CCME

Canadian Council of Ministers of the Environment
Le Conseil canadien des ministres de l'environnement

Scientific Criteria Document for the Development of the Canadian Water Quality Guidelines for the Protection of Aquatic Life

CHLORIDE ION

PN 1460

ISBN 978-1-896997- 77-3 PDF

NOTE TO READERS

The Canadian Council of Ministers of the Environment (CCME) is the major intergovernmental forum in Canada for discussion and joint action on environmental issues of national concern. The 14 member governments work as partners in developing nationally consistent environmental standards and practices.

This document provides the background information and rationale for the development of the Canadian Water Quality Guidelines for the chloride ion. For additional scientific information regarding these water quality guidelines, please contact:

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This scientific supporting document is available in English only. Ce document scientifique du soutien n'est disponible qu'en anglais avec un résumé en français.

Reference listing:

CCME. 2011. Canadian Water Quality Guidelines : Chloride Ion. Scientific Criteria Document. Canadian Council of Ministers of the Environment, Winnipeg.

PN 1460
ISBN 978-1-896997- 77-3 PDF

ACKNOWLEDGEMENTS

This document was prepared under the direction of the National Guidelines and Standards Office (NGSO) of Environment Canada. An original draft of this document was prepared by Cecilia Tolley and Ruth Hull (of Intrinsik). Additional work and completion of the document was done by Monica Nowierski and Tim Fletcher, both of the Standards Development Branch of the Ontario Ministry of the Environment. Thanks are extended to the following individuals for their help in reviewing and commenting on the chloride Water Quality Guideline Supporting Document:

- Doug Spry and Uwe Schneider (National Guidelines Standards Office, Environment Canada)
- Yves Couillard (Science and Technology Branch, Environment Canada)
- Patty Gillis (National Water Research Institute, Environment Canada)
- Lise Trudel (Products Division, Environment Canada)
- Emilie Morin (Environmental Stewardship Branch, Environment Canada)
- Aaron Todd (Environmental Monitoring and Reporting Branch, Ontario Ministry of the Environment)
- Cindy Meays (British Columbia Ministry of the Environment)
- Cindy Crane (Water Management Division, PEI Department of Environment, Energy & Forestry)
- O.S. (Arasu) Thirunavukkarasu (Saskatchewan Ministry of Environment)
- William Dimond (Water Bureau, Michigan Department of Environmental Quality)
- Charles Stephan (United States Environmental Protection Agency)
- Bruce Kilgour (Kilgour and Associates Ltd.)
- Morton Satin (Salt Institute)
- Scott Hall (ENVIRON International Corporation)
- James Elphick (Nautilus Environmental)

Thanks are also extended to the following:

- Ken Doe (Environment Canada) for assistance with the review of both short-term and long-term amphibian studies and for providing data for a long-term study with the Northern leopard frog (*Rana pipiens*), and Paula Jackman (Environment Canada) for providing NaCl reference toxicant data for *Daphnia magna*, Leopard frog tadpoles (*Lithibates pipiens* previously *Rana pipiens*), and Wood frog tadpoles (*Lithibates sylvatica* previously *Rana sylvatica*).
- Gerry Mackie (Professor emeritus, Department of Integrative Biology, University of Guelph) for providing expert advice with respect to inclusion of freshwater mussel glochidia data for the derivation of the chloride short-term benchmark concentration and long term guideline.
- Bob Truelson (Water Resources, Yukon Department of Environment) for providing chloride water quality monitoring data.
- Robert Bringolf (Assistant Professor, University of Georgia) for providing freshwater mussel glochidia data.
- Chris Ingersoll and Ning Wang (United States Geological Survey) for providing NaCl reference toxicity data for 12 invertebrates.

- Darrell Taylor (Nova Scotia Department of Environment) and Denis Parent (Enviroment Canada) for providing ambient chloride measurements for lakes and streams in Nova Scotia.

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BC MOE	British Columbia Ministry of the Environment
CA	Conservation Authority
$CaCl_2$	Calcium Chloride
$CaCO_3$	Calcium Carbonate; where water hardness is presented as milligrams of $CaCO_3$ per litre
CALA	Canadian Association for Laboratory Accreditation
CCC	The Criterion Continuous Concentration (CCC) is the US EPA national chronic water quality criteria recommendation for the highest concentration of a material in surface water to which an aquatic community can be exposed indefinitely without resulting in an unacceptable effect.
CCME	Canadian Council of Ministers of the Environment
CCOHS	Canadian Centre for Occupational Health and Safety
CEPA	*Canadian Environmental Protection* Act
CMC	The Criteria Maximum Concentration (CMC) is the US EPA national acute water quality criteria recommendation for the highest concentration of a material in surface water to which an aquatic community can be exposed briefly without resulting in an unacceptable effect.
COSEWIC	Committee on the Status of Endangered Wildlife in Canada
CWQG	Canadian Water Quality Guideline
DWSP	Drinking Water Surveillance Program; Ontario
EC	Environment Canada
EC_X	Effective Concentration; the concentration which causes the specified (X) percentage of the population of the experimental biota to show an observed effect. The effect may be immobilization, changes in reproductive potential, growth, or some other ecologically relevant endpoint.
ERL Duluth	Environmental Research Laboratory Duluth
FAV	The Final Acute Value (FAV) is an estimate of the concentration of a toxicant corresponding to a cumulative probability of 0.05 in the acute toxicity values for all genera for which acceptable acute tests have been conducted on the toxicant. The FAV is used in the derivation of both the US EPA acute (CMC) and chronic (CCC) water quality criteria. The acute criterion is equal to one-half of the FAV. The chronic criterion is determined by dividing the FAV by a final acute-to-chronic ratio.
FHWA	Federal Highway Administration; U.S. Department of Transportation
FL	Fiducial Limit; reported along with the HC5 or guideline value, and are similar to confidence intervals. FLs help assess the fit of the selected curve or model to the dataset.

	As the number of data points plotted on an SSD increases, the fit of FLs should be tighter. FLs can also be used to help interpret monitoring data, particularly if the guideline and method detection limit are close. Only the HC5 is used as the guideline.
GLEC	Great Lakes Environmental Centre
IC_X	Inhibitory Concentration; the concentration of inhibitor which causes the specified percentage (X) of inhibition in the target (e.g. molecule, enzyme, cell, microorganism, etc.)
INHS	Illinois Natural History Survey
KCl	Potassium Chloride
log K_{ow}	Octanol-water partition coefficient; is the ratio of the concentration of a chemical in octanol and in water at equilibrium and at a specified temperature and is used as a surrogate to estimate bioaccumulative potential of a chemical. In general, the higher the log K_{ow} value, the more bioaccumulative the chemical.
LC_X	Lethal Concentration; the concentration which is lethal to the specified (X) percentage of the experimental biota.
LOEC	Lowest Observed Effect Concentration; the lowest concentration at which an effect significantly different from control is observed.
MATC	Maximum Acceptable Toxicant Concentration; calculated as the geometric mean of the NOEC and LOEC, and is regarded as an improved estimate of the actual NOEC.
Max	Maximum measured value
MDDEP	Ministère du Dèveloppement Durable, de l'Environnment et des Parcs; Quèbec
$MgCl_2$	Magnesium Chloride
Min	Minimum measured value
OMOE	Ontario Ministry of the Environment
n	Number of chloride measurements
NaCl	Sodium Chloride
NAQUADAT	National Water Quality Data Bank; Environment Canada
NOEC	No Observed Effect Concentration; the highest concentration at which there is no statistically different response when compared to control.
NLET	National Laboratory for Environmental Testing; Environment Canada
NRC	National Research Council of Canada
OECD	Organisation for Economic Co-operation and Development
PMRA	Pest Management Regulatory Agency
PWQMN	Provincial Water Quality Monitoring Network; Ontario
SARA	*Species at Risk* Act
SSD	Species Sensitivity Distribution
SD	Standard Deviation
TLm	Median Tolerance Limit; the concentration of material at which 50% of test organisms survive after a specified time of exposure (e.g. 96 hours). The TLm has been replaced

	by the LC50 in current scientific literature.
US EPA	United States Environmental Protection Agency
WHO	World Health Organization
WISLOH	Wisconsin State Laboratory of Hygiene

