/L NaCl (1,820 mg/L Cl^-)	28 days	unknown	unknown	unknown	Teeter 1965
Potamogeton pectinatus (pondweed) 9-week old plant	reduced dry weight	2,925 mg/L NaCl (1,820 mg/L Cl^-)	32 days	unknown	unknown	unknown	Teeter 1965
Potamogeton pectinatus (pondweed) 13-week old plant	reduced shoots and dry weight	2,925 mg/L NaCl (1,820 mg/L Cl^-)	32 days	unknown	unknown	unknown	Teeter 1965
Sphagnum recurvum (bog moss)	reduced growth in length with increased Cl^- concentration (from NaCl)	0 mg/L as Cl^- = mean increase in length of 3.22 cm; 1500 mg/L as Cl^- = mean increase in length of 1.40 cm	1080 hours (45 days)	19 (with 16 hours of light)	chloride-free bog water	Pinhook Bog, La Porte, Indiana	Wilcox 1984

Table B-1: Continued

Species of Concern	Toxic Response[2]	Exposure Concentration[1]	Exposure Time	Water Temperature (°C)	Chemical Composition of Test Water	Source of Test Organism	Reference
Sphagnum fimbriatum (bog moss)	reduced growth in length with increased Cl^- concentration (from NaCl)	0 mg/L Cl^- = mean increase in length of 2.61 cm; 5000 mg/L Cl^- = mean increase in length of 0.03 cm	1800 hours (75 days)	19 (with 16 hours of light)	bog water (from an uncontaminated site)	Pinhook Bog, La Porte, Indiana	Wilcox and Andrus 1987
Zooplankton							
Ceriodaphnia dubia (water flea)	NOEC	1,296 mg/L NaCl	unknown	unknown	soft and hard waters	unknown	Cowgill and Milazzo 1990
Ceriodaphnia dubia (water flea)	50% mortality	1,794 mg/L NaCl	168 hours (7 days)	unknown	soft and hard waters	unknown	Cowgill and Milazzo 1990
Ceriodaphnia dubia (water flea)	50% mortality	2,308 mg/L NaCl	48 hours	unknown	soft and hard waters	unknown	Cowgill and Milazzo 1990
Ceriodaphnia laticaudata (water flea)	weakening	2,922 mg/L NaCl (0.05 M)	unknown	unknown	unknown	unknown	Ramult 1925 (in Anderson 1948)
Cyclops serrulatus (copepod)	maximum NaCl tolerance	394 mg/L NaCl	unknown	20	unknown	unknown	Kanygina & Lebedeva 1957 (in McKee and Wolf 1963)
Cyclops vernalis (copepod)	immobilization	6,079 mg/L NaCl (0.104 M)	unknown	20-25	Lake Erie water	unknown	Anderson et al. 1948 (in Anderson 1948; McKee and Wolf 1963)
Diaptomus oregonensis (copepod)	immobilization	3,039 mg/L M NaCl (0.052 M)	unknown	20-25	Lake Erie water	unknown	Anderson et al. 1948 (in Anderson 1948; McKee and Wolf 1963)
Daphnia longispina (water flea)	death	2,922 mg/L NaCl (0.05 M)	66 hours	unknown	well water	unknown	Fowler 1931 (in Anderson 1948)
Daphnia magna (water flea)	death	1 mg/L NaCl	3 hours	unknown	distilled	unknown	Ellis 1937 (in McKee & Wolf 1963)

Table B-1: Continued

Species of Concern	Toxic Response[2]	Exposure Concentration[1]	Exposure Time	Water Temperature (°C)	Chemical Composition of Test Water	Source of Test Organism	Reference
Daphnia magna (water flea)	maximum NaCl tolerance	200 mg/L NaCl	unknown	20	unknown	unknown	Kanygina & Lebedeva 1957 (in McKee and Wolf 1963)
Daphnia magna (water flea)	maximum NaCl tolerance	800 mg/L NaCl	unknown	3	unknown	unknown	Kanygina & Lebedeva 1957 (in McKee and Wolf 1963)
Daphnia magna (water flea)	reproductive impairment	1,707 mg/L NaCl (1,062 mg/L Cl^-)	21 days	unknown	unknown	unknown	Biesinger and Christensen 1972
Daphnia magna (water flea)	50% mortality	4,226 mg/L NaCl (2,565 mg/L Cl^-)	48 hours	unknown	unknown	unknown	Biesinger and Christensen 1972
Daphnia magna (water flea)	50% mortality	1,470 mg/L NaCl	unknown	unknown	stream water	unknown	Birge et al. 1985
Daphnia magna (water flea)	immobilization	2,922 mg/L NaCl (0.05 M)	unknown	unknown	soft and hard water	unknown	Naumann 1934 (in Anderson 1948)
Daphnia magna (water flea)	threshold toxicity	3,170 mg/L NaCl	unknown	unknown	dissolved oxygen = 1.48 mg/L	unknown	Fairchild 1955 (in McKee and Wolf 1963)
Daphnia magna (water flea)	immobilization (50% of test animals)	3,680 mg/L NaCl	64 hours	25	Lake Erie water	unknown	Anderson 1948
Daphnia magna (water flea) (adult)	threshold toxicity	4,600 mg/L NaCl	unknown	20-25	Lake Erie water	unknown	Anderson et al. 1948 (in McKee and Wolf 1963)
Daphnia magna (water flea)	50% mortality	4,746$\pm$170 mg/L NaCl (81.2$\pm$2.9 mM/L)	48 hours	20	5ml infusion + test solution + synthetic water (diluted to 100ml)	laboratory culture	Arambasic et al. 1995

Table B-1: Continued

Species of Concern	Toxic Response[2]	Exposure Concentration[1]	Exposure Time	Water Temperature (°C)	Chemical Composition of Test Water	Source of Test Organism	Reference
Daphnia magna (water flea)	threshold toxicity	5,093 mg/L NaCl	unknown	unknown	dissolved oxygen = 6.4 mg/L	unknown	Fairchild 1955 (in McKee and Wolf 1963)
Daphnia magna (water flea)	death	5,494 mg/L NaCl (0.094 M)	unknown	unknown	unknown	unknown	Warren 1899 (in Anderson 1948)
Daphnia magna (water flea)	NOEC	1,296 mg/L NaCl	unknown	unknown	soft and hard water	unknown	Cowgill and Milazzo 1990
Daphnia magna (water flea)	50% mortality	5,777 mg/L NaCl	168 hours (7 days)	unknown	soft and hard water	unknown	Cowgill and Milazzo 1990
Daphnia magna (water flea)	50% mortality	7,754 mg/L NaCl	48 hours	unknown	soft and hard water	unknown	Cowgill and Milazzo 1990
Daphnia pulex (water flea)	50% mortality	1470 mg/L NaCl	48 hours	unknown	stream water	unknown	Birge et al. 1985
Daphnia pulex (water flea)	50% mortality	3100 mg/L NaCl	48 hours	unknown	reconstituted water	unknown	Birge et al. 1985
Daphnia pulex (water flea)	failure to develop	5,845 mg/L NaCl (0.1 M)	unknown	unknown	pond water	unknown	Ramult 1925 (in Anderson 1948)
Leptodora kindtii (giant water flea)	immobilization	3,682 mg/L NaCl (0.063 M)	unknown	20-25	Lake Erie water	unknown	Anderson et al. 1948 (in Anderson 1948; McKee and Wolf 1963)
Benthic Invertebrates							
Anaobolia nervosa (caddisfly larvae)	50% mortality	7,014 mg/L NaCl (120 mM/L)	72 hours (3 days)	14-17	tap water & sea water diluted with tap water	River Blyth, Britain	Sutcliffe 1961b
Anaobolia nervosa (caddisfly larvae)	75% mortality	9,936 mg/L NaCl (170 mM/L)	72 hours (3 days)	14-17	tap water & sea water diluted with tap water	River Blyth, Britain	Sutcliffe 1961b
Baetis tricaudatus (mayfly)	50% mortality	5330 mg/L NaCl	48 hours	unknown	stream water with velocity of 6 cm/sec	unknown	Lowell et al. 1995
Baetis tricaudatus (mayfly)	50% mortality	5440 mg/L NaCl	48 hours	unknown	stream water with velocity of 12 cm/sec	unknown	Lowell et al. 1995

Table B-1: Continued

Species of Concern	Toxic Response[2]	Exposure Concentration[1]	Exposure Time	Water Temperature (°C)	Chemical Composition of Test Water	Source of Test Organism	Reference
Caenorhabditis elegans (nematode)	mortality not significantly different from controls	15,460 mg/L NaCl (total - includes NaCl in test water)	24 hours	20	K-medium (2.36 g KCl + 3.0 g NaCl per liter distilled water)	unknown	Khanna et al. 1997
Caenorhabditis elegans (nematode)	mortality not significantly different from controls	15,500 mg/L NaCl (total - includes NaCl in test water)	96 hours	20	K-medium (2.36 g KCl + 3.0 g NaCl/L distilled water)	unknown	Khanna et al. 1997
Caenorhabditis elegans (nematode)	mortality not significantly different from controls	20,500 mg/L NaCl (total - includes NaCl in test water)	24 hours	20	moderately hard reconstituted water (96 mg $NaHCO_3$ + 60 mg $CaSO_4 \cdot 2H_2O$ + 60 mg $MgSO_4$ + 4 mg KCl per liter distilled water)	unknown	Khanna et al. 1997
Caenorhabditis elegans (nematode)	mortality not significantly different from controls	20,950 mg/L NaCl (total - includes NaCl in test water)	96 hours	20	moderately hard reconstituted water (96 mg $NaHCO_3$ + 60 mg $CaSO_4 \cdot 2H_2O$ + 60 mg $MgSO_4$ + 4 mg KCl per liter distilled water)	unknown	Khanna et al. 1997
Chironomus attenatus (chironomid)	50% mortality	7,996 mg/L NaCl (136.8 mM/L)	48 hours	25	dechlorinated, oxygenated water	laboratory populations	Thornton and Sauer 1972
Chironomus attenatus (chironomid)	50% mortality	9,995 mg/L NaCl (171 mM/L)	12 hours	25	dechlorinated, oxygenated water	laboratory populations	Thornton and Sauer 1972
Chironomus attenatus (chironomid)	100% mortality	12,000 mg/L NaCl	12 hours	25	dechlorinated, oxygenated water	Laboratory populations	Thornton and Sauer 1972
Cricotopus trifascia (chironomid)	100% mortality	8,865 mg/L NaCl (Hamilton et al. determined value from regression)	48 hours	12	filtered lake water	Lake Michigan	Hamilton et al. 1975
Culex sp. (mosquito) larvae	50% mortality	10,254 mg/L NaCl (6,222 mg/L Cl^-)	48 hours	unknown	unknown	unknown	Dowden and Bennett 1965

Table B-1: Continued

Species of Concern	Toxic Response[2]	Exposure Concentration[1]	Exposure Time	Water Temperature (°C)	Chemical Composition of Test Water	Source of Test Organism	Reference
Dreissena polymorpha (zebra mussel)	Veligers = 100% mortality; settlers = 70% mortality	10,000 mg/L NaCl	6 hours	17	hardness = 140±10 mg/L as $CaCO_3$	Lake Michigan	Waller et al. 1996
Dreissena polymorpha (zebra mussel)	Veligers = 100% mortality; settlers = 98% mortality	10,000 mg/L NaCl	24 hours	12	hardness = 140±10 mg/L as $CaCO_3$	Lake Michigan	Waller et al. 1996
Dreissena polymorpha (zebra mussel)	Veligers = 100% mortality; settlers = 99% mortality	20,000 mg/L NaCl	6 hours	17	hardness = 140±10 mg/L as $CaCO_3$	Lake Michigan	Waller et al. 1996
Gammarus pseudolimnaeus (amphipod)	20% mortality	4121 mg/L NaCl (2500 mg/L Cl^-)	24 hours	unknown	static water test	Laurel Creek, Waterloo, Ontario	Crowther and Hynes (1977)
Hydropsyche betteni (caddisfly)	survival and pupate	1,319 mg/L NaCl (800 mg/L Cl^-)	240 hours (10 days)	unknown	unknown	unknown	Kersey 1981
Hydropsyche bronta (caddisfly)	survival and pupate	1,319 mg/L NaCl 800 mg/L Cl^-)	240 hours (10 days)	unknown	unknown	unknown	Kersey 1981
Hydropsyche betteni (caddisfly)	80% mortality	9,890 mg/L NaCl (6,000 mg/L Cl^-)	144 hours (6 days)	unknown	unknown	unknown	Kersey 1981
Hydropsyche slossonae (caddisfly)	survival and pupate	1,319 mg/L NaCl (800 mg/L Cl^-)	240 hours (10 days)	unknown	unknown	unknown	Kersey 1981
Hydroptila angusta (caddisfly)	100% mortality	10,136 mg/L NaCl (Hamilton et al. determined value from regression)	48 hours	12	filtered lake water	Lake Michigan	Hamilton et al. 1975
Lirceus fontinalis (isopod)	50% mortality	4,863 mg/L NaCl (2,950 mg/L Cl^-)	96 hours	unknown	reconstituted water	unknown	Birge et al. 1985
Limnephilus stigma (caddisfly)	50% mortality	7,014 mg/L NaCl (120 mM/L)	72 hours (3 days)	14-17	tap water and sea water diluted with tap water	Gosfoth Park, Northumberland, Britain	Sutcliffe 1961b