EXECUTIVE SUMMARY

Chloride occurs in the natural environment as salts of sodium (NaCl), potassium (KCl), calcium ($CaCl_2$), and magnesium ($MgCl_2$). The chloride ion is naturally occurring, and therefore detection of increased levels of chloride in surface waters does not necessarily imply an anthropogenic source. Natural sources of chloride in aquatic systems include naturally-occurring saline lakes and groundwater discharges from saline aquifers. Canada has many known naturally occurring salt deposits. Major salt (marine evaporite) deposits are found in Nova Scotia, New Brunswick, Quebec, Ontario, Manitoba, Saskatchewan and Alberta. In Canada, marine evaporate deposits include the Salina Formation in Ontario (halite [NaCl] and gypsum [$CaSO_4 \bullet 2H_2O$]), the Windsor Group in the Appalachian region (halite [NaCl], sylvite [KCl], gypsum [$CaSO_4 \bullet 2H_2O$], celestite [$SrSO_4$]), in the Prairie Formation in Saskatchewan (sylvite [KCl], halite [NaCl], brine), at Gypsumville Manitoba (gypsum [$CaSO_4 \bullet 2H_2O$]) and at Windermere British Columbia (gypsum [$CaSO_4 \bullet 2H_2O$]). Other natural sources include volcanic emanations, sea spray, seawater intrusion in coastal areas, as well as wildfires and logging (remobilization of major ions in lake watersheds impacted by these perturbations).

A major non-industrial anthropogenic source of chloride to the environment is the application and storage of road salts for snow and ice control in the winter, especially in highly urbanized areas of Canada. It is estimated that 97% of road salt used in Canada is in the form of NaCl, 2.9% in the form of $CaCl_2$, and 0.1% as $MgCl_2$ and KCl. Road salt is the single largest use of salt and the largest non-industrial source of chloride loading to the environment in highly urban areas. In the winter of 1997 to 1998, an estimated 4,750,000 tonnes of sodium chloride and 110,000 tonnes of calcium chloride were used for the deicing of Canadian roads. An often unquantified and significant use of road salt is that what is applied as a result of private deicing operations, for example, applications onto sidewalks, driveways, and parking lots. Elevated concentrations of chloride associated with deicing have been documented in groundwater, wetlands, streams, and ponds adjacent to snow dumps and salt-storage areas, and also those draining major roadways and urban areas in Canada. Other sources also include disposal of snow cleared from roadways and application of chloride brine solutions for dust suppression in the summer. Additonal examples include oil sands operations, municipal wastewater effluent, diamond mining, industrial effluent, domestic sewage, landfill leachates, and irrigation drainage.

Chloride-containing salt compounds are highly soluble and easily dissociate into the chloride anion and corresponding cations. Once in surface water, chloride is not susceptible to degradation, and does not adsorb to sediment, therefore concentrations can remain high in surface water and sediment pore water. Overall, inorganic chloride is generally considered to be a hydrologically and chemically inert substance. Fairly recent research has revealed that a large portion of inorganic chloride that is deposited in terrestrial environments is transformed to organic chloride (chlorinated organic matter) in soil or vegetation (and vice versa), although the underlying mechanisms are not fully understood. Investigations are underway in order to understand how anthropogenic sources of chloride influence this biogeochemical cycling, whether it enhances or diminishes the natural formation of chlorinated compounds. High chloride concentrations in wetlands and stormwater management ponds can lead to the development of meromixis (chemical induced stratification resistant to mixing). High chloride can also exacerbate meromixis in inland lakes (where lakes do not experience complete overturn or complete vertical mixing) that are meromictic due to natural hydrological and geological conditions (e.g. Little Round Lake in Ontario). Meromixis can result in low to no dissolved oxygen in the bottom layers of water bodies (near the sediment-water interface). The resulting anaerobic condition can be detrimental to organisms that reside at the sediment-water interface. The anaerobic environment can also lead to increased mobilization of metals from sediments, causing increased levels of dissolved metals in solution.

Ambient chloride concentrations in the Atlantic region (Newfoundland and Labrador, Nova Scotia, New Brunswick and Prince Edward Island) of Canada are normally <10 mg/L in inland lakes, with concentrations as high as 20 mg/L in lakes located closer to coastal areas. Unimpacted lakes on the Canadian shield of Canada's central region (Quebec and Ontario) have measured chloride concentrations of <1 to 7 mg/L, with higher concentrations (10 to 30 mg/L) measured in the lower Great Lakes and the St. Lawrence River. Chloride concentrations above background are commonly detected in densely populated areas (e.g. small urban watersheds) where road densities are high, and in fact is a commonly used indication of increasing urbanization. In the case of Canada's prairie region (Manitoba, Saskatchewan, and Alberta), low chloride concentrations (<5 mg/L) are reported in lakes located in the northern portions of the provinces outside of the Interior Plains Region. However, this region is also an area with naturally elevated salinity (total dissolved solids) due to the underlying geology, where inland lakes have measured chloride concentrations as high as 33,750 mg/L. The measured mean chloride concentrations are substantially lower, with measurements of 71, 1,914, 1,028 and 3,793 mg Cl^-/L for the Eastern Prairies, Central Saskatchewan, South-west Saskatchewan/South-east Alberta and West-central Saskatchewan/East-central Alberta, respectively. In areas such as this, where natural background levels of the chloride ion can potentially exceed the guideline value, a site-specific guideline (or objective) can be derived. An important point to note is that the saline lakes located within Canada's northern prairie region (stretching from Winnipeg, Manitoba, westward to the Rocky mountain foothills) are mostly dominated by sulphate or bicarbonate/carbonate anions, with variation in the predominant cations. Chloride dominated saline lakes are more rare and are located in northern Alberta, with a few also located in the Saskatchewan River Delta and on the interior plateau of British Columbia. For the Pacific region (British Columbia), the chloride concentration in unimpacted water bodies is <5 mg/L, however, several lakes in the southern interior plateau had measured chloride concentrations >100 mg/L. Water quality monitoring data in the Yukon showed that dissolved chloride concentrations are low, ranging from 0.1 to 4.6 mg/L. No chloride monitoring data were found for the Northwest Territories or Nunavut.

There is a strong need to develop a CWQG for chloride. The Priority Substances List Assessment Report for Road Salts was published on December 1, 2001. The report concluded that Road Salts that contain inorganic chloride salts with or without ferrocyanide salts have adverse impacts on the environment and are therefore toxic under subsections 64(*a*) and (*b*) of the *Canadian Environmental Protection Act, 1999* (CEPA 1999). This decision has led to the publication in April 2004, of a Code of Practice for the Environmental Management of Road Salts. This Code of Practice is aimed at helping municipalities and other road authorities better manage their use of road salts in a way that reduces the harm they cause to the environment while still maintaining road safety. As well, monitoring data strongly indicates that chloride concentrations in surface waters are increasing, especially in small urban watersheds where road densities are high. This is true for all regions of Canada, where studies have indicated that lakes and rivers in developed watersheds were found to have elevated chloride concentrations compared to lakes and rivers located in rural areas. This is a result of continuous seasonal road salt application, whereby chloride is accumulating in the environment with each successive winter. The application of road salts is beneficial for ensuring road safety, however, maintaining healthy water supplies and healthy aquatic ecosystems is also of great benefit.