Limnephilus stigma (caddisfly)	75% mortality	9,936 mg/L NaCl (170 mM/L)	72 hours (3 days)	14-17	tap water and sea water diluted with tap water	Gosfoth Park, Northumberland, Britain	Sutcliffe 1961b

Table B-1: Continued

Species of Concern	Toxic Response[2]	Exposure Concentration[1]	Exposure Time	Water Temperature (^oC)	Chemical Composition of Test Water	Source of Test Organism	Reference
Musculium securis (clam)	reduced natality (mean # of newborns/# parents) with increasing NaCl	0 mg/L NaCl = mean natality per dish of 54.3; 1000 mg/L NaCl = mean natality per dish of 0	1440-1920 hours (60-80 days)	unknown	distilled or deionized water with air-dried soil and willow or elm leaves	Carp Pond, Ontario	Mackie 1978
Nais variabilis (oligochaete)	100% mortality	3,735 mg/L NaCl (Hamilton et al. determined value from regression)	48 hours	12	filtered lake water	Lake Michigan	Hamilton et al. 1975
oligochaete	maximum NaCl tolerance	1576 mg/L NaCl	unknown	20	unknown	unknown	Kanygina and Lebedeva 1957 (in McKee and Wolf 1963)
oligochaete	maximum NaCl tolerance	2000 mg/L NaCl	unknown	3	unknown	unknown	Kanygina and Lebedeva 1957 (in McKee and Wolf 1963)
Polycelis nigra (flatworm)	survival for 48 hours	11,109 mg/L NaCl (4370 mg/L Na^+)	48 hours	15-18	distilled water; planaria noted as surviving several weeks in distilled water	unknown	Jones 1940; 1941
Physa gyrina (snail)	50% mortality	2,540 mg/L Cl^-	96 hours	unknown	reconstituted water	unknown	Birge et al. 1985
Stictochironomus sp. (chironomid)	maximum NaCl tolerance	788 mg/L NaCl	unknown	20	unknown	unknown	Kanygina and Lebedeva 1957 (in McKee and Wolf 1963)

Stictochironomus sp. (chironomid)	maximum NaCl tolerance	1,000 mg/L NaCl	unknown	3	unknown	unknown	Kanygina and Lebedeva 1957 (in McKee and Wolf 1963)

Table B-1: Continued

Species of Concern	Toxic Response[2]	Exposure Concentration[1]	Exposure Time	Water Temperature (°C)	Chemical Composition of Test Water	Source of Test Organism	Reference
Fish							
Exposure time = short (24 hours or less)							
Carassius auratus (goldfish)	killed or immobilized	11,765 mg/L NaCl	17 hours	unknown	distilled	unknown	Ellis 1937 (in McKee and Wolf 1963)
Carassius auratus (goldfish)	50% mortality	13,480 mg/L NaCl (8,388 mg/L Cl^-)	24 hours	unknown	unknown	unknown	Dowden and Bennett 1965
Carassius auratus (goldfish)	death	35,100 mg/L NaCl	0.47-0.63 hours	~21	unknown	unknown	Powers 1917 (in Hammer 1977; Doudoroff and Katz 1953)
Catla catla, Labeo rohoto, Cirrhinius mrigalo (three species of Indian carp fry)	50% mortality	7,500 mg/L NaCl	24 hours	26	unknown	unknown	Gosh and Pal 1969 (in Hammer 1977)
Ictalurus punctatus (channel catfish)	0% mortality	10,000 mg/L NaCl	24 hours	12 and 17 (two different experiments)	hardness = 140$\pm$10 mg/L $CaCO_3$	Upper Mississippi Science Center in La Crosse, Wisconsin	Waller et al. 1996
Ictalurus punctatus (channel catfish)	100% mortality	20,000 mg/L NaCl	6 hours	17	hardness = 140$\pm$10 mg/L $CaCO_3$	Upper Mississippi Science Center in La Crosse, Wisconsin	Waller et al. 1996
Lepomis macrochirus (bluegill sunfish)	50% mortality	14,194 mg/L NaCl (8,616 mg/L Cl^-)	24 hours	unknown	unknown	unknown	Dowden and Bennett 1965
Lepomis macrochirus (bluegill sunfish)	100% mortality	14,800 mg/L NaCl	6 hours	18.8-20.1	Chautauqua Lake water as control and dilution	Zetts Fish Hatchery, West	Kszos et al. 1990

Young of the Year					water	Virginia		

Table B-1: Continued

Species of Concern	Toxic Response[2]	Exposure Concentration[1]	Exposure Time	Water Temperature (°C)	Chemical Composition of Test Water	Source of Test Organism	Reference
Lepomis macrochirus (bluegill sunfish)	0% mortality	10,000 mg/L NaCl	24 hours	12 and 17 (two different experiments)	hardness = 140±10 mg/L $CaCO_3$	Upper Mississippi Science Center in La Crosse, Wisconsin	Waller et al. 1996
Lepomis macrochirus (bluegill sunfish)	50% mortality	14,100 mg/L NaCl	24 hours	unknown	synthetic river water	unknown	Abegg 1949; 1950 (in Doudoroff and Katz 1953)
Lepomis macrochirus (bluegill sunfish)	47% mortality	20,000 mg/L NaCl	6 hours	17	hardness = 140±10 mg/L $CaCO_3$	Upper Mississippi Science Center in La Crosse, Wisconsin	Waller et al. 1996
Micropterus dolomieu (smallmouth bass)	3.3% mortality	10,000 mg/L NaCl	24 hours	12	hardness = 140±10 mg/L $CaCO_3$	Upper Mississippi Science Center in La Crosse, Wisconsin	Waller et al. 1996
minnows	killed or immobilized	10,000 mg/L NaCl	6 hours	unknown	distilled	unknown	LeClerc 1960; LeClerc and Devlaminck 1950; 1955 (in McKee and Wolf 1963)
minnows	killed or immobilized	11,500-12,000 mg/L NaCl	6 hours	unknown	hard	unknown	LeClerc 1960; LeClerc and Devlaminck 1950; 1955 (in McKee and Wolf 1963)
Notemigonus crysoleucas (golden shiners)	average survival time	15,000 mg/L NaCl	4.73 hours	22-22.5	aerated, distilled water + tap water	unknown	Wiebe et al. 1934

Notemigonus crysoleucas (golden shiners)	average survival time	20,000 mg/L NaCl	1.33 hours	22-22.5	aerated, distilled water + tap water	unknown	Wiebe et al. 1934

Table B-1: Continued

Species of Concern	Toxic Response[2]	Exposure Concentration[1]	Exposure Time	Water Temperature (°C)	Chemical Composition of Test Water	Source of Test Organism	Reference
Oncorhynchus mykiss (rainbow trout)	0% mortality	10,000 mg/L NaCl	24 hours	12 and 17 (two different experiments)	hardness = 140±10 mg/L $CaCO_3$	Upper Mississippi Science Center in La Crosse, Wisconsin	Waller et al. 1996
Oncorhynchus mykiss (rainbow trout)	40% mortality	20,000 mg/L NaCl	6 hours	17	hardness = 140±10 mg/L $CaCO_3$	Upper Mississippi Science Center in La Crosse, Wisconsin	Waller et al. 1996
Orizias latipes (small freshwater cyprinodont)	death	14,612-29,224 mg/L NaCl (0.25 - 0.5 M)	24 hours	unknown	unknown	unknown	Iwao 1936 (in Doudoroff and Katz 1953)
Perca flavescens (yellow perch)	0% mortality	10,000 mg/L NaCl	24 hours	12 and 17 (two different experiments)	hardness = 140±10 mg/L $CaCO_3$	Upper Mississippi Science Center in La Crosse, Wisconsin	Waller et al. 1996
Pimephales promelas (fathead minnow)	0% mortality	10,000 mg/L NaCl	24 hours	12 and 17 (two different experiments)	hardness = 140±10 mg/L $CaCO_3$	Upper Mississippi Science Center in La Crosse, Wisconsin	Waller et al. 1996
Pimephales promelas (fathead minnows)	100% mortality	20,000 mg/L NaCl	6 hours	17	hardness = 140±10 mg/L $CaCO_3$	Upper Mississippi Science Center in La Crosse, Wisconsin	Waller et al. 1996
Salmo gairdneri (rainbow trout)	50% mortality	5,496 mg/L NaCl (3,336 mg/L Cl^-)	24 hours	unknown	unknown	unknown	Kostecki and Jones 1983

Table B-1: Continued

Species of Concern	Toxic Response[2]	Exposure Concentration[1]	Exposure Time	Water Temperature (°C)	Chemical Composition of Test Water	Source of Test Organism	Reference
Salmo trutta (brown trout)	0% mortality	10,000 mg/L NaCl	24 hours	12	hardness = 140±10 mg/L $CaCO_3$	Upper Mississippi Science Center in La Crosse, Wisconsin	Waller et al. 1996
Salvelinus fontinalis (brook trout)	survival and recovery	30,000 mg/L NaCl	0.5-1 hour	unknown	unknown	unknown	Phillips 1944
Salvelinus fontinalis (brook trout)	50% mortality	50,000 mg/L NaCl	0.25 hour (15 minutes)	unknown	unknown	unknown	Phillips 1944
Salvelinus namaycush (lake trout)	0% mortality	10,000 mg/L NaCl	24 hours	12	hardness = 140±10 mg/L $CaCO_3$	Upper Mississippi Science Center in La Crosse, Wisconsin	Waller et al. 1996
Stizostedion vitreum (walleye)	0% mortality	10,000 mg/L NaCl	24 hours	12 and 17 (two different experiments)	hardness = 140±10 mg/L $CaCO_3$	Upper Mississippi Science Center in La Crosse, Wisconsin	Waller et al. 1996
Exposure time = long (greater than 24 hours)							
Anguilla japonica (American eel) young	survival	11,690 mg/L NaCl (0.2 M)	50 hours	20-22	unknown	unknown	Oshima 1931 (in Doudoroff and Katz 1953)
Anguilla rostrata (American eel) glass eel stage	50% mortality	17,880 mg/L NaCl	96 hour	unknown	unknown	unknown	Hinton and Eversole 1978
Anguilla rostrata (American eel) black eel stage	50% mortality	21,450 mg/L NaCl	96 hour	unknown	unknown	unknown	Hinton and Eversole 1978
Morone sp. (bass)	0% mortality	14,000 mg/L NaCl	336 hours (14 days)	unknown	unknown	unknown	Black 1950 (in Hanes et al. 1970)

Carassius auratus (goldfish)	survival	5,000 mg/L NaCl	240 hours	unknown	unknown	unknown	Ellis 1937 (in Hammer 1977)

Table B-1: Continued

Species of Concern	Toxic Response[2]	Exposure Concentration[1]	Exposure Time	Water Temperature (°C)	Chemical Composition of Test Water	Source of Test Organism	Reference
Carassius auratus. (goldfish)	survival (unharmed)	5,000 mg/L NaCl	600 hours (25 days)	unknown	Mississippi River water	unknown	Ellis 1937 (in Hanes et al. 1970; Anderson 1948)
Carassius auratus (goldfish)	50% mortality	7,341 mg/L NaCl	96 hours	25	80% deionized water supplemented with lab water and buffering solution	Ozark Fisheries Inc., Stoutland, Missouri	Adelman et al. 1976
Carassius auratus. (goldfish)	death	10,000 mg/L NaCl	240 hours or less (10 days or less)	unknown	Mississippi River water	unknown	Ellis 1937 (in Hanes et al. 1970; Anderson 1948)
Carassius auratus (goldfish)	death	11,700 mg/L NaCl	17-154 hours	~21	unknown	unknown	Powers 1917 (in Hammer 1977; Doudoroff and Katz 1953)
Catla catla, *Labeo rohoto*, *Cirrhinius mrigalo* (three species of Indian carp fry)	50% mortality	6,000 mg/L NaCl	48 hours	unknown	unknown	unknown	Gosh and Pal 1969 (in Hammer 1977)
Gambusia affinis (mosquito-fish)	50% mortality	17,500 mg NaCl	96 hours	unknown	turbid	unknown	Wallen et al. 1957 (in Hammer 1977; McKee and Wolf 1963)
Lepomis macrochirus (bluegill sunfish)	50% mortality	5,840 mg/L NaCl	96 hours	unknown	reconstituted water	unknown	Birge et al. 1985
Lepomis macrochirus (bluegill sunfish Young of the Year	50% mortality	12,200 mg/L NaCl	288 hours (12 days)	18.8-20.1	Chautauqua Lake water as control and dilution water	Zetts Fish Hatchery, West Virginia	Kszos et al. 1990
Lepomis macrochirus (bluegill sunfish)	50% mortality	12,964 mg/L NaCl	96 hours	unknown	aerated	unknown	Trama 1954 (in McKee and

							Wolf 1963)

Table B-1: Continued

Species of Concern	Toxic Response[2]	Exposure Concentration[1]	Exposure Time	Water Temperature (°C)	Chemical Composition of Test Water	Source of Test Organism	Reference
Lepomis macrochirus (bluegill sunfish)	50% mortality	12,946 mg/L NaCl	96 hours	18 ± 2	soft dilution water	unknown	Patrick et al. 1968
Micropterus salmoides (largemouth black bass)	0% mortality	5,000 mg/L NaCl	200-250 hours	22-22.5	aerated, distilled water + tap water	unknown	Wiebe et al. 1934
Micropterus salmoides (largemouth black bass)	100% mortality	10,000 mg/L NaCl	142-148 hours	22-22.5	aerated, distilled water + tap water	unknown	Wiebe et al. 1934
Notemigonus crysoleucas (golden shiners)	average survival time	5,000 mg/L NaCl	148 hours	22-22.5	aerated, distilled water + tap water	unknown	Wiebe et al. 1934
Notemigonus crysoleucas (golden shiners)	average survival time	10,000 mg/L NaCl	97 hours	22-22.5	aerated, distilled water + tap water	unknown	Wiebe et al. 1934
Notropis atherinoides (lake emerald shiner)	threshold toxicity	2,500 mg/L NaCl	120 hours	18	unknown	unknown	Van Horn et al. 1949
Notropis blennius (river shiner)	death	2,500 mg/L NaCl	216 - 576 hours	room temperature	distilled water	unknown	Garrey 1916 (in Hammer 1977; Doudoroff and Katz 1953)
Notropis spilopterus (spotfin shiner)	threshold toxicity	2,500 mg/L NaCl	120 hours	18	unknown	unknown	Van Horn et al. 1949
Oncorhynchus mykiss (rainbow trout) eggs	2.5-4.1% mortality	250, 500, 1000, 2000 mg/L NaCl	168 hours (7 days)	unknown	unknown	unknown	Beak 1999
Oncorhynchus mykiss (rainbow trout) fingerlings	survival (no effect)	800 mg/L NaCl	196 hours (~8 days)	15-16	soft water (23 mg/L $CaCO_3$)	Spanish ICONA trout hatchery	Camargo and Tarazona 1991
Oncorhynchus mykiss (rainbow trout) embryo	10.8-18.5% nonviable embryo	control up to 1000 mg/L NaCl	27 day embryo-alvin test	unknown	unknown	unknown	Beak 1999

Table B-1: Continued

Species of Concern	Toxic Response[2]	Exposure Concentration[1]	Exposure Time	Water Temperature (°C)	Chemical Composition of Test Water	Source of Test Organism	Reference
Oncorhynchus mykiss (rainbow trout) embryo	31% nonviable embryo	2000 mg/L NaCl	27 day embryo-alvin test	unknown	unknown	unknown	Beak 1999
Oncorhynchus mykiss (rainbow trout) embryo	90.8% nonviable embryo	4000 mg/L NaCl	27 day embryo-alvin test	unknown	unknown	unknown	Beak 1999
Oncorhynchus mykiss (rainbow trout) eggs	86.3% mortality	4000 mg/L NaCl	168 hours (7 days)	unknown	unknown	unknown	Beak 1999
Oncorhynchus mykiss (rainbow trout) embryo	100% nonviable embryo	8000 mg/L NaCl	27 day embryo-alvin test	unknown	unknown	unknown	Beak 1999
Oncorhynchus mykiss (rainbow trout) eggs	100% mortality	8000 mg/L NaCl	168 hours (7 days)	unknown	unknown	unknown	Beak 1999
Perca flavescens (yellow perch)	survival	gradual increase from 9,100 mg/L to 17,500 mg/L NaCl	720 hours (1 month)	unknown	unknown	unknown	Young 1923 (in Hanes et al. 1970)
Perca flavescens (yellow perch)	0% mortality	14,000 mg/L NaCl	336 hours (14 days)	unknown	unknown	unknown	Black 1950 (in Hanes et al. 1970)
Pimephales promelas (fathead minnow) fry	smaller size	734 mg/L NaCl and 1,057 mg/L NaCl	33 days	unknown	unknown	unknown	Birge et al. 1985
Pimephales promelas (fathead minnow) eggs	80% mortality	1,054 - 1,060 mg/L NaCl	33 days	unknown	unknown	unknown	Birge et al. 1985
Pimephales promelas (fathead minnow) larvae <24 hours	NOEC	2,000 mg/L NaCl	168 hours (7 days)	unknown	unknown	unknown	Beak 1999
Pimephales promelas (fathead minnow) larvae <24 hours	10% mortality	2,000 mg/L NaCl	168 hours (7 days)	unknown	unknown	unknown	Beak 1999

Table B-1: Continued

Species of Concern	Toxic Response[2]	Exposure Concentration[1]	Exposure Time	Water Temperature (°C)	Chemical Composition of Test Water	Source of Test Organism	Reference
Pimephales promelas (fathead minnow) embryo <36 hours	90% mortality	2,000 mg/L NaCl	168 hours (7 days)	unknown	unknown	unknown	Beak 1999
Pimephales promelas (fathead minnow) larvae <24 hours	LOEC	4000 mg/L NaCl	168 hours (7 days)	unknown	unknown	unknown	Beak 1999
Pimephales promelas (fathead minnow) larvae <24 hours	surviving fish had impaired growth and swimming behavior	4,000 mg/L NaCl	168 hours (7 days)	unknown	unknown	unknown	Beak 1999
Pimephales promelas (fathead minnow) larvae <24 hours	28% mortality	4,000 mg/L NaCl	168 hours (7 days)	unknown	unknown	unknown	Beak 1999
Pimephales promelas (fathead minnow) embryo <36 hours	100% mortality	4,000 mg/L NaCl	168 hours (7 days)	unknown	unknown	unknown	Beak 1999
Pimephales promelas (fathead minnow) 1 to 7 days old	NOEC	4,000 mg/L NaCl	168 hours (7 days)	25$\pm$1	moderately hard reconstituted water	EPA Newtown Facility	Pickering et al. 1996
Pimephales promelas (fathead minnow) 1 to 7 days old	NOEC	4,000 mg/L NaCl	168 hours (7 days)	25$\pm$1	moderately hard reconstituted water	EPA Newtown Facility	Pickering et al. 1996
Pimephales promelas (fathead minnow)	50% mortality	6,570 mg/L Cl^-	96 hours	unknown	reconstituted water	unknown	Birge et al. 1985
Pimephales promelas (fathead minnows)	50% mortality (LC_{50})	7,650 mg/L NaCl	96 hours	25	80% deionized water supplimented with lab water and buffering solution	National Water Quality Laboratory, Duluth	Adelman et al. 1976

Table B-1: Continued

Species of Concern	Toxic Response[2]	Exposure Concentration[1]	Exposure Time	Water Temperature (°C)	Chemical Composition of Test Water	Source of Test Organism	Reference
Pimephales promelas (fathead minnow) larvae <24 hours	surviving fish had impaired growth and swimming behavior	8,000 mg/L NaCl	168 hours (7 days)	unknown	unknown	unknown	Beak 1999
Pimephales promelas (fathead minnow) larvae <24 hours	75% mortality	8,000 mg/L NaCl	168 hours (7 days)	unknown	unknown	unknown	Beak 1999
Pimephales promelas (fathead minnow) embryo <36 hours	100% mortality	8,000 mg/L NaCl	168 hours (7 days)	unknown	unknown	unknown	Beak 1999
Pimephales promelas (fathead minnow) 1 to 7 days old	LOEC	8,000 mg/L NaCl	168 hours (7 days)	25±1	moderately hard reconstituted water	EPA Newtown Facility	Pickering et al. 1996
Salmo trutta (brown trout) fingerlings	NOEC	1,000 mg/L NaCl	196 hours (~8 days)	15-16	soft water (23 mg/L $CaCO_3$)	Spanish ICONA trout hatchery	Camargo and Tarazona 1991
Exposure time = unknown							
Carassius auratus (goldfish)	killed or immobilized	14,000 mg/L NaCl	unknown	unknown	unknown	unknown	Jones 1957 (in McKee and Wolf 1963)
Coregonus sp. (whitefish)	killed or immobilized	3,850 mg/L NaCl	unknown	unknown	natural	unknown	Anderson 1944 (in McKee and Wolf 1963)
Coregonus clupeaformis (lake whitefish) fry	immobilization	16,500 mg/L NaCl	unknown	unknown	Lake Erie water	unknown	Edmister & Gray 1948 (in Anderson 1948; Doudoroff and Katz 1953)
Gastrosteus sp. (stickleback)	killed or immobilized	11,680-14,600 mg/L NaCl	unknown	unknown	unknown	unknown	ORVWSC 1950 (in McKee and Wolf 1963)
Stizostedion vitreum (walleye)	killed or immobilized	3,850 mg/L NaCl	unknown	unknown	natural	unknown	Anderson 1944 (in McKee and

							Wolf 1963)

Table B-1: Continued

Species of Concern	Toxic Response[2]	Exposure Concentration[1]	Exposure Time	Water Temperature (°C)	Chemical Composition of Test Water	Source of Test Organism	Reference
Stizostedion vitreum (walleye) fry	immobilization	3,859 mg/L NaCl	unknown	unknown	Lake Erie water	unknown	Edmister & Gray 1948 (in Anderson 1948; Doudoroff and Katz 1953)
Amphibians							
Microhyla ornata (frog embryos and larvae)	incomplete closure of neural tube	5,000-6,000 mg/L NaCl (0.5% - 0.6%)	96 hours	23-27	Total hardness <75 mg/L as $CaCO^3$	natural ponds/temporary rainwater pools in India	Padhye and Ghate 1992
Rana pipiens (leopard frog)	reduced maximum rate of reaction of pyruvate kinase	11,690 mg/L NaCl (200 mM)	unknown	22	unknown	commercial supplier	Grundy and Storey 1994
Scaphiopus couchii (spadefoot toad)	reduced maximum rate of reaction of pyruvate kinase	11,690 mg/L NaCl (200 mM)	unknown	22	unknown	Tucson, Arizona	Grundy and Storey 1994
Xenopus leavis (African clawed frog)	90-97% survival	250, 500, 1,000 and 2000 mg/L NaCl	168 hours (7 days)	23	unknown	unknown	Beak 1999
Xenopus leavis (African clawed frog)	50% survival	2,940 mg/L NaCl	168 hours (7 days)	23	unknown	unknown	Beak 1999
Xenopus leavis (African clawed frog)	6.7% survival	4000 mg/L NaCl	168 hours (7 days) *most mortality occurred day 2	23	unknown	unknown	Beak 1999
Xenopus leavis (African clawed frog)	0% survival	8000 mg/L NaCl	168 hours (7 days) *most mortality occurred day 2	23	unknown	unknown	Beak 1999
Reptiles							
No data were found							

Table B-1: Continued

Species of Concern	Toxic Response[2]	Exposure Concentration[1]	Exposure Time	Water Temperature (°C)	Chemical Composition of Test Water	Source of Test Organism	Reference
Birds							
Anas platyrhynchos (Peking duck)	increased body water (drinking freshwater = 63% of body mass; drinking NaCl water = 73% of body mass)	17,534 mg/L NaCl (0.3 M)	1 month	unknown	tap water	unknown	Hughes et al. 1991

[1] Concentrations given in molarity were converted to mg/L using the following molecular weights (Fessenden and Fessenden 1986):

Na^+ = 22.991 g/mol; Ca^{2+} = 40.08 g/mol; Mg^{2+} = 24.32 g/mol; K^+ = 39.1g/mol; Cl^- = 35.457 g/mol; C = 12.011 g/mol; N = 14.008 g/mol.

[2] Definitions of commonly used terms for toxic response (Hammer 1977; McKee and Wolf 1963):

- Lethal Concentration (LC_{50}) or Median Tolerance Limit (TLm) = concentration at which 50% of the test organisms die within a certain time period.
- NOEC = No Observed Effects Concentration
- LOEC = Lowest Observed Effects Concentration

[3] Tris buffer = Tris (hydroxymethyl) aminomethane-hydrochloric acid buffer, pH 7.2.