Aquatic toxicity tests assessing the affects of the chloride ion have been conducted through the addition of chloride salts such as sodium chloride (NaCl), calcium chloride ($CaCl_2$), magnesium chloride ($MgCl_2$) and potassium chloride (KCl). Results of tests with KCl and $MgCl_2$ suggest toxic effects observed are due to the K^+ and Mg^{2+} cation, rather than the Cl^- anion. Conversely, it has been observed that the effects of $CaCl_2$ and NaCl are likely due to the Cl^- anion. Generally speaking, the approximate order of chloride salt toxicity to freshwater organisms is KCl > $MgCl_2$ > $CaCl_2$ > NaCl. Based on these observations, chloride toxicity to freshwater organisms was only evaluated using tests with $CaCl_2$ and NaCl. As well, sources of $CaCl_2$ (e.g. dust suppressants) and NaCl (e.g. road salt) are one of the most significant anthropogenic non-industrial sources of chloride to the aquatic environment, specifically in densely populated regions of Canada.

In the case of the short-term toxicity data, 1 species of freshwater mussel (tested at the glochidia life-stage, and COSEWIC assessed as endangered) was found to be more sensitive to short-term chloride exposure when compared to a daphnid species (*Daphnia magna*, neonate life-stage). The short-term data met the toxicological and statistical requirements for the SSD (Type A) guideline derivation method. The log-Normal model was used for short-term benchmark concentration derivation. A total of 51 data points (both LC50 and EC50 values) from 51 species were used in the derivation of the short-term benchmark concentration. In general, invertebrate species were found to be grouped towards the lower end of the short-term SSD, while the fish species were grouped towards the upper end of the short-term SSD. This can be interpreted as invertebrates being more sensitive to acute chloride exposures when compared to fish.

In the case of the long-term toxicity data, a similar pattern with respect to chloride sensitivity was observed. Two species of freshwater mussels (all tested at the glochidia life-stage, with one mussel designated as COSEWIC endangered and a second as COSEWIC special concern) and 1 species of freshwater clam (newborn life-stage) were found to be more sensitive to long-term chloride exposures when compared to a daphnid species (*Daphnia ambigua*, neonate life-stage). The long-term data met the toxicological and statistical requirements for the SSD (Type A) guideline derivation method. The log-Logistic model was used for long-term guideline derivation. A total of 28 data points (including L/EC10, MATC, NOEC, E/IC25, LOEC values) from 28 species were used in the derivation of the guideline. In general, the most sensitive invertebrate species (mussels, clams, daphnids and amphipods) were grouped towards the lower end of the long-term SSD, with the fish species grouped midway. Algal species were found to be the most tolerant of long-term chloride exposures, as these were grouped towards the upper end of the long-term SSD.

Toxicity testing with non-traditional bioassay organisms has indicated that daphnids may not be the most sensitive species to both short-term and long-term chloride exposures, as traditionally thought.

Neither a short-term benchmark concentration nor a long-term guideline were developed for marine waters. Sea water salt concentrations are approximately 35,000 mg/L of which approximately 55% is chloride, which equates to 19,250 mg chloride/L. For this reason, brine discharges to marine environments were not evaluated.

Canadian Water Quality Guideline for the chloride ion[a] for the protection of aquatic life

	Long-Term Exposure [b] (mg Cl^-/L)	**Short-Term Exposure[c] (mg Cl^-/L)**
Freshwater	120[d]	640
Marine	NRG	NRG

[a]Derived from toxicity tests utilizing both $CaCl_2$ and NaCl salts

[b]Derived with mostly no- and some low-effect data and are intended to protect against negative effects to aquatic ecosystem structure and function during indefinite exposures (e.g. abide by the guiding principle as per CCME 2007).

[c]Derived with severe-effects data (such as lethality) and are not intended to protect all components of aquatic ecosystem structure and function but rather to protect most species against lethality during severe but transient events (e.g. inappropriate application or disposal of the substance of concern).

[d]The long-term CWQG may not be protective of certain species of endangered and special concern freshwater mussels (as designated by the Committee on the Status of Endangered Wildlife in Canada, or COSEWIC). This specifically applies to two species; the wavy-rayed lampmussel (*Lampsilis fasciola*) (COSEWIC, 2010a) and the northern riffleshell mussel (*Epioblasma torulosa rangiana*) (COSEWIC, 2010b) (table below). The wavy-rayed lampmussel is indigenous to the lower Great Lakes and associated tributaries, specifically western Lake Erie, the Detroit River, Lake St. Clair and several southwestern Ontario streams. The northern riffleshell mussel is indigenous to the Ausable, Grand, Sydenham and Thames Rivers in Ontario, as well as the Lake St. Clair delta. Discussion with provincial regulators should occur if there is a need to develop more protective site specific values.

NRG = no recommended guideline

24h EC10 values (survival of glochidia) for 2 species of COSEWIC assessed freshwater mussels.

COSEWIC Assessed Species	24h EC10 (mg Cl^-/L)	95% Confidence Intervals	Reference
Lampsilis fasciola Wavy-rayed lampmussel (COSEWIC special concern)	24	-79[1], 127	Bringolf, 2010
Epioblasma torulosa rangiana Northern riffleshell mussel (COSEWIC endangered)	42	24, 57	Gillis, 2009

[1] The negative lower fiducial limit is an artefact of the statistics. Biologically this can be interpreted as meaning that a 10% effect can be observed between a concentration of 0 and the upper 95% confidence limit. Therefore, the effect is not significantly different from the control (no-effect concentration) and could be due to natural variability.

The short-term benchmark concentration and long-term CWQG for chloride are set to provide protection for short- and long-term exposure periods, respectively. They are based on generic environmental fate and behaviour and toxicity data. The guideline is a conservative value below which all forms of aquatic life, during all life stages and in all Canadian aquatic systems, should be protected. Because the guideline is not corrected for any toxicity modifying factors (e.g. hardness), it is a generic value that does not take into account any site-specific factors. Moreover, since it is mostly based on toxicity tests using naïve (i.e., non-tolerant) laboratory organisms, the guideline may not be relevant for areas with a naturally elevated concentration of chloride and associated adapted ecological community. Thus, if an exceedence of the guideline is observed (due to anthropogenically enriched water or because of elevated natural background concentrations), it does not necessarily suggest that toxic effects will be observed, but rather indicates the need to determine whether or not there is a potential for adverse environmental effects. In some situations, such as where an exceedence is observed, it may be necessary or advantageous to derive a site-specific guideline that takes into account local conditions (water chemistry such as hardness, natural background concentration, genetically adapted organisms, community structure).

The guideline should be used as a screening and management tool to ensure that chloride does not lead to the degradation of the aquatic environment. The CWQG for chloride could, for example, be the basis for the derivation of site-specific guidelines and objectives (derived with site-specific data as well as consideration of technological, site-specific, socioeconomic or management factors).