Table B-2: Calcium Chloride

Species of Concern	Toxic Response[2]	Exposure Concentration[1]	Exposure Time	Water Temperature (°C)	Chemical Composition of Test Water	Source of Test Organism	Reference
Bacteria, Protozoa, and Fungi							
aquatic hyphomycetes (fungi)	increased sporulation above controls	554 mg/L $CaCl_2$ (200 mg/L Ca^{+2})	48 hours	20	soft water	Boss Brook, Fenwick, Nova Scotia	Sridhar and Barlocher 1997
aquatic hyphomycetes (fungi)	decreased sporulation below controls	2,215 mg/L $CaCl_2$ (800 mg/L Ca^{+2})	48 hours	20	soft water	Boss Brook, Fenwick, Nova Scotia	Sridhar and Barlocher 1997
Phytoplankton							
Nitzschia linearis (diatom)	50% reduction in number of cells	3,130 mg/L $CaCl_2$	120 hours	unknown	soft dilution water	unknown	Patrick et al. 1968
Periphyton							
No data were found							
Macrophytes							
No data were found							
Zooplankton							
Cyclops vernalis (copepod)	immobilization (threshold)	1,730 mg/L $CaCl_2$	unknown	unknown	unknown	unknown	Anderson et al. 1948 (in Anderson 1948)
Daphnia longispina (water flea)	death	5,550 mg/L $CaCl_2$ (0.05 M)	41 hours	unknown	well water	unknown	Fowler 1931 (in Anderson 1948)
Daphnia pulex (water flea)	inhibited egg development and weakening	1,854 mg/L $CaCl_2$ (0.0167 M)	48 hours (2 days)	unknown	pond water	unknown	Ramult 1925 (in Anderson 1948)
Daphnia magna (water flea)	50% mortality	151 mg/L $CaCl_2$ (92 mg/L $Cl_.$)	unknown	unknown	unknown	unknown	Biesinger and Christensen 1972
Daphnia magna (water flea)	reproductive impairment	338 mg/L $CaCl_2$ (206 mg/L Cl^-)	21 days	unknown	unknown	unknown	Biesinger and Christensen 1972
Daphnia magna	survival	733 mg/L $CaCl_2$ (0.0066	24 hours	unknown	soft water	unknown	Naumann 1934

(water flea)	(no effect)	M)					(in Anderson 1948)

Table B-2: Continued

Species of Concern	Toxic Response[2]	Exposure Concentration[1]	Exposure Time	Water Temperature (°C)	Chemical Composition of Test Water	Source of Test Organism	Reference
Daphnia magna (water flea)	immobilization (50% of test animals)	920 mg/L $CaCl_2$	64 hours	25	Lake Erie water	unknown	Anderson 1948
Daphnia magna (water flea)	weakening	1,831 mg/L $CaCl_2$ (0.0165 M)	24 hours	unknown	soft water	unknown	Naumann 1934 (in Anderson 1948)
Daphnia magna (water flea)	survival (no effect)	1,831 mg/L $CaCl_2$ (0.0165 M)	24 hours	unknown	hard water	unknown	Naumann 1934 (in Anderson 1948)
Daphnia magna (water flea)	immobilization	3,662 mg/L $CaCl_2$ (0.033 M)	24 hours	unknown	hard water	unknown	Naumann 1934 (in Anderson 1948)
Mesacyclops leuckarti (copepod)	immobilization (threshold)	1,440 mg/L $CaCl_2$	unknown	unknown	unknown	unknown	Anderson et al. 1948 (in Anderson 1948)
Benthic Invertebrates							
Dreissena polymorpha (zebra mussel)	veligers = 100% mortality; settlers = 95% mortality	10,000 mg/L $CaCl_2$	3 hours	17	hardness = 140$\pm$10 mg/L as $CaCO_3$	Lake Michigan	Waller et al. 1996
Dreissena polymorpha (zebra mussel)	veligers = 100% mortality; settlers = 99.5% mortality	10,000 mg/L $CaCl_2$	6 hours	17	hardness = 140$\pm$10 mg/L as $CaCO_3$	Lake Michigan	Waller et al. 1996
Dreissena polymorpha (zebra mussel)	veligers = 100% mortality; settlers = 100% mortality	10,000 mg/L $CaCl_2$	12 hours	17	hardness = 140$\pm$10 mg/L as $CaCO_3$	Lake Michigan	Waller et al. 1996
Dreissena polymorpha (zebra mussel)	veligers = 100% mortality; settlers = 100% mortality	10,000 mg/L $CaCl_2$	24 hours	17	hardness = 140$\pm$10 mg/L as $CaCO_3$	Lake Michigan	Waller et al. 1996
Dreissena polymorpha (zebra mussel)	veligers = 98% mortality; settlers = 99% mortality	10,000 mg/L $CaCl_2$	12 hours	12	hardness = 140$\pm$10 mg/L as $CaCO_3$	Lake Michigan	Waller et al. 1996
Dreissena polymorpha (zebra mussel)	veligers = 100% mortality; settlers = 100% mortality	10,000 mg/L $CaCl_2$	24 hours	12	hardness = 140$\pm$10 mg/L as $CaCO_3$	Lake Michigan	Waller et al. 1996

Table B-2: Continued

Species of Concern	Toxic Response[2]	Exposure Concentration[1]	Exposure Time	Water Temperature (°C)	Chemical Composition of Test Water	Source of Test Organism	Reference
Musculium securis (clam)	reduced natality (mean # of newborns/# parents) with increasing $CaCl_2$ until 400mg/L $CaCl_2$ after which see increase but not to levels of control	0 mg/L $CaCl_2$ = mean natality per dish of 54.3; 400 mg/L $CaCl_2$ = mean natality of 22.0; 1000 mg/L $CaCl_2$ = mean natality of 30.3	1440-1920 hours (60-80 days)	unknown	distilled or deionized water with air-dried soil and willow or elm leaves	Carp Pond,Ontario	Mackie 1978
Polycelis nigra (flatworm)	survival for 48 hours	7200 mg/L $CaCl_2$ (2600 mg/L Ca^{2+})	48 hours	15-18	distilled water; planaria noted as surviving several weeks in distilled water	unknown	Jones 1940
Fish							
Exposure time = short (24 hours or less)							
Carassius auratus (goldfish)	killed or injured	7,752 mg/L $CaCl_2$	22-27 hours	unknown	distilled water	unknown	Ellis 1937 (in McKee and Wolf 1963)
Ictalurus punctatus (channel catfish)	37% mortality	10,000 mg/L $CaCl_2$	12 hours	12	hardness = 140$\pm$10 mg/L $CaCO_3$	Upper Mississippi Science Center in La Crosse, Wisconsin	Waller et al. 1996
Ictalurus punctatus (channel catfish)	63% mortality	10,000 mg/L $CaCl_2$	12 hours	17	hardness = 140$\pm$10 mg/L $CaCO_3$	Upper Mississippi Science Center in La Crosse, Wisconsin	Waller et al. 1996

Table B-2: Continued

Species of Concern	Toxic Response[2]	Exposure Concentration[1]	Exposure Time	Water Temperature (°C)	Chemical Composition of Test Water	Source of Test Organism	Reference
Ictalurus punctatus (channel catfish)	100% mortality	10,000 mg/L $CaCl_2$	24 hours	12	hardness = 140±10 mg/L $CaCO_3$	Upper Mississippi Science Center in La Crosse, Wisconsin	Waller et al. 1996
Ictalurus punctatus (channel catfish)	100% mortality	10,000 mg/L $CaCl_2$	24 hours	17	hardness = 140±10 mg/L $CaCO_3$	Upper Mississippi Science Center in La Crosse, Wisconsin	Waller et al. 1996
Ictalurus punctatus (channel catfish)	83% mortality	20,000 mg/L $CaCl_2$	3 hours	12	hardness = 140±10 mg/L $CaCO_3$	Upper Mississippi Science Center in La Crosse, Wisconsin	Waller et al. 1996
Lepomis macrochirus (bluegill sunfish)	50% mortality	8,363 mg/L $CaCl_2$ (5,344 mg/L Cl^-)	24 hours	unknown	unknown	unknown	Dowden and Bennett 1965
Lepomis macrochirus (bluegill sunfish)	50% mortality	8,400 mg/L $CaCl_2$	24 hours	unknown	synthetic river water	unknown	Abegg 1949, 1950 (in Doudoroff and Katz 1953)
Lepomis macrochirus (bluegill sunfish)	0% mortality	10,000 mg/L $CaCl_2$	12 hours	12 and 17 (two different experiments)	hardness = 140±10 mg/L $CaCO_3$	Upper Mississippi Science Center in La Crosse, Wisconsin	Waller et al. 1996
Lepomis macrochirus (bluegill sunfish)	0% mortality	10,000 mg/L $CaCl_2$	24 hours	17	hardness = 140±10 mg/L $CaCO_3$	Upper Mississippi Science Center in La Crosse, Wisconsin	Waller et al. 1996

Table B-2: Continued

Species of Concern	Toxic Response[2]	Exposure Concentration[1]	Exposure Time	Water Temperature (^oC)	Chemical Composition of Test Water	Source of Test Organism	Reference
Lepomis macrochirus (bluegill sunfish)	3.3% mortality	10,000 mg/L $CaCl_2$	24 hours	12	hardness = 140$\pm$10 mg/L $CaCO_3$	Upper Mississippi Science Center in La Crosse, Wisconsin	Waller et al. 1996
Lepomis macrochirus (bluegill sunfish)	average survival time	15,000 mg $CaCl_2$	17.7 hours	22-22.5	aerated, distilled water + tap water	unknown	Wiebe et al. 1934
Lepomis macrochirus (bluegill sunfish)	average survival time	20,000 mg $CaCl_2$	19.5 hours	22-22.5	aerated, distilled water + tap water	unknown	Wiebe et al. 1934
Lepomis macrochirus (bluegill sunfish)	97% mortality	20,000 mg/L $CaCl_2$	3 hours	12	hardness = 140$\pm$10 mg/L $CaCO_3$	Upper Mississippi Science Center in La Crosse, Wisconsin	Waller et al. 1996
Micropterus dolomieu (smallmouth bass)	0% mortality	10,000 mg/L $CaCl_2$	12 hours	12	hardness = 140$\pm$10 mg/L $CaCO_3$	Upper Mississippi Science Center in La Crosse, Wisconsin	Waller et al. 1996
Micropterus dolomieu (smallmouth bass)	0% mortality	10,000 mg/L $CaCl_2$	24 hours	12	hardness = 140$\pm$10 mg/L $CaCO_3$	Upper Mississippi Science Center in La Crosse, Wisconsin	Waller et al. 1996
Notemigonus crysoleucas (golden shiners)	average survival time	20,000 mg $CaCl_2$	6.4 hours	22-22.5	aerated, distilled water + tap water	unknown	Wiebe et al. 1934
Notemigonus crysoleucas (golden shiners)	average survival time	15,000 mg $CaCl_2$	17 hours	22-22.5	aerated, distilled water + tap water	unknown	Wiebe et al. 1934

Notemigonus crysoleucas (golden shiners)	average survival time	10,000 mg $CaCl_2$	27.6 hours	22-22.5	aerated, distilled water + tap water	unknown	Wiebe et al. 1934

Table B-2: Continued

Species of Concern	Toxic Response[2]	Exposure Concentration[1]	Exposure Time	Water Temperature (°C)	Chemical Composition of Test Water	Source of Test Organism	Reference
Oncorhynchus mykiss (rainbow trout)	10% mortality	10,000 mg/L $CaCl_2$	12 hours	12	hardness = 140$\pm$10 mg/L $CaCO_3$	Upper Mississippi Science Center in La Crosse, Wisconsin	Waller et al. 1996
Oncorhynchus mykiss (rainbow trout)	16% mortality	10,000 mg/L $CaCl_2$	24 hours	12	hardness = 140$\pm$10 mg/L $CaCO_3$	Upper Mississippi Science Center in La Crosse, Wisconsin	Waller et al. 1996
Oncorhynchus mykiss (rainbow trout)	40% mortality	10,000 mg/L $CaCl_2$	12 hours	17	hardness = 140$\pm$10 mg/L $CaCO_3$	Upper Mississippi Science Center in La Crosse, Wisconsin	Waller et al. 1996
Oncorhynchus mykiss (rainbow trout)	49% mortality	10,000 mg/L $CaCl_2$	24 hours	17	hardness = 140$\pm$10 mg/L $CaCO_3$	Upper Mississippi Science Center in La Crosse, Wisconsin	Waller et al. 1996
Oncorhynchus mykiss (rainbow trout)	20% mortality	20,000 mg/L $CaCl_2$	3 hours	12	hardness = 140$\pm$10 mg/L $CaCO_3$	Upper Mississippi Science Center in La Crosse, Wisconsin	Waller et al. 1996
Oncorhynchus mykiss (rainbow trout)	63% mortality	20,000 mg/L $CaCl_2$	3 hours	17	hardness = 140$\pm$10 mg/L $CaCO_3$	Upper Mississippi Science Center in La Crosse, Wisconsin	Waller et al. 1996

Orizias latipes (small freshwater cyprinodont)	death	13,874 mg/L $CaCl_2$ (0.125 M)	24 hours	unknown	unknown	unknown	Iwao 1936 (in Doudoroff and Katz 1953)

Table B-2: Continued

Species of Concern	Toxic Response[2]	Exposure Concentration[1]	Exposure Time	Water Temperature (°C)	Chemical Composition of Test Water	Source of Test Organism	Reference
Perca flavescens (yellow perch)	10% mortality	10,000 mg/L $CaCl_2$	12 hours	12	hardness = 140±10 mg/L $CaCO_3$	Upper Mississippi Science Center in La Crosse, Wisconsin	Waller et al. 1996
Perca flavescens (yellow perch)	80% mortality	10,000 mg/L $CaCl_2$	24 hours	12	hardness = 140±10 mg/L $CaCO_3$	Upper Mississippi Science Center in La Crosse, Wisconsin	Waller et al. 1996
Perca flavescens (yellow perch)	83% mortality	10,000 mg/L $CaCl_2$	12 hours	17	hardness = 140±10 mg/L $CaCO_3$	Upper Mississippi Science Center in La Crosse, Wisconsin	Waller et al. 1996
Perca flavescens (yellow perch)	83% mortality	10,000 mg/L $CaCl_2$	24 hours	17	hardness = 140±10 mg/L $CaCO_3$	Upper Mississippi Science Center in La Crosse, Wisconsin	Waller et al. 1996
Pimephales promelas (fathead minnow)	47% mortality	10,000 mg/L $CaCl_2$	12 hours	12	hardness = 140±10 mg/L $CaCO_3$	Upper Mississippi Science Center in La Crosse, Wisconsin	Waller et al. 1996
Pimephales promelas (fathead minnow)	100% mortality	10,000 mg/L $CaCl_2$	24 hours	12 and 17 (two different experiments)	hardness = 140±10 mg/L $CaCO_3$	Upper Mississippi Science Center in La Crosse, Wisconsin	Waller et al. 1996

Table B-2: Continued

Species of Concern	Toxic Response[2]	Exposure Concentration[1]	Exposure Time	Water Temperature (°C)	Chemical Composition of Test Water	Source of Test Organism	Reference
Pimephales promelas (fathead minnow)	100% mortality	20,000 mg/L $CaCl_2$	3 hours	12 and 17 (two different experiments)	hardness = 140$\pm$10 mg/L $CaCO_3$	Upper Mississippi Science Center in La Crosse, Wisconsin	Waller et al. 1996
Salmo trutta (brown trout)	20% mortality	10,000 mg/L $CaCl_2$	24 hours	12	hardness = 140$\pm$10 mg/L $CaCO_3$	Upper Mississippi Science Center in La Crosse, Wisconsin	Waller et al. 1996
Salvelinus namaycush (lake trout)	0% mortality	10,000 mg/L $CaCl_2$	24 hours	12	hardness = 140$\pm$10 mg/L $CaCO_3$	Upper Mississippi Science Center in La Crosse, Wisconsin	Waller et al. 1996
Stizostedion vitreum (walleye)	0% mortality	10,000 mg/L $CaCl_2$	12 hours	12 and 17 (two different experiments)	hardness = 140$\pm$10 mg/L $CaCO_3$	Upper Mississippi Science Center in La Crosse, Wisconsin	Waller et al. 1996
Stizostedion vitreum (walleye)	0% mortality	10,000 mg/L $CaCl_2$	24 hours	17	hardness = 140$\pm$10 mg/L $CaCO_3$	Upper Mississippi Science Center in La Crosse, Wisconsin	Waller et al. 1996
Stizostedion vitreum (walleye)	3.3% mortality	10,000 mg/L $CaCl_2$	24 hours	12	hardness = 140$\pm$10 mg/L $CaCO_3$	Upper Mississippi Science Center in La Crosse, Wisconsin	Waller et al. 1996

Table B-2: Continued

Species of Concern	Toxic Response[2]	Exposure Concentration[1]	Exposure Time	Water Temperature (°C)	Chemical Composition of Test Water	Source of Test Organism	Reference
Stizostedion vitreum (walleye)	17% mortality	20,000 mg/L $CaCl_2$	3 hours	12	hardness = 140±10 mg/L $CaCO_3$	Upper Mississippi Science Center in La Crosse, Wisconsin	Waller et al. 1996
Stizostedion vitreum (walleye)	100% mortality	20,000 mg/L $CaCl_2$	3 hours	17	hardness = 140±10 mg/L $CaCO_3$	Upper Mississippi Science Center in La Crosse, Wisconsin	Waller et al. 1996
Tinca vulgaris (tench)	death	10,000 mg/L $CaCl_2$	3 hours	20	fresh water	unknown	Wiebe et al. 1934
<u>Exposure time = greater than 24 hours</u>							
Ambloplites rupestris (rock bass)	killed or injured	555 mg/L $CaCl_2$	168 hours (1 week)	unknown	tap water	unknown	Ellis 1937 (in McKee and Wolf 1963)
Anguilla japonica (young eel)	survival	11,099 mg/L $CaCl_2$ (0.1 M)	50 hours	20-22	unknown	unknown	Oshima 1931 (in Doudoroff and Katz 1953)
Fish	death	gradual increase from 9,500 to 13,500 mg/L $CaCl_2$	within 288 hours (within 12 days)	unknown	unknown	unknown	Young 1923 (in Hanes et al. 1970)
Gambusia affinis (mosquito-fish)	50% mortality	13,400 mg/L $CaCl_2$	96 hours	unknown	turbid	unknown	Wallen et al. 1957 (in McKee and Wolf 1963)
Lepomis macrochirus (bluegill sunfish)	50% mortality	9,500 mg/L $CaCl_2$	96 hours	unknown	standard	unknown	Cairns & Scheier 1958 (in McKee and Wolf 1963)
Lepomis macrochirus (bluegill sunfish)	average survival time	10,000 mg $CaCl_2$	48.8 hours	22-22.5	aerated, distilled water + tap water	unknown	Wiebe et al. 1934

Table B-2: Continued

Species of Concern	Toxic Response[2]	Exposure Concentration[1]	Exposure Time	Water Temperature (°C)	Chemical Composition of Test Water	Source of Test Organism	Reference
Lepomis macrochirus (bluegill sunfish)	50% mortality	10,650 mg/L $CaCl_2$	96 hours	18$\pm$2	soft dilution water (see Patrick et al. 1968 for details)	unknown	Patrick et al. 1968
Notemigonus crysoleucas (golden shiners)	average survival time	5,000 mg $CaCl_2$	143.5 hours	22-22.5	aerated, distilled water + tap water	unknown	Wiebe et al. 1934
Notemigonus crysoleucas (golden shiners)	death	5,000 mg/L $CaCl_2$	143 hours	unknown	unknown	unknown	Ellis 1937 (in Anderson 1948)
Notropis blennius (river shiner)	death	277 mg/L $CaCl_2$ (0.0025 M)	840-1176 hours (5-7 weeks)	room temperature	distilled water	unknown	Garrey 1916 (in Doudoroff and Katz 1953)
Notropis blennius (river shiner)	death	832 mg/L $CaCl_2$ (0.0075 M)	336-504 hours (14-21 days)	room temperature	distilled water	unknown	Garrey 1916 (in Doudoroff and Katz 1953)
Notropis blennius (river shiner)	death	2,775 mg/L $CaCl_2$ (0.025 M)	48-96 hours (2-4 days)	room temperature	distilled water	unknown	Garrey 1916 (in Doudoroff and Katz 1953)
Exposure time = unknown							
Coregonus clupeaformis (lake whitefish) fry	immobilization	22,080 mg/L $CaCl_2$	unknown	unknown	Lake Erie water	unknown	Edmister & Gray 1948 (in Anderson 1948; Doudoroff and Katz 1953)
Stizostedion vitreum (walleye) fry	immobilization	12,060 mg/L $CaCl_2$	unknown	unknown	Lake Erie water	unknown	Edmister & Gray 1948 (in Anderson 1948; Doudoroff and Katz 1953)
Stizostedion vitreum (walleye)	threshold toxicity	12,060 mg/L $CaCl_2$	unknown	unknown	unknown	unknown	ORVWSC 1950 (in Hanes et al. 1970)
Amphibians							
No data were found							

Table B-2: Continued

Species of Concern	Toxic Response[2]	Exposure Concentration[1]	Exposure Time	Water Temperature (°C)	Chemical Composition of Test Water	Source of Test Organism	Reference
Reptiles							
No data were found							
Birds							
No data were found							

[1] Concentrations given in molarity were converted to mg/L using the following molecular weights (Fessenden and Fessenden 1986):

Na^+ = 22.991 g/mol; Ca^{2+} = 40.08 g/mol; Mg^{2+} = 24.32 g/mol; K^+ = 39.1g/mol; Cl^- = 35.457 g/mol; C = 12.011 g/mol; N = 14.008 g/mol.

[2] Definitions of commonly used terms for toxic response (Hammer 1977; McKee and Wolf 1963):

- Lethal Concentration (LC_{50}) or Median Tolerance Limit (TLm) = concentration at which 50% of the test organisms die within a certain time period.
- NOEC = No Observed Effects Concentration
- LOEC = Lowest Observed Effects Concentration

Table B-3: Magnesium Chloride

Species of Concern	Toxic Response[2]	Exposure Concentration[1]	Exposure Time	Water Temperature (°C)	Chemical Composition of Test Water	Source of Test Organism	Reference
Bacteria, Protozoa, and Fungi							
Aphanomyces astaci (fungus)	no spore production	1,904 mg/L $MgCl_2$ (20 mM $MgCl_2$)	120 hours (5 days)	13	water from Lake Halsjon, Sweden	unknown	Rantamaki et al. 1992
Boekelovia hooglandii (euryhaline, golden-brown microflagellate alga)	no growth	13,333 mg/L $MgCl_2$ (0.14 M)	12 days (incubation time)	25	culture medium	W.R. Barclay, University of Denver	Fuji and Hellebust 1994
Phytoplankton							
No data were found							
Periphyton							
No data were found							
Macrophytes							
No data were found							
Zooplankton							
Ceriodaphnia reticulata (water flea)	reproduction (continued production of living young)	60 mg/L $MgCl_2$	15 days	unknown	pond water (total solids=91.7mg/L; Ca^{2+}=16.5mg/L; Mg^{2+}=1.7mg/L; Cl^-=2.6 mg/L)	unknown	Hutchinson 1932
Daphnia longispina (clone 2) (water flea)	reproduction (continued production of living young)	30 mg/L $MgCl_2$	unknown	unknown	pond water (total solids=91.7mg/L; Ca^{2+}=16.5mg/L; Mg^{2+}=1.7mg/L; Cl^-=2.6 mg/L)	unknown	Hutchinson 1932
Daphnia longispina (clone 1) (water flea)	reproduction (continued production of living young)	60 mg/L $MgCl_2$	unknown	unknown	pond water (total solids=91.7mg/L; Ca^{2+}=16.5mg/L; Mg^{2+}=1.7mg/L; Cl^-=2.6	unknown	Hutchinson 1932

					mg/L)		