RÉSUMÉ

Les chlorures existent dans la nature sous forme de sels – chlorure de sodium (NaCl), chlorure de potassium (KCl), chlorure de calcium ($CaCl_2$) et chlorure de magnésium ($MgCl_2$). L'ion chlorure existe à l'état naturel; par conséquent, la détection de concentrations élevées en chlorures dans les eaux de surface n'indique pas nécessairement la présence d'une source anthropique. Parmi les sources naturelles de chlorures dans les systèmes aquatiques figurent les lacs salins naturels et les rejets d'eaux souterraines provenant d'aquifères salins. On connaît de nombreux dépôts de sels naturels au Canada, dont les principaux (dépôts évaporitiques d'origine marine) se trouvent en Nouvelle-Écosse, au Nouveau-Brunswick, au Québec, en Ontario, au Manitoba, en Saskatchewan et en Alberta. Au Canada, les dépôts évaporitiques se trouvent dans la formation de Salina, en Ontario (halite [NaCl] et gypse [$CaSO_4 \bullet 2H_2O$]), le groupe de Windsor, dans la région des Appalaches (halite [NaCl], sylvite [KCl], gypse [$CaSO_4 \bullet 2H_2O$], célestite [$SrSO_4$]), la formation de Prairie, en Saskatchewan (sylvite [KCl], halite [NaCl], saumure), à Gypsumville, au Manitoba (gypse [$CaSO_4 \bullet 2H_2O$]), et à Windermere, en Colombie-Britannique (gypse [CaSO4•2H2O]). Les émanations volcaniques, les embruns, l'intrusion d'eau de mer dans les zones côtières ainsi que les feux de forêt et l'exploitation forestière constituent d'autres sources naturelles de chlorures (ces perturbations ont une incidence sur la remobilisation des principaux ions dans les bassins versants des lacs).

L'application et le stockage de sels de voirie destinés à éliminer la glace et la neige, pendant la période hivernale, constitue une importante source anthropique non industrielle de chlorures, surtout dans les régions densément peuplées du Canada. On estime que 97 % des sels de voirie employés au Canada sont sous forme de NaCl, 2,9 %, sous forme de $CaCl_2$, et 0,1 %, sous forme de $MgCl_2$ et de KCl. Le principal usage des chlorures se trouve dans les sels de voirie, et cet usage constitue aussi la principale charge non industrielle de chlorures dans l'environnement dans les zones fortement urbanisées. À l'hiver 1997-1998, des quantités approximatives de 4 750 000 tonnes de chlorure de sodium et de 110 000 tonnes de chlorure de calcium ont été épandues sur les chaussées canadiennes pour les déglacer. Une quantité souvent indéfinie, mais considérable de sels de voirie est épandue dans le privé, par exemple sur les trottoirs, les voies d'accès et les terrains de stationnement. Des concentrations élevées de chlorures de déglaçage ont été trouvées dans des eaux souterraines, des milieux humides, des cours d'eau et des étangs qui se trouvent à proximité de décharges à neige et de dépôts de sels, ou qui drainent les principales routes et zones urbaines du Canada. Les autres sources comprennent également les dépôts où l'on stocke la neige enlevée des chaussées ainsi que l'application de solutions de saumure chlorurée dépoussiérante pendant l'été. L'exploitation des sables bitumineux, les effluents d'eaux usées municipales, l'extraction des diamants, les effluents industriels, les eaux usées d'origine domestique, le lixiviat des décharges ainsi que l'irrigation et le drainage pour l'irrigation constituent d'autres exemples de telles sources.

Les sels chlorurés sont très solubles et se dissocient facilement en anions chlorures et en cations. Une fois dans les eaux de surface, les chlorures ne sont pas susceptibles de se

dégrader et ne s'adsorbent pas aux sédiments; il peut donc demeurer en forte concentration dans les eaux de surface et dans l'eau de porosité des sédiments. Dans l'ensemble, les chlorures inorganiques sont généralement considérés comme des substances inertes d'un point de vue hydrologique et chimique. Des recherches relativement récentes ont révélé qu'une vaste portion des chlorures inorganiques déposée en milieu terrestre est transformée en chlorures organiques (matière organique chlorée) dans les sols ou dans la végétation (l'inverse a lieu également), mais les mécanismes par lesquels cela se produit ne sont pas entièrement élucidés. On a entrepris des travaux visant à comprendre comment les sources anthropiques de chlorures influent sur ce cycle biogéochimique, et si ces sources accroissent ou réduisent la formation naturelle de composés chlorés. De fortes concentrations en chlorures dans les milieux humides et les étangs de gestion des eaux pluviales peuvent occasionner la méromixie (stratification imputable à la présence de substances chimiques qui empêchent le mélange des eaux). La forte présence de chlorures peut aussi exacerber la méromixie de lacs intérieurs (lacs où le brassage ou le mélange vertical ne sont pas complets) qui sont déjà méromictiques en raison de facteurs hydrologiques et géologiques naturels (c'est le cas du petit lac Round en Ontario). La méromixie prive d'oxygène les couches de fond des masses d'eau. Le milieu anaérobie qui en résulte nuit aux organismes qui vivent à l'interface eau-sédiments. Il peut aussi accroître la mobilisation des métaux présents dans les sédiments, augmentant la quantité de métaux dissous dans l'eau.

Les concentrations ambiantes de chlorures dans le Canada atlantique (Terre-Neuve, Nouvelle-Écosse, Nouveau-Brunswick et Île-du-Prince-Édouard) sont habituellement inférieures à 10 mg/L dans les lacs intérieurs et peuvent grimper à 20 mg/L dans les lacs près des côtes. Les lacs non perturbés du Bouclier canadien dans le centre du Canada (Québec et Ontario) ont des concentrations mesurées en chlorures de < 1 à 7 mg/L. Les concentrations mesurées sont plus élevées (10 à 30 mg/L) dans les Grands Lacs d'aval et le fleuve Saint-Laurent. Des concentrations en chlorures supérieures aux concentrations de fond sont communément trouvées dans les secteurs très peuplés (p. ex. les petits bassins versants urbains) où les réseaux routiers sont denses. En fait, cette chloruration est souvent prise comme le signe d'une urbanisation croissante. Dans la région des Prairies (Manitoba, Saskatchewan et Alberta), on signale de faibles concentrations en chlorures (< 5 mg/L) dans les lacs situés dans la portion septentrionale des provinces, hors des plaines intérieures. Toutefois, il s'agit aussi d'une région où la salinité naturelle est élevée (matières dissoutes totales) en raison de ses caractéristiques géologiques et où les lacs intérieurs ont des concentrations mesurées en chlorures qui peuvent aller jusqu'à 33 750 mg/L. Les concentrations moyennes en chlorures mesurées sont considérablement plus faibles : 71, 1 914, 1 028 et 3 793 mg Cl^-/L, respectivement, dans l'est des Prairies, le centre de la Saskatchewan, le sud-ouest de la Saskatchewan/sud-est de l'Alberta, ainsi que le centre-ouest de l'Alberta. Dans des secteurs comme ceux-là, où les concentrations de fond naturelles en chlorures peuvent dépasser la recommandation, on peut fixer une recommandation (ou un objectif) propre au site. Il est à noter que les lacs salins situés dans le nord des Prairies canadiennes (soit de Winnipeg, au Manitoba, jusqu'aux contreforts des Rocheuses, vers l'ouest) renferment surtout des anions sulfate et bicarbonate/carbonate, tandis que les principaux cations varient. Les lacs salins où prédomine les chlorures sont plus rares, et on les trouve surtout dans le nord de l'Alberta, sauf pour quelques-uns qui sont situés dans le delta de la rivière Saskatchewan et sur le plateau intérieur de la Colombie-Britannique. Dans la région du Pacifique (Colombie-Britannique), la concentration en chlorures dans les masses d'eau non perturbées est inférieure à 5 mg/L. Toutefois, la concentration mesurée est supérieure à 100 mg/L dans plusieurs lacs du plateau intérieur sud. La surveillance de la qualité de l'eau au Yukon a montré que les concentrations en chlorures dissous sont faibles, soit entre 0,1 et 4,6 mg/L. On n'a pas trouvé de données de surveillance en chlorures pour les Territoires du Nord-Ouest et le Nunavut.