Table B-3: Continued

Species of Concern	Toxic Response[2]	Exposure Concentration[1]	Exposure Time	Water Temperature (°C)	Chemical Composition of Test Water	Source of Test Organism	Reference
Daphnia longispina (water flea)	death	3,524 mg/L $MgCl_2$ (0.037 M)	43 hours	unknown	well water	unknown	Fowler 1931 (in Anderson 1948)
Daphnia magna (water flea)	reproductive impairment	417 mg/L $MgCl_2$ (239 mg/L Cl^-)	21 days	unknown	unknown	unknown	Biesinger and Christensen 1972
Daphnia magna (water flea)	reproduction (continued production of living young)	240 mg/L $MgCl_2$	19 days	unknown	pond water (total solids=91.7mg/L; Ca^{2+}=16.5mg/L; Mg^{2+}=1.7mg/L; Cl^-=2.6 mg/L)	unknown	Hutchinson 1932
Daphnia magna (water flea)	50% mortality	714 mg/L $MgCl_2$ (409 mg/L Cl^-)	unknown	unknown	unknown	unknown	Biesinger and Christensen 1972
Daphnia magna (water flea)	immobilization (50% of test animals)	740 mg/L $MgCl_2$	64 hours	25	Lake Erie water	unknown	Anderson 1948
Daphnia magna (water flea)	harmful	1571 mg/L $MgCl_2$ (0.0165 M)	unknown	unknown	soft and hard water	unknown	Naumann 1934 (in Anderson 1948)
Daphnia pulex (water flea)	reproduction (continued production of living young)	120 mg/L $MgCl_2$	unknown	unknown	pond water (total solids=91.7mg/L; Ca^{2+}=16.5mg/L; Mg^{2+}=1.7mg/L; Cl^-=2.6 mg/L)	unknown	Hutchinson 1932
Daphnia thomsoni (water flea)	reproduction (continued production of living young)	30 mg/L $MgCl_2$	unknown	unknown	pond water (total solids=91.7mg/L; Ca^{2+}=16.5mg/L; Mg^{2+}=1.7mg/L; Cl^-=2.6 mg/L)	unknown	Hutchinson 1932
Moina macrocopa (water flea)	reproduction (continued production of living young)	180 mg/L $MgCl_2$	25 days	unknown	pond water (total solids=91.7mg/L; Ca^{2+}=16.5mg/L; Mg^{2+}=1.7mg/L; Cl^-=2.6 mg/L)	unknown	Hutchinson 1932

Table B-3: Continued

Species of Concern	Toxic Response[2]	Exposure Concentration[1]	Exposure Time	Water Temperature (°C)	Chemical Composition of Test Water	Source of Test Organism	Reference
Benthic Invertebrates							
Polycelis nigra (flatworm)	survival for 48 hours	3,798 mg/L $MgCl_2$ (970 mg/L Mg^{2+})	48 hours	15 - 18	distilled water; planaria noted as surviving several weeks in distilled water	unknown	Jones 1940
Fish							
Exposure time = short (24 hours or less)							
Notemigonus crysoleucas (golden shiners)	average survival time	10,000 mg $MgCl_2$	4.6 hours	22-22.5	aerated, distilled water + tap water	unknown	Wiebe et al. 1934
Notemigonus crysoleucas (golden shiners)	average survival time	15,000 mg $MgCl_2$	0.8 hours	22-22.5	aerated, distilled water + tap water	unknown	Wiebe et al. 1934
Notemigonus crysoleucas (golden shiners)	average survival time	20,000 mg $MgCl_2$	0.5 hours	22-22.5	aerated, distilled water + tap water	unknown	Wiebe et al. 1934
Orizias latipes (small freshwater cyprinodont)	death	23,809 mg/L $MgCl_2$ (0.25 M)	24 hours	unknown	unknown	unknown	Iwao 1936 (in Doudoroff and Katz 1953)
Exposure time = greater than 24 hours)							
Anguilla japonica (young eel)	survival	9,523 mg/L $MgCl_2$ (0.1 M)	50 hours	20 - 22	unknown	unknown	Oshima 1931 (in Doudoroff and Katz 1953)
Carassius auratus (goldfish)	death	6,757 mg/L $MgCl_2$	72-504 hours (3-21 days)	unknown	distilled	unknown	Ellis 1937 (in McKee and Wolf 1963)
Gambusia affinis (mosquito-fish)	50% mortality	13,400 mg/L $MgCl_2$	96 hours	unknown	turbid	unknown	Wallen et al. 1957 (in McKee and Wolf 1963)
Notemigonus crysoleucas	average survival time	5,000 mg $MgCl_2$	96.5 hours	22-22.5	aerated, distilled water + tap water	unknown	Wiebe et al. 1934

(golden shiners)							

Table B-3: Continued

Species of Concern	Toxic Response[2]	Exposure Concentration[1]	Exposure Time	Water Temperature (°C)	Chemical Composition of Test Water	Source of Test Organism	Reference
minnows	death	476 mg/L $MgCl_2$	96-144 hours (4-6 days)	unknown	distilled	unknown	Ellis 1937 (in McKee and Wolf 1963)
Notropis blennius (river shiner)	death	476 mg/L $MgCl_2$ (0.005 M)	96-144 hours (4-6 days)	room temperature	distilled water	unknown	Garrey 1916 (in Doudoroff and Katz 1953)
Notropis blennius (river shiner)	death	2,381 mg/L $MgCl_2$ (0.025 M)	~48 hours (2 days)	room temperature	distilled water	unknown	Garrey 1916 (in Doudoroff and Katz 1953)
Exposure time = unknown							
Cyprinus sp. (carp)	death	8132 mg/L $MgCl_2$	unknown	unknown	unknown	unknown	Pfiefer 1953 (in McKee and Wolf 1963)
Amphibians							
No data were found							
Reptiles							
No data were found							
Birds							
No data were found							

[1] Concentrations given in molarity were converted to mg/L using the following molecular weights (Fessenden and Fessenden 1986):

Na^+ = 22.991 g/mol; Ca^{2+} = 40.08 g/mol; Mg^{2+} = 24.32 g/mol; K^+ = 39.1g/mol; Cl^- = 35.457 g/mol; C = 12.011 g/mol; N = 14.008 g/mol.

[2] Definitions of commonly used terms for toxic response (Hammer 1977; McKee and Wolf 1963):

- Lethal Concentration (LC_{50}) or Median Tolerance Limit (TLm) = concentration at which 50% of the test organisms die within a certain time period.
- NOEC = No Observed Effects Concentration
- LOEC = Lowest Observed Effects Concentration

Table B-4: Potassium Chloride

Species of Concern	Toxic Response[2]	Exposure Concentration[1]	Exposure Time	Water Temperature (°C)	Chemical Composition of Test Water	Source of Test Organism	Reference
Bacteria, Protozoa, and Fungi							
Boekelovia hooglandii (euryhaline, golden-brown microflagellate alga)	no growth	14,911 mg/L KCl (0.2 M)	12 days (incubation time)	25	culture medium	W.R. Barclay, University of Denver	Fuji and Hellebust 1994
Phytoplankton							
Nitzschia linearis (diatom)	50% reduction in number of cells	1,337 mg/L KCl	120 hours	unknown	soft dilution water	unknown	Patrick et al. 1968
Periphyton							
No data were found							
Macrophytes							
No data were found							
Zooplankton							
Cyclops vernalis (copepod)	immobilization (threshold)	640 mg/L KCl	unknown	20-25	Lake Erie water	unknown	Anderson et al. 1948 (in McKee and Wolf 1963)
Daphnia magna (water flea)	reproductive impairment	65 mg/L KCl (44 mg/L Cl^-)	21 days	unknown	unknown	unknown	Biesinger and Christensen 1972
Daphnia magna (water flea)	50% mortality	127 mg/L KCl (86 mg/L Cl^-)	unknown	unknown	unknown	unknown	Biesinger and Christensen 1972
Daphnia magna (young) (water flea)	immobilization (threshold)	430 mg/L KCl	unknown	20-25	Lake Erie water	unknown	Anderson et al. 1948 (in McKee and Wolf 1963)
Daphnia magna (water flea)	immobilization (50% of test animals)	432 mg/L KCl	64 hours	25	Lake Erie water	unknown	Anderson 1948

Daphnia magna (water flea)	immobilization	746 mg/L KCl (0.01 M)	unknown	unknown	soft water	unknown	Naumann 1934 (in Anderson 1948)

Table B-4: Continued

Species of Concern	Toxic Response[2]	Exposure Concentration[1]	Exposure Time	Water Temperature (°C)	Chemical Composition of Test Water	Source of Test Organism	Reference
Daphnia magna (water flea)	irritation	746 mg/L KCl (0.01 M)	unknown	unknown	hard water	unknown	Naumann 1934 (in Anderson 1948)
Daphnia longispina (water flea)	death	1,864 mg/L KCl (0.025 M)	<7 hours	unknown	well water	unknown	Fowler 1931 (in Anderson 1948)
Daphnia pulex (water flea)	survival (no effect)	619 mg/L KCl (0.0083 M)	unknown	unknown	pond water	unknown	Ramult 1925 (in Anderson 1948)
Daphnia pulex (water flea)	no egg development	1,245 mg/L KCl (0.0167 M)	unknown	unknown	pond water	unknown	Ramult 1942 (in Anderson 1948)
Daphnia pulex (water flea)	death	1,245 mg/L KCl (0.0167 M)	unknown	unknown	pond water	unknown	Ramult 1925 (in Anderson 1948)
Diaptomus oregonensis (copepod)	immobilization (threshold)	134 mg/L KCl	unknown	20-25	Lake Erie water	unknown	Anderson et al. 1948 (in McKee and Wolf 1963)
Leptodora kindtii (giant water flea)	immobilization (threshold)	127 mg/L KCl	unknown	20-25	Lake Erie water	unknown	Anderson et al. 1948 (in McKee and Wolf 1963)
Mesocyclops leuckarti (copepod)	immobilization (threshold)	566 mg/L KCl	unknown	20-25	Lake Erie water	unknown	Anderson et al. 1948 (in McKee and Wolf 1963)
Benthic Invertebrates							
Caenorhabditis elegans (nematode)	mortality not significantly different from controls	11,510 mg/L KCl (total - includes KCl in test water)	24 hours	20	K-medium (2.36 g KCl + 3.0 g NaCl/L distilled water)	unknown	Khanna et al. 1997
Caenorhabditis elegans (nematode)	mortality not significantly different from controls	11,510 mg/L NaCl (total - includes KCl in test water)	96 hours	20	K-medium (2.36 g KCl + 3.0 g NaCl/L distilled water)	unknown	Khanna et al. 1997

Table B-4: Continued

Species of Concern	Toxic Response[2]	Exposure Concentration[1]	Exposure Time	Water Temperature (°C)	Chemical Composition of Test Water	Source of Test Organism	Reference
Caenorhabditis elegans (nematode)	mortality not significantly different from controls	18,850 mg/L KCl (total - includes KCl in test water)	24 hours	20	moderately hard reconstituted water (96 mg $NaHCO_3$ + 60 mg $CaSO_4 \cdot 2H_2O$ + 60 mg $MgSO_4$ + 4 mg KCl per L distilled water)	unknown	Khanna et al. 1997
Caenorhabditis elegans (nematode)	mortality not significantly different from controls	18,900 mg/L NaCl (total - includes KCl in test water)	96 hours	20	moderately hard reconstituted water (96 mg $NaHCO_3$ + 60 mg $CaSO_4 \cdot 2H_2O$ + 60 mg $MgSO_4$ + 4 mg KCl per liter distilled water)	unknown	Khanna et al. 1997
Cricotopus trifascia (chironomid midge)	100% mortality	4,896 mg/L KCl (calculated from regression)	48 hours	12	filtered lake water	Lake Michigan	Hamilton et al. 1975
Dreissena polymorpha (zebra mussel)	veligers = 100% mortality; settlers = 96% mortality	2,500 mg/L KCl	24 hours	17	hardness = 140$\pm$10 mg/L as $CaCO_3$	Lake Michigan	Waller et al. 1996
Dreissena polymorpha (zebra mussel)	veligers = 100% mortality; settlers = 100% mortality	2,500 mg/L KCl	24 hours	12	hardness = 140$\pm$10 mg/L as $CaCO_3$	Lake Michigan	Waller et al. 1996
Dreissena polymorpha (zebra mussel)	veligers = 100% mortality; settlers = 89% mortality	10,000 mg/L KCl	3 hours	12	hardness = 140$\pm$10 mg/L as $CaCO_3$	Lake Michigan	Waller et al. 1996
Hydroptila angusta (caddisfly)	100% mortality	6,317 mg/L KCl (calculated from regression)	48 hours	12	filtered lake water	Lake Michigan	Hamilton et al. 1975
Laccophilus maculosis (water beetle)	increased movement in 50% of test organisms	5,800 mg/L KCl	unknown	unknown	unknown	unknown	Hodgson 1951 (in McKee and Wolf 1963)
Nais variabilis (oligochaete)	100% mortality	204 mg/L KCl (calculated from regression)	48 hours	12	filtered lake water	Lake Michigan	Hamilton et al. 1975