Il est impératif d'élaborer une recommandation canadienne pour la qualité des eaux (RCQE) visant les chlorures. Le rapport d'évaluation de la Liste des substances d'intérêt prioritaire concernant les sels de voirie a été publié le 1er décembre 2001. Le rapport conclut que les sels de voirie qui contiennent des sels inorganiques de chlorures avec ou sans sels de ferrocyanure ont des effets nocifs sur l'environnement et sont donc toxiques selon les alinéas *a)* et *b)* de l'article 64 de la *Loi canadienne sur la protection de l'environnement* (1999). Cette conclusion a mené à la publication, en avril 2004, du Code de pratique pour la gestion environnementale des sels de voirie. Ce code de pratique est destiné à aider les municipalités et autres administrations routières à gérer leur emploi des sels de voirie de façon à moins nuire à l'environnement tout en maintenant la sécurité des routes. Aussi, les données de surveillance indiquent nettement que les concentrations en chlorures augmentent dans les eaux de surface, en particulier dans les petits bassins versants urbains où les réseaux routiers sont denses. Ce constat s'avère dans toutes les régions du Canada; les études indiquent que les lacs et les cours d'eau dans les bassins versants urbanisés ont des concentrations en chlorures élevées par comparaison à celles des lacs et cours d'eau situés en milieu rural. Cela est attribuable à l'application saisonnière continuelle de sels de voirie, qui entraîne une accumulation des chlorures dans l'environnement d'un hiver à l'autre. L'application de sels de voirie améliore la sécurité sur les routes, mais la préservation de la salubrité des réserves en eau ainsi que de la santé des écosystèmes aquatiques est également d'une importance capitale.

Des essais sur la toxicité en milieu aquatique visant à évaluer les effets de l'ion chlorure ont été effectués par l'ajout de sels tels que le chlorure de sodium (NaCl), le chlorure de calcium ($CaCl_2$), le chlorure de magnésium ($MgCl_2$) et le chlorure de potassium (KCl). Selon les résultats des essais avec KCl et $MgCl_2$, les effets toxiques observés seraient imputables aux cations K^+ et Mg^2, plutôt qu'à l'anion chlorure. Inversement, il a été observé que les effets du $CaCl_2$ et du NaCl sont probablement dus à l'anion chlorure. D'une manière générale, la toxicité des sels de chlorures pour les organismes d'eau douce s'ordonne à peu près comme suit : KCl > $MgCl_2$ > $CaCl_2$ > NaCl. D'après ces observations, la toxicité des chlorures pour les organismes d'eau douce n'a été évaluée que par des essais sur le $CaCl_2$ et le NaCl. En outre, les sources de $CaCl_2$ (p. ex., les dépoussiérants) et de NaCl (p. ex., les sels de voirie) constituent l'une des sources anthropiques non industrielles les plus importantes des chlorures se retrouvant dans les milieux aquatiques, surtout dans les régions densément peuplées du Canada.

Pour ce qui est des données sur la toxicité à court terme, on a constaté qu'une espèce de mulettes (ayant fait l'objet d'essais au stade de glochidies et étant désignée en voie de disparition par le COSEPAC) était plus sensible aux expositions à court terme aux chlorures que les daphnies (*Daphnia magna*, stade de néonates). Les données à court terme répondaient aux exigences toxicologiques et statistiques de la méthode de détermination de recommandations d'après la DSE (type A). Le modèle log-normal a été employé pour le calcul de la concentration limite pour une exposition à court terme. Au total, 51 valeurs (CL_{50} et CE_{50}) concernant 51 espèces ont été utilisées pour établir la concentration limite pour une exposition à court terme. De manière générale, on a observé que les espèces d'invertébrés étaient regroupées dans la portion inférieure de la DSE à court terme, tandis que les espèces de poissons se concentraient dans la portion supérieure de la DSE à court terme. On peut en conclure que les invertébrés sont plus sensibles à une exposition à des concentrations aiguës en chlorures que les poissons.

Pour ce qui est des données sur la toxicité à long terme, on a noté une répartition similaire quant à la sensibilité aux chlorures. On a constaté que deux espèces de mulettes (toutes ayant fait l'objet d'essais au stade de glochidies, une espèce étant désignée en voie de disparition par le COSEPAC, et l'autre, désignée préoccupante par le même organisme) et une espèce de clam d'eau douce (au stade de néonates) étaient plus sensibles aux expositions à long terme aux chlorures que les daphnies (*Daphnia ambigua*, stade de néonates). Les données à long terme répondaient aux exigences toxicologiques et statistiques de la méthode de détermination de recommandations d'après la DSE (type A). Le modèle log-logistique a été employé pour le calcul de la recommandation. Au total, 28 valeurs (L/CE_{10}, CMAT, CSEO, E/CI_{25}, CMEO) concernant 28 espèces ont été utilisées pour établir la recommandation. De manière générale, on a observé que les espèces d'invertébrés les plus sensibles (mulettes, clams, daphnies et amphipodes) étaient regroupées dans la portion inférieure de la DSE à long terme, tandis que les espèces de poissons se concentraient dans la portion médiane. Il a été déterminé que les algues sont les espèces qui supportent le mieux les expositions à long terme aux chlorures; en effet, elles sont regroupées dans la portion supérieure de la DES à long terme.

Les essais de toxicité chez des organismes d'essai non traditionnels ont montré que les daphnies ne sont peut-être pas les espèces les plus sensibles aux expositions de courte et de longue durée aux chlorures, contrairement à ce qu'on croyait auparavant.

Il n'a pas été établi de concentration limite à court terme ni de recommandation à long terme pour le milieu marin. Les concentrations de sel dans l'eau de mer atteignent environ 35 000 mg/L, dont approximativement 55 % des chlorures, soit 19 250 mg Cl^-/L. Les rejets de saumure en milieu marin n'ont donc pas fait l'objet d'une évaluation.

Recommandation canadienne à long terme pour la qualité de l'eau et concentration limite à court terme d'ion chlorure[a] aux fins de la protection de la vie aquatique

	Recommandation canadienne pour la qualité des eaux à long terme[b] (mg Cl^-/L)	Concentration limite concernant l'exposition à court terme (mg Cl/L)[c]
Eau douce	120[d]	640
Eau de mer	AR	AR

[a]D'après les essais de toxicité sur des sels de $CaCl_2$ et de NaCl.
[b]Valeur établie d'après des concentrations principalement sans effet et quelques concentrations avec faible effet; elle n'est pas destinée à protéger contre les effets néfastes sur la structure et le fonctionnement de l'écosystème aquatique associés à des expositions de durée indéfinie (c'est-à-dire en conformité avec le principe directeur défini dans CCME [2007]).
[c]Valeur établie d'après des données sur les effets graves (comme la létalité) et non destinée à protéger toutes les composantes de la structure et du fonctionnement de l'écosystème aquatique, mais plutôt à protéger la plupart des espèces contre les effets létaux lors d'épisodes d'exposition grave, mais transitoire (par exemple, l'application ou l'élimination inappropriée d'une substance préoccupante).
[d]La RCQE pourrait ne pas assurer la protection de certaines espèces de mulettes désignées en voie de disparition ou préoccupantes (par le Comité sur la situation des espèces en péril au Canada, ou COSEPAC), en particulier deux espèces : la lampsile fasciolée (*Lampsilis fasciola*) (COSEPAC, 2010a) et l'épioblasme ventrue (*Epioblasma torulosa rangiana*) (COSEPAC, 2010b) (tableau 2). La lampsile fasciolée est une espèce indigène des Grands Lacs inférieurs et de leurs affluents, plus précisément de l'ouest du lac Érié, de la rivière Détroit, du lac Sainte-Claire et de plusieurs cours d'eau du sud-ouest de l'Ontario. L'épioblasme ventrue est une espèce indigène des rivières Ausable, Grand, Sydenham et Thames en Ontario, ainsi que du delta du lac Sainte-Claire. Les organismes de réglementation provinciaux doivent être consultés s'il s'avère nécessaire de définir des valeurs procurant une plus grande protection à des sites en particulier.
AR = aucune recommandation.

Valeurs de CE_{10} (survie des glochidies) pour deux espèces de mulettes évaluées par le COSEPAC.

Espèces évaluées par le COSEPAC	CE_{10} sur 24 h (mg Cl^-/L)	Limites de confiance à 95 %	Référence
Lampsilis fasciola Lampsile fasciolée (espèce désignée préoccupante par le COSEPAC)	24	-79[1], 127	Bringolf, 2010
Epioblasma torulosa rangiana Épioblasme ventrue (espèce désignée en voie de disparition par le COSEPAC)	42	24, 57	Gillis, 2009

[1] La borne inférieure négative de l'intervalle de confiance est le résultat du calcul statistique effectué. D'un point de vue biologique, on peut considérer que cela signifie qu'un effet sur 10 % des sujets peut être observé à une concentration située entre 0 et la borne supérieure de l'intervalle de confiance à 95 %. Par conséquent, l'effet n'est pas significativement différent de celui observé chez les témoins (concentration sans effet), et il pourrait être attribuable à la variabilité naturelle.

La concentration limite pour une exposition à court terme ainsi que la RCQE à long terme établie pour les chlorures ont été fixées de manière à assurer une protection contre les expositions à court terme et à long terme, respectivement. Elles sont fondées sur des données génériques concernant le devenir, le comportement et la toxicité dans l'environnement. La recommandation canadienne pour la qualité des eaux est une valeur prudente en deçà de laquelle toutes les formes de vie aquatique, à tous leurs stades de vie et dans tous les systèmes aquatiques au Canada, doivent être protégées. Comme la recommandation n'est corrigée en fonction d'aucun facteur modifiant la toxicité (p. ex. la dureté), elle constitue une valeur générique ne prenant pas en compte les éventuels facteurs propres à un site. En outre, la recommandation étant principalement fondée sur des essais toxiques portant sur des sujets de laboratoire naïfs (c'est-à-dire non tolérants), elle pourrait conférer une protection excessive dans les secteurs où la concentration en chlorures est élevée à l'état naturel et où la biocénose est adaptée à ces conditions (CCME, 2007). Par conséquent, s'il y a dépassement de la recommandation (en raison d'un apport d'origine humaine dans l'eau ou de concentrations de fond naturellement élevées), cela ne signifie pas nécessairement que des effets de toxicité seront observés, mais bien plutôt qu'il faut vérifier si des effets néfastes se produisent ou non dans l'environnement. Dans certains cas, par exemple lorsqu'il y a dépassement, il peut être nécessaire ou profitable de calculer une recommandation propre au site prenant en considération les conditions locales (chimie de l'eau, concentrations de fond naturelles, organismes génétiquement adaptés, structure de la communauté).

Les recommandations devraient être employées comme outil de dépistage et de gestion afin de prévenir la dégradation des milieux aquatiques par les chlorures. Les RCQE relatives aux chlorures pourraient, par exemple, être utilisées pour élaborer des recommandations et des objectifs propres à un site donné (fixés à partir de données propres au site visé ainsi que de facteurs technologiques, socioéconomiques, administratifs ou propres à ce site).

1.0 INTRODUCTION

This report describes the development of a Canadian Water Quality Guideline (CWQG) for the chloride ion for the protection of freshwater life. No marine CWQG has been developed at this time. Sea water salt concentrations are approximately 35,000 mg/L of which approximately 55% is chloride, which equates to 19,250 mg chloride/L. For this reason, brine discharges to marine environments were not evaluated. CWQGs are numerical limits based on the benchmarks designed to protect, sustain and enhance the present and potential uses of a water body. CWQGs are used by provincial, territorial, and federal jurisdictions to evaluate water quality issues and manage competing uses of water. The guideline values derived for the chloride ion are intended to protect all forms of aquatic life and all aspects of aquatic life cycles, including the most sensitive life stage of the most sensitive species over the long term.

This document describes production and uses, sources, and pathways for the entry of the more common chloride salts into the Canadian environment. Available data on environmental fate and persistence of the chloride ion are summarised. A comprehensive assessment of the toxicity of the sodium chloride (NaCl) and calcium chloride ($CaCl_2$) salts to aquatic life is also presented to evaluate environmental hazards posed by these chemicals. Together, this information is used, in accordance with “A Protocol for the Derivation of Water Quality Guidelines for the Protection of Aquatic Life 2007” (CCME 2007) to derive numerical water quality guidelines (WQGs) for protection of all aquatic organisms.

The focus of chloride ion toxicity to aquatic organisms is restricted to studies utilizing NaCl and $CaCl_2$ salts, since it has been observed that the toxicity of these salts is attributed to the chloride ion. In the case of salts such as KCl and $MgCl_2$, the toxicity has been attributed to the cation (K^+ or Mg^{2+}), thereby masking the effect of chloride. It is for this reason that discussions in this document have been mostly limited to an assessment of the affects of NaCl and $CaCl_2$ salts.

For a comprehensive overview on the assessment of road salts under CEPA (1999) refer to Environment Canada / Health Canada (2001). For a comprehensive overview on the assessment of the effects of road salts on aquatic ecosystems, refer to Evans and Frick (2001).

2.0 PHYSICAL AND CHEMICAL PROPERTIES

2.1 Chemistry of Chloride Salts

The chloride ion (Cl^-) is the negatively charged chlorine atom (Cl) (CAS No. 7782-50-5, atomic mass 35.45 $g{\cdot}mol^{-1}$) formed when the chlorine atom picks up one electron. The chlorine atom is a halogen (boiling point of 33.9°C), and never exists in free form in the environment (Nagpal *et al*., 2003). The chloride ion commonly occurs as a salt. Some common chloride salts include NaCl, KCl, $MgCl_2$ (for deicing of roads and walkways),

$CaCl_2$ (used as a dust suppressant on roads), $AlCl_2$ (used in municipal drinking water and wastewater treatment facilities for removal of suspended particles and bacteria from the water), and $FeCl_3$ (to enhance the removal of phosphours at wastewater treatment plants). Chloride-containing compounds are highly soluble in water (e.g. solubility of NaCl is 35.7g/100g water at $0^{o}C$), hence they easily dissociate and tend to remain in their ionic forms (e.g. Na^+ and Cl^-) once dissolved in water. The chloride ion is highly mobile, and concentrations in water are not affected by chemical reactions. Hence chloride does not biodegrade, readily precipitate, volatilize, or bioaccumulate. The chloride ion does not adsorb readily onto mineral surfaces and therefore concentrations remain high in surface water and sediment pore water, and low in sediment (Mayer *et al.*, 1999; Evans and Frick, 2001; WHO, 2003) (see physical-chemical properties listed in Table 2.1). Overall, inorganic chloride is generally considered to be a hydrologically and chemically inert substance. However, recent research by Oberg (2006) has revealed that a large portion of inorganic chloride that is deposited in terrestrial environments is transformed to organic chloride (chlorinated organic matter) in soil or vegetation (and vice versa), although the underlying mechanisms are not fully understood.