Table B-4: Continued

Species of Concern	Toxic Response[2]	Exposure Concentration[1]	Exposure Time	Water Temperature (°C)	Chemical Composition of Test Water	Source of Test Organism	Reference
Physa heterostropha (fresh-water snail)	50% mortality	940 mg/L KCl (LC_{50})	96 hours	20±2	soft dilution water (see Patrick et al. 1968 for details)	unknown	Patrick et al. 1968
Polycelis nigra (flatworm)	survival for 48 hours	1259 mg/L KCl (660 mg/L K^+)	48 hours	15 - 18	distilled water; planaria noted as surviving several weeks in distilled water	unknown	Jones 1940
Fish							
Exposure time = 24 hours or less							
Carassius auratus (goldfish)	death	74.6 mg/L KCl	4.5-15 hours	unknown	distilled	unknown	Ellis 1937 (in McKee and Wolf 1963)
Carassius auratus (goldfish)	death	7,700 mg/L KCl	4.6-15 hours	~21	unknown	unknown	Powers 1917 (in Hammer 1977; Doudoroff and Katz 1953)
Carassius auratus (goldfish)	death	32,800 mg/L KCl	0.23-0.28 hours	~21	unknown	unknown	Powers 1917 (in Hammer 1977; Doudoroff and Katz 1953)
Ictalurus punctatus (channel catfish)	0% mortality	2,500 mg/L KCl	24 hours	17	hardness = 140±10 mg/L $CaCO_3$	Upper Mississippi Science Center in La Crosse, Wisconsin	Waller et al. 1996
Ictalurus punctatus (channel catfish)	0% mortality	10,000 mg/L KCl	3 hours	12 and 17 (two different experiments)	hardness = 140±10 mg/L $CaCO_3$	Upper Mississippi Science Center in La Crosse, Wisconsin	Waller et al. 1996

Table B-4: Continued

Species of Concern	Toxic Response[2]	Exposure Concentration[1]	Exposure Time	Water Temperature (°C)	Chemical Composition of Test Water	Source of Test Organism	Reference
Ictalurus punctatus (channel catfish)	0% mortality	10,000 mg/L KCl	6 hours	12 and 17 (two different experiments)	hardness = 140$\pm$10 mg/L $CaCO_3$	Upper Mississippi Science Center in La Crosse, Wisconsin	Waller et al. 1996
Ictalurus punctatus (channel catfish)	3.9% mortality	2,500 mg/L KCl	24 hours	12	hardness = 140$\pm$10 mg/L $CaCO_3$	Upper Mississippi Science Center in La Crosse, Wisconsin	Waller et al. 1996
Lepomis macrochirus (bluegill sunfish)	0% mortality	2,500 mg/L KCl	24 hours	12	hardness = 140$\pm$10 mg/L $CaCO_3$	Upper Mississippi Science Center in La Crosse, Wisconsin	Waller et al. 1996
Lepomis macrochirus (bluegill sunfish)	20.0% mortality	2,500 mg/L KCl	24 hours	17	hardness = 140$\pm$10 mg/L $CaCO_3$	Upper Mississippi Science Center in La Crosse, Wisconsin	Waller et al. 1996
Lepomis macrochirus (bluegill sunfish)	50% mortality	3,896 mg/L KCl (2,640 mg/L Cl^-)	24 hours	unknown	unknown	unknown	Dowden and Bennett 1965
Lepomis macrochirus (bluegill sunfish)	50% mortality	5,500 mg/L KCl	24 hours	unknown	synthetic river water	unknown	Abegg 1949, 1950 (in Doudoroff and Katz 1953)
Lepomis macrochirus (bluegill sunfish)	0% mortality	10,000 mg/L KCl	3 hours	12 and 17 (two different experiments)	hardness = 140$\pm$10 mg/L $CaCO_3$	Upper Mississippi Science Center in La Crosse, Wisconsin	Waller et al. 1996

Table B-4: Continued

Species of Concern	Toxic Response[2]	Exposure Concentration[1]	Exposure Time	Water Temperature (°C)	Chemical Composition of Test Water	Source of Test Organism	Reference
Lepomis macrochirus (bluegill sunfish)	0% mortality	10,000 mg/L KCl	6 hours	12 and 17 (two different experiments)	hardness = 140±10 mg/L $CaCO_3$	Upper Mississippi Science Center in La Crosse, Wisconsin	Waller et al. 1996
Micropterus dolomieu (smallmouth bass)	3.9% mortality	2,500 mg/L KCl	24 hours	12	hardness = 140±10 mg/L $CaCO_3$	Upper Mississippi Science Center in La Crosse, Wisconsin	Waller et al. 1996
Micropterus dolomieu (smallmouth bass)	50% mortality	10,000 mg/L KCl	6 hours	12	hardness = 140±10 mg/L $CaCO_3$	Upper Mississippi Science Center in La Crosse, Wisconsin	Waller et al. 1996
minnows	death	373 mg/L KCl	12-29 hours	unknown	distilled	unknown	Ellis 1937 (in McKee and Wolf 1963)
Notropis blennius (river shiner)	death	400 mg/L KCl	12-29 hours	room temperature	distilled water	unknown	Garrey 1916 (in Hammer 1977; Doudoroff and Katz 1953)
Oncorhynchus mykiss (rainbow trout)	0% mortality	2,500 mg/L KCl	24 hours	12 and 17 (two different experiments)	hardness = 140±10 mg/L $CaCO_3$	Upper Mississippi Science Center in La Crosse, Wisconsin	Waller et al. 1996
Oncorhynchus mykiss (rainbow trout)	22.1% mortality	10,000 mg/L KCl	6 hours	12	hardness = 140±10 mg/L $CaCO_3$	Upper Mississippi Science Center in La Crosse, Wisconsin	Waller et al. 1996

Table B-4: Continued

Species of Concern	Toxic Response[2]	Exposure Concentration[1]	Exposure Time	Water Temperature (°C)	Chemical Composition of Test Water	Source of Test Organism	Reference
Oncorhynchus mykiss (rainbow trout)	30% mortality	10,000 mg/L KCl	3 hours	17	hardness = 140±10 mg/L $CaCO_3$	Upper Mississippi Science Center in La Crosse, Wisconsin	Waller et al. 1996
Oncorhynchus mykiss (rainbow trout)	93.3% mortality	10,000 mg/L KCl	6 hours	17	hardness = 140±10 mg/L $CaCO_3$	Upper Mississippi Science Center in La Crosse, Wisconsin	Waller et al. 1996
Orizias latipes (small freshwater cyprinodont)	death	1937 mg/L KCl (0.0312 M)	24 hours	unknown	unknown	unknown	Iwao 1936 (in Doudoroff and Katz 1953)
Perca flavescens (yellow perch)	0% mortality	10,000 mg/L KCl	6 hours	12 and 17 (two different experiments)	hardness = 140±10 mg/L $CaCO_3$	Upper Mississippi Science Center in La Crosse, Wisconsin	Waller et al. 1996
Perca flavescens (yellow perch)	46.7% mortality	2,500 mg/L KCl	24 hours	12	hardness = 140±10 mg/L $CaCO_3$	Upper Mississippi Science Center in La Crosse, Wisconsin	Waller et al. 1996
Perca flavescens (yellow perch)	80% mortality	2,500 mg/L KCl	24 hours	17	hardness = 140±10 mg/L $CaCO_3$	Upper Mississippi Science Center in La Crosse, Wisconsin	Waller et al. 1996

Table B-4: Continued

Species of Concern	Toxic Response[2]	Exposure Concentration[1]	Exposure Time	Water Temperature (°C)	Chemical Composition of Test Water	Source of Test Organism	Reference
Pimephales promelas (fathead minnow)	0% mortality	2,500 mg/L KCl	24 hours	12 and 17 (two different experiments)	hardness = 140$\pm$10 mg/L $CaCO_3$	Upper Mississippi Science Center in La Crosse, Wisconsin	Waller et al. 1996
Pimephales promelas (fathead minnow)	0% mortality	10,000 mg/L KCl	3 hours	12 and 17 (two different experiments)	hardness = 140$\pm$10 mg/L $CaCO_3$	Upper Mississippi Science Center in La Crosse, Wisconsin	Waller et al. 1996
Pimephales promelas (fathead minnow)	0% mortality	10,000 mg/L KCl	6 hours	12	hardness = 140$\pm$10 mg/L $CaCO_3$	Upper Mississippi Science Center in La Crosse, Wisconsin	Waller et al. 1996
Salmo trutta (brown trout)	0% mortality	2,500 mg/L KCl	24 hours	12	hardness = 140$\pm$10 mg/L $CaCO_3$	Upper Mississippi Science Center in La Crosse, Wisconsin	Waller et al. 1996
Salmo trutta (brown trout)	0% mortality	10,000 mg/L KCl	6 hours	12	hardness = 140$\pm$10 mg/L $CaCO_3$	Upper Mississippi Science Center in La Crosse, Wisconsin	Waller et al. 1996
Salvelinus namaycush (lake trout)	0% mortality	2,500 mg/L KCl	24 hours	12	hardness = 140$\pm$10 mg/L $CaCO_3$	Upper Mississippi Science Center in La Crosse, Wisconsin	Waller et al. 1996

Table B-4: Continued

Species of Concern	Toxic Response[2]	Exposure Concentration[1]	Exposure Time	Water Temperature (°C)	Chemical Composition of Test Water	Source of Test Organism	Reference
Salvelinus namaycush (lake trout)	0% mortality	10,000 mg/L KCl	3 hours	12	hardness = 140±10 mg/L $CaCO_3$	Upper Mississippi Science Center in La Crosse, Wisconsin	Waller et al. 1996
Salvelinus namaycush (lake trout)	0% mortality	10,000 mg/L KCl	6 hours	12	hardness = 140±10 mg/L $CaCO_3$	Upper Mississippi Science Center in La Crosse, Wisconsin	Waller et al. 1996
Stizostedion vitreum (walleye)	100% mortality	2,500 mg/L KCl	24 hours	12 and 17 (two different experiments)	hardness = 140±10 mg/L $CaCO_3$	Upper Mississippi Science Center in La Crosse, Wisconsin	Waller et al. 1996
Stizostedion vitreum (walleye)	0% mortality	10,000 mg/L KCl	3 hours	12	hardness = 140±10 mg/L $CaCO_3$	Upper Mississippi Science Center in La Crosse, Wisconsin	Waller et al. 1996
Stizostedion vitreum (walleye)	6.7% mortality	10,000 mg/L KCl	3 hours	17	hardness = 140±10 mg/L $CaCO_3$	Upper Mississippi Science Center in La Crosse, Wisconsin	Waller et al. 1996
Stizostedion vitreum (walleye)	63.3% mortality	10,000 mg/L KCl	6 hours	12	hardness = 140±10 mg/L $CaCO_3$	Upper Mississippi Science Center in La Crosse, Wisconsin	Waller et al. 1996

Table B-4: Continued

Species of Concern	Toxic Response[2]	Exposure Concentration[1]	Exposure Time	Water Temperature (°C)	Chemical Composition of Test Water	Source of Test Organism	Reference
Stizostedion vitreum (walleye)	93.3% mortality	10,000 mg/L KCl	6 hours	17	hardness = 140±10 mg/L $CaCO_3$	Upper Mississippi Science Center in La Crosse, Wisconsin	Waller et al. 1996
Stizostedion vitreum (walleye)	50% mortality	12,060 mg/L KCl	24 hours	unknown	unknown	unknown	ORVWSC 1950 (in McKee and Wolf 1963)
Exposure time = greater than 24 hours							
Anguilla japonica (young eel)	survival	6,209 mg/L KCl (0.1 M)	50 hours	20 - 22	unknown	unknown	Oshima 1931 (in Doudoroff and Katz 1953)
Coregonus clupeaformis (lake whitefish) fry	immobilization	10,368 mg/L KCl	unknown	unknown	Lake Erie water	unknown	Edmister & Gray 1948 (in Doudoroff and Katz 1953)
Gambusia affinis (mosquito-fish)		920 mg/L KCl	96 hours	unknown	turbid	unknown	Wallen et al. 1957 (in Hammer 1977; McKee and Wolf 1963)
Lepomis macrochirus (bluegill sunfish)	50% mortality	2,000 mg/L KCl	96 hours	unknown	unknown	unknown	Trama 1954 (in Hammer 1977)
Lepomis macrochirus (bluegill sunfish)	50% mortality	2,010 mg/L KCl	96 hours	unknown	unknown	unknown	Anon. 1960 (in McKee and Wolf 1963)
Lepomis macrochirus (bluegill sunfish)	50% mortality	2,010 mg/L KCl	96 hours	18±2	soft dilution water (see Patrick et al. 1968 for details)	unknown	Patrick et al. 1968
Perca flavescens (yellow perch)	death	1,360 mg/L KCl	72 hours (3 days)	unknown	well water	unknown	Young 1923 (in McKee and Wolf 1963)

Pimephales promelas (fathead minnow) 1 to 7 days old)	NOEC	500 mg/L KCl	168 hours (7 days)	25$\pm$1	moderately hard reconstituted water	EPA Newtown Facility	Pickering et al. 1996