Table 2.1 Summary of selected physical and chemical properties for chloride ion and selected chloride salts.

Property	Chloride Ion	Sodium Chloride	Calcium Chloride[a]	Potassium Chloride	Magnesium Chloride	Reference
CAS #	7782-50-5	7647-14-5	10043-52-4	7447-40-7	7786-30-3	NaCl: HSDB 2007a $CaCl_2$: HSDB 2003a KCl: HSDB 2007b $MgCl_2$: HSDB 2003b
Molecular formula	Cl^-	NaCl	$CaCl_2$	KCl	$MgCl_2$	
Physical structure	—	Colorless, transparent crystals or white, crystalline powder	Colorless, cubic crystals, granules or fused masses	Colorless, elongated, prismatic, or cubical crystals or as a white granular powder	Thin white to gray granules and/or flakes; colorless or white crystals	
Molecular weight ($g \cdot mol^{-1}$)	35.45	58.44	110.99	74.55	95.21	
Melting point (°C)	—	801	772	771	712 deg C (rapid heating)	
Boiling point (°C)	—	1465	1670	Sublimes at 1500 deg C	1,412 deg C	
Density / Specific gravity	—	2.17 @ 25 deg C	2.152 @ 15 deg C	1.988	2.32	
Solubility in cold water ($g \cdot mL^{-1}$)	—	35.7 g/100 ml of water at 0 deg C	37.1 g/100 ml of water at 0 deg C	34.4 g/100 ml of water at 0 deg C	54.3 g/100 ml of water at 0 deg C	Evans and Frick 2001
pH	—	6.7 to 7.3; its aqueous solution is neutral	—	Of saturated aqueous solution at 15 deg C: about 7	—	NaCl: HSDB 2007a $CaCl_2$: HSDB 2003a KCl: HSDB 2007b
Notes on use	—	•essential nutrient •chemical feedstock used in the manufacturing of sodium hydroxide, soda ash, hydrogen chloride, chlorine, metallic sodium •used in	•to melt ice and snow, dust control on unpaved roads, antifreeze mixtures •used in fire extinguishers, used to preserve wood, stone; in	•electrolyte replenisher •used in aluminum recycling, in the production of potassium hydroxide, in metal electroplating, oil-well drilling mud, snow	•Source of magnesium metal, disinfectants, fire extinguisher, fireproofing wood, magnesium oxychloride cement, refrigerating brines, ceramics, cooling drill tools, textiles	$MgCl_2$: HSDB 2003b

Property	Chloride Ion	Sodium Chloride	Calcium Chloride[a]	Potassium Chloride	Magnesium Chloride	Reference
		ceramic glazes, metallurgy, curing hides, food preservative, mineral waters, soap manufacture (salting out), home water softeners, highway deicing, regeneration of ion-exchange resins, photography, food seasoning, herbicide, fire extinguishing, nuclear reactors, mouthwash, medicine (heat exhaustion), salting out dyestuffs, supercooled solutions. •single crystals are used for spectroscopy, UV and infrared transmissions.	conrete mixes to give quicker initial set and greater strength •fireproofing fabrics; coagulant in rubber manufacturing •component of oil and gas well fluids	and ice melting, steel heat-treating and water softening •fertilizer component; chemical intermediate in the production of other potassium salts •Spectroscopy; salt substitute; lab reagent; food additive	(size, dressing and filling of cotton and woolen fabrics, thread lubricant, carbonization of wool), paper manufacture, road dust-laying compounds, floor-sweeping compounds, flocculating agent, catalyst.	

[a]For physical-chemical properties of liquid ($CaCl_2$•$2H_2O$, 37%) and brine (35% $CaCl_2$) $CaCl_2$ solutions, refer to Evans and Frick (2001).

2.2 Laboratory Detection Limits

Environment Canada's National Laboratory for Environmental Testing (NLET) analyzes anions, such as chloride, in water by ion chromatography using carbonate/bicarbonate as an eluent with a method detection limit of 0.02 mg/L (C. Cannon 2009, pers. com.). The Ontario Ministry of the Environment (MOE), Canadian Association for Laboratory Accreditation (CALA) accredited method (E3016) determines the concentration of chloride in drinking water, surface water, sewage and industrial waste by colourimetry. The current MOE laboratory minimum reporting value (w) for chloride in surface waters is 0.2 mg/L and the detection limit (t) is five times that, 1.0 mg/L (P. Wilson, 2009, pers.

com.). The MOE CALA accredited method for detection of water-extractable chloride in sediment and soils (E3013) uses ion chromatography, with a minimum reporting value (w) of 0.5 μg/g and a detection limit (t) of 2.5 μg/g (P. Wilson, 2011, pers.com.).

Other methods used for detection of chloride in water samples include the use of wet-chemistry methods (titrations), or by correlation with electrical conductivity measurements.

3.0 PRODUCTION AND USES

Chloride occurs in the natural environment as salts of sodium (NaCl), potassium (KCl), calcium ($CaCl_2$), and magnesium ($MgCl_2$) (Nagpal *et al*., 2003; WHO, 2003). The greatest quantities of chloride are distributed in the world's oceans. Chloride also constitutes approximately 0.05% of the earth's outermost crust (lithosphere) (NRC, 1977). Naturally-occurring saline lakes occur in Canada in the Prairies and British Columbia (Evans and Frick, 2001; Derry *et al.*, 2003; Hammer 1993; Last 1990). Underground salt deposits have been found throughout Canada, with bedded deposits (interspersed among rock layers) in southwestern Ontario, Saskatchewan and Alberta, and dome deposits (homogeneous formations) in Nova Scotia, New Brunswick, Ontario, Manitoba, Saskatchewan, and Alberta (Prud'Homme, 1986) (see Section 4.1 Natural Background). The Canadian salt industry produces approximately 12.5 million metric tones per year from domestic salt mines (Nagpal *et al*., 2003). The source of Canadian salt production includes major rock salt (halite) mines in Ontario, Quebec, and New Brunswick, and from vacuum pan refineries[1] in Alberta, Saskatchewan, Ontario, New Brunswick, and Nova Scotia (Dumont, 2008). Over three quarters of this mined salt is used primarily for road deicing purposes (Nagpal *et al*., 2003; Dumont, 2008).