Table B-4: Continued

Species of Concern	Toxic Response[2]	Exposure Concentration[1]	Exposure Time	Water Temperature (°C)	Chemical Composition of Test Water	Source of Test Organism	Reference
Pimephales promelas (fathead minnow) larvae <24 hours	50% mortality	861 mg/L KCl	168 hours (7 days)	unknown	unknown	unknown	Beak unpublished
Pimephales promelas (fathead minnow) 1 to 7 days old)	LOEC	1,000 mg/L KCl	168 hours (7 days)	25±1	moderately hard reconstituted water	EPA Newtown Facility	Pickering et al. 1996
Exposure time = unknown							
Stizostedion vitreum (walleye) fry	immobilization	751 mg/L KCl	unknown	unknown	Lake Erie water	unknown	Edmister & Gray 1948 (in Doudoroff and Katz 1953)
Stizostedion vitreum (walleye) fry	death	751 mg/L KCl	unknown	unknown	Lake Erie water	unknown	ORVWSC 1950; Anderson et al. 1948 (in McKee and Wolf 1963)
Amphibians							
Microhyla ornata (frog embryos and larvae)	swollen head that increased buoyancy	2,000 mg/L KCl (0.2%)	96 hours	23 - 27	total hardness <75 mg/L as $CaCO^3$	natural ponds/temporary rainwater pools in India	Padhye and Ghate 1992
Rana pipiens (leopard frog)	reduced maximum rate of reaction of pyruvate kinase and phospho-fructokinase	14,911 mg/L KCl (200 mM)	unknown	22	unknown	commercial supplier	Grundy and Storey 1994

Table B-4: Continued

Species of Concern	Toxic Response[2]	Exposure Concentration[1]	Exposure Time	Water Temperature (°C)	Chemical Composition of Test Water	Source of Test Organism	Reference
Scaphiopus couchii (spadefoot toad)	reduced maximum rate of reaction of pyruvate kinase and phospho-fructokinase	14,911 mg/L KCl (200 mM)	unknown	22	unknown	Tucson, Arizona	Grundy and Storey 1994
Reptiles							
No data were found							
Birds							
No data were found							

[1] Concentrations given in molarity were converted to mg/L using the following molecular weights (Fessenden and Fessenden 1986):

Na^+ = 22.991 g/mol; Ca^{2+} = 40.08 g/mol; Mg^{2+} = 24.32 g/mol; K^+ = 39.1g/mol; Cl^- = 35.457 g/mol; C = 12.011 g/mol; N = 14.008 g/mol.

[2] Definitions of commonly used terms for toxic response (Hammer 1977; McKee and Wolf 1963):

- Lethal Concentration (LC_{50}) or Median Tolerance Limit (TLm) = concentration at which 50% of the test organisms die within a certain time period.
- NOEC = No Observed Effects Concentration
- LOEC = Lowest Observed Effects Concentration

APPENDIX C: Summary of the Biological Effects of Road Salts on Aquatic Ecosystems

Table C-1: Canada and the United States

Ecological Characteristic or Species Affected	Specific Location	Response to Salt Loading	Duration of Response	Baseline or Upstream Concentrations	Value of Main Loading into Ecosystem	Source of Loading	New Concentration After Loading	Duration of New Concentration	Additional Notes	Reference
Streams [1]										
macro-invertebrate community structure	20 springs in southeastern ON	several taxa associated with high Cl^- levels (e.g., Tipulidae & Ceratopo-gonidae), whereas others (e.g., *Gammarus pseudo-limnaeus* & Turbellaria) found only in springs with low Cl^-	unknown	9.3 mg/L Cl^- (pristine spring at Glen Major Conservation Area)	unknown	ground-water believed to be contaminated with Cl^- from road salt	8.1-1148.6 mg/L Cl^- (for 19 springs)	unknown	length of stream affected unknown; Cl^- concentrations correlated with level of urbanization	Williams et al. 1997; Williams et al. 1999
drift of stream benthic invertebrates	Lutteral Creek, southern ON	increased drift after 1000 mg/L Cl^- in creek	6 hours; drift returned to normal levels after concentration fell below 800 mg/L Cl^-	<100 mg/L Cl^-	unknown	experi-mental application	increase up to 2165 mg/L Cl^- followed by decline	approx. 20 hours	length of stream affected unknown	Crowther & Hynes 1977

Table C-1: Continued

Ecological Characteristic or Species Affected	Specific Location	Response to Salt Loading	Duration of Response	Baseline or Upstream Concentrations	Value of Main Loading into Ecosystem	Source of Loading	New Concentration After Loading	Duration of New Concentration	Additional Notes	Reference
composition and diversity of stream invertebrates	Humber River, northwest Toronto, ON	no affect; road salting in area was lower than usual during time of study	unknown	13.8-24.1 mg/L Cl^- (Albion Hills; February and March, 1980	unknown	winter use of road salt	17.0-34.8 mg/L Cl^- (Cedar Mills; February and March, 1980)	unknown	-	Kersey 1981
density of bacteria and algae	Heyworth Stream, near Heyworth, PQ	increased bacterial density and decreased algal density with higher salt concentration	for algae: last 3 weeks of 4 week experiment; unknown for bacteria	2-3 mg/L NaCl (upstream)	unknown	experi-mental application	~1000 mg/L NaCl	experiment = 28 days	length of stream affected unknown	Dickman and Gochnauer 1978
benthic invertebrate community	Sugar Creek, WI, USA	increased chloride and sodium concentration	see Duration of New Concentration	unknown	unknown	road salt	53 mg/L Cl^- and 28 mg/L Na^+ (peak concentration)	peak concentrations occurred after a snow melt event	stream velocity 0.23-0.64 m/sec; differences in the abundance and diversity of benthic invertebrates (pollution sensitive taxa were especially different between sites)	Smith and Kaster 1983

Table C-1: Continued

Ecological Characteristic or Species Affected	Specific Location	Response to Salt Loading	Duration of Response	Baseline or Upstream Concentrations	Value of Main Loading into Ecosystem	Source of Loading	New Concentration After Loading	Duration of New Concentration	Additional Notes	Reference
diversity of aquatic insects colonizing artificial substrates	4 streams by town of Newcomb in Adirondack region of northern NY, USA	decreased diversity in downstream versus upstream locations	unknown	0.61 mg/L Cl^- (overall mean)	unknown	winter use of road salt	5.23 mg/L Cl^- (overall mean)	unknown	smaller flow rate, higher downstream Cl^- concentration, and greater difference in diversity between upstream and downstream	Demers 1992
benthic invertebrate abundance and biomass	Rio en Medio Stream, Sante Fe Ski Basin, Sante Fe, NM, USA	increased sodium and chloride concentration	see Duration of New Concentration	1.89 mg/L Na^+; 0.38 mg/L Cl^-; conductivity = 31.9 µS/cm (upstream concentration);	unknown	road salt	4.87 mg/L Na^+; 8.61 Cl^-; conductivity = 68.8 µS/cm (200 m below road); 3.45 mg/L Na^+; 5.63 Cl^-; conductivity = 55.2 µS/cm (2.2 km below road)	highest concentrations in area 3 from February to April; yearly averages greater for Area 3 than Area 1	at least 2.2 km	Molles and Gosz 1980
alteration of algal community	various streams in Cumbria and North Lancashire, England	pollution from road runoff	unknown	unknown	unknown	runoff from M6 motorway	unknown	unknown	algal abundance and diversity was affected, but it is not clear if this change was due to road salt or increased nutrient concentrations	Dussart 1984

Table C-1: Continued

Ecological Characteristic or Species Affected	Specific Location	Response to Salt Loading	Duration of Response	Baseline or Upstream Concentrations	Value of Main Loading into Ecosystem	Source of Loading	New Concentration After Loading	Duration of New Concentration	Additional Notes	Reference
water quality, sediment quality and biota	Pidgeon Bridge Brook, Butterthwaite Ditch and Rockley Dike, northern England	increased chloride concentration	unknown	65.1-105.7 mg/L Cl^- (upstream sites)	unknown	runoff from M1 motorway	86.2-229.1 mg/L Cl^-(downstream sites)	unknown	other contaminants (e.g., lead and zinc) were also elevated	Matlby et al. 1995a; b
Lakes[2]										
meromixis and changes in algal community	Little Round Lake, ON	incomplete vertical mixing; cultural eutrophication	unknown	unknown	unknown	winter application of road salt	monolimnion concentration 58.4 mg/L Na^+, 103.7 mg/L Cl^-	approximately 30 years	pre-settlement algae = *Cyclotella* spp. and *Mallomonas pesudocoronata*; post-settlement algae = *Stephanodiscus hantzschii* and trace levels of *Synedra* spp.	Smol et al. 1983
incomplete spring vertical mixing (salt-induced density gradient; changes in benthic invertebrate density	Lake Wabekayne, a storm-water impoundment for Mississauga, ON	anoxic conditions in bottom waters and decreased benthic invertebrate diversity	primarily in spring	50 mg/L Cl^- in August, 1979	unknown	urban road salt application in winter	282 mg/L Cl^- in February, 1979	concentrations elevated for approx. 3 months (February to April), especially on bottom	surface area = 1.9 ha; mean depth = 1.84 m; volume = 3.5×10^4 m^3	Free & Mulamoottil 1983

Table C-1: Continued

Ecological Characteristic or Species Affected	Specific Location	Response to Salt Loading	Duration of Response	Baseline or Upstream Concentrations	Value of Main Loading into Ecosystem	Source of Loading	New Concentration After Loading	Duration of New Concentration	Additional Notes	Reference
incomplete vertical mixing due to salt gradient	Fonda Lake, MI, USA	increased salt concentration	unknown	12 mg/L (Frains Lake); 15 mg/L (Portage Lake)	unknown	seepage from salt storage facility	235 mg/L	unknown	asphalt pad constructed in early 1970s, reducing salt input, but still remained elevated	Tuchman et al. 1984; Zeeb and Smol 1991
Wetlands										
alterations of plant species diversity	Pinhook Bog, LaPorte County, IN (in Indiana Dunes National Lakeshore)	absence of numerous native species such as *Sphagnum* spp. and *Larix laricina* and invasion of salt tolerant species such as *Typha angustifolia*	late 1960s to 1980 (when salt storage at the site was discontinued)	5-6 mg/L Cl^- (control sites; 1980 and 1981)	total Cl^- inputs to bog over 10 year period: from salt pile = 2.3 million kg; from road salting = 0.4 million kg; from direct precipi-tation = 0.012 million kg	road salt storage pile, road salting of nearby highway, natural precipi-tation	max. single daily reading: 1468 mg/L Cl^- in 1979; 982 mg/L in 1980; 570 mg/L in 1981	late 1960s to 1980 (when salt storage at the site was discontinued)	area = 44 ha	Wilcox 1982

Table C-1: Continued

Ecological Characteristic or Species Affected	Specific Location	Response to Salt Loading	Duration of Response	Baseline or Upstream Concentrations	Value of Main Loading into Ecosystem	Source of Loading	New Concentration After Loading	Duration of New Concentration	Additional Notes	Reference
number of egg masses deposited by spotted salamanders (*Ambystoma maculatum*)	35 temporary wetlands in central Pennsylvania	lower number of egg masses with higher concentration of total cations (Na^+, K^+, Mg^{2+}, Ca^{2+})	see Duration of New Concentration	0.337 mg/L Na^+; 0.240 mg/L K^+; 0.478 mg/L Mg^{2+}; 0.220 mg/L Ca^{2+} (minimum concentration); maximum number of egg masses = 456	unknown	unknown; although higher Ca^{2+} may be associated with presence of limestone bedrock	0.788 mg/L Na^+; 0.1.903 mg/L K^+; 0.820 mg/L Mg^{2+}; 7.282 mg/L Ca^{2+} (maximum concentration); minimum number of egg masses = 0	March to May, 1991	-	Rowe and Dunson 1993

[1] streams include streams, creeks, springs, and rivers.
[2] lakes include lakes and ponds.

Environment Canada / Environnement Canada

Canada

Canada Centre for Inland Waters
P.O. Box 5050
867 Lakeshore Road
Burlington, Ontario
L7R 4A6 Canada

Centre canadien des eaux intérieures
Case postale 5050
867, chemin Lakeshore
Burlington (Ontario)
L7R 4A6 Canada

National Hydrology Research Centre
11 Innovation Boulevard
Saskatoon, Saskatchewan
S7N 3H5 Canada

Centre national de recherche en hydrologie
11, boul. Innovation
Saskatoon (Saskatchewan)
S7N 3H5 Canada

St. Lawrence Centre
105 McGill Street
Montreal, Quebec
H2Y 2E7 Canada

Centre Saint-Laurent
105, rue McGill
Montréal (Québec)
H2Y 2E7 Canada

Place Vincent Massey
351 St. Joseph Boulevard
Gatineau, Quebec
K1A 0H3 Canada

Place Vincent-Massey
351 boul. St-Joseph
Gatineau (Québec)
K1A 0H3 Canada