Sodium chloride is used to produce industrial chemicals such as caustic soda, chlorine, soda ash, sodium chlorite, sodium bicarbonate, and sodium hypochlorite, which are utilized for various industrial applications such as pulp and paper, textiles, soaps and detergents, bleach manufacturing, petroleum products, and aluminum production (Health Canada, 1987). Sodium chloride is also the active ingredient in the pesticide product AdiosAmbros, used for the control of ragweed on roadsides, highways, walkways, vacant lots and other non-cropland sites (PMRA, 2006). Based on registrant data, it is estimated that the total use of AdiosAmbros (sodium chloride) per season in Canada is approximately 243 tonnes (PMRA, 2006). Magnesium chloride is utilized in the manufacturing of industrial products in addition to being a source of magnesium metal (Prud'Homme, 1986). Calcium chloride is used as a drying agent (FHWA, 1999), and potassium chloride is most commonly used for fertilizer production (NRC, 1977; FHWA, 1999). An estimated 45% of the salt consumed in Canada is used for snow and ice control on roads (Prud'Homme, 1986). Sodium chloride, calcium chloride, potassium chloride, and magnesium chloride are all used for road deicing, with NaCl being the most

[1] Vacuum pan refineries evaporate water from brine using steam-powered vapor recompression evaporators under a vacuum, reducing energy requirements. What remains is a crystallized salt slurry which is dewatered using a centrifuge, dried, potentially treated with any additives (e.g. potassium iodide or iodate making iodized salt) and packaged (Salt Institute, 2011).

widely used (Prud'Homme, 1986; Mayer *et al.*, 1999; Evans and Frick, 2001). It is estimated that 97% of road salt is in the form of NaCl, 2.9% in the form of $CaCl_2$, and 0.1% as $MgCl_2$ and KCl (Environment Canada, 2004). In the winter of 1997 to 1998, an estimated 4,750,000 tonnes of sodium chloride and 110,000 tonnes of calcium chloride were used for the deicing of Canadian roads (Environment Canada, 2001; Nagpal *et al.*, 2003). Another significant source of road salt is that what is applied as a result of private deicing operations, for example, applications onto sidewalks, driveways, and parking lots. Road salt contribution from this source is significant, and is not often quantified. A study of chloride mass balance in a City of Toronto urban watershed (Highland Creek) indicated that approximately 38% of the road salt applied in the watershed came from applications onto parking lots and driveways by private contractors (Perera *et al.*, 2009). It was also noted that the amount of salt applied by private contractors (non-regulated applications) could often be several times higher than the amount applied onto paved roads (regulated applications) (Perera *et al.*, 2009; Chapra *et al.*, 2009). For example, in the city of Madison (Wisconsin, USA), parking lots were estimated to have as much salt applied to them as was applied to public streets (Chapra *et al.*, 2009).

4.0 AQUATIC SOURCES AND FATE

4.1 Natural Sources

The chloride ion is naturally occurring, and therefore detection of increased levels of chloride in surface waters does not necessarily imply an anthropogenic source. Natural sources of chloride in aquatic systems include naturally-occurring saline lakes, groundwater discharges from saline aquifers, volcanic emanations, sea spray, and seawater intrusion in coastal areas (NRC, 1977). As well, wildfires and logging have a significant influence on the remobilization of major ions in lake watersheds impacted by these perturbations (Pinel-Alloul *et al.*, 2002).

Canada has many known naturally occurring salt deposits. Major salt deposits are found in Nova Scotia, New Brunswick, Quebec, Ontario, Manitoba, Saskatchewan and Alberta (Dumont, 2008; CANMET 1991) (Figure 4.1). Many of these salt deposits have been discovered while exploring for oil and gas and potash, due to similar geological conditions for these deposits (Dumont, 2008). Marine evaporate deposits are the most important sources of salt in Canada (11 to 12 x 10^6 tonnes per year) (Bell, 1996). Marine evaporate deposits are essentially comprised of pure halite (NaCl), and any or all of gypsum, anhydrite ($CaSO_4 \cdot 2H_2O$, $CaSO_4$), sulphur, sylvite (KCl) and various halides (Cl, Br, I) of Ca, K, Na, Mg and Sr (Bell, 1996). In Canada, marine evaporate deposits include the Salina Formation in Ontario (halite [NaCl] and gypsum [$CaSO_4 \cdot 2H_2O$]), the Windsor Group in the Appalachian region (halite [NaCl], sylvite [KCl], gypsum [$CaSO_4 \cdot 2H_2O$], celestite [$SrSO_4$]), in the Prairie Formation in Saskatchewan (sylvite [KCl], halite [NaCl], brine), at Gypsumville Manitoba (gypsum [$CaSO_4 \cdot 2H_2O$]) and at Windermere British Columbia (gypsum [$CaSO_4 \cdot 2H_2O$]) (Bell, 1996).

The largest salt bed deposit is found in western Canada, covering an area of 390,000 km^2, with an average thickness of 122m (Dumont, 2008). This salt bed extends from the Northwest Territories, down into Alberta, Saskatchewan and Manitoba (Dumont, 2008). Inland saline lakes where Cl is the dominant anion are relatively rare in Canada (most are dominated by SO_4 and CO_3 anions) (Derry *et al.*, 2003). This is in contrast to other parts of the world, where inland saline lakes are commonly Cl-dominated, such as in South Africa and Australia. These naturally Cl-dominated lakes in Canada are largely located on the boreal plain in the south-eastern part of the Northwest Territories as well as in the north-eastern part of Alberta. Chloride-dominated saline lakes are found in Wood Buffalo National Park (boreal mixed-wood forest interspersed with wetlands, prairies and salt flats) as well as south of the park sites near Fort McMurray (Derry *et al.*, 2003). The bedrock geology of this area with Cl-dominated saline lakes is characterized by Middle Devonian limestone ($CaCO_3$), gypsum ($CaSO_4 \bullet 2H_2O$), and dolostone shale ($CaMg(CO_3)_2$), covered with a thin layer of glacial, glacial-lacustrine, lacustrine and aeolian depostis (Derry *et al*., 2003). The source of high chloride in surface waters are deep groundwater springs that discharge NaCl salt from along the dissolution edge of the Cold Lake Formation of marine evaporitic halite (NaCl). Some Cl-dominated lakes within the vicinity of the Athabasca River tend to get periodically diluted (Derry *et al.*, 2003). The SO_4/CO_3 dominated surface waters in western Canada have a bedrock geology comprised of Upper Cretaceous deposits of sandstone, shale, coal and bentonite (Derry *et al*., 2003). Gypsum, mirabilite ($Na_2SO_4 \bullet 10H_2O$) and thenardite (Na_2SO_4) are SO_4-based minerals dominant in the prairies and aspen parklands (Derry *et al.*, 2003). Brine springs have been found in British Columbia (Dumont, 2008).

In Ontario (the second largest salt bed in Canada), salt deposits are located along the shores of Lake Huron and Lake Erie, and are part of what is known as the Michigan Basin. The Michigan basin contains some of the most highly saline brine found in any sedimentary basin (Wilson and Long, 1993). The brine contains high levels of Ca, Br and Cl, and relatively low concentrations of Mg, SO_4 and HCO_3 (Wilson and Long, 1993).

In the Atlantic provinces, major salt deposits have been found underlying Prince Edward Island, New Brunswick, Nova Scotia, part of Newfoundland and Labrador, as well as the gulf of the St. Lawrence River (all remains of ancient inland seas) (Dumont, 2008). For detailed information on salt production in Canada, see Dumont (2008).

Weathering of various rocks, such as shale and limestone, leaches chloride into the aquatic ecosystem (WHO, 2003). Chloride-containing salt compounds are highly soluble, hence they easily dissociate and tend to remain in their ionic forms once dissolved in water. It has been well known and frequently cited in many documents that the chloride ion is highly mobile, and concentrations in water are not affected by chemical reactions, hence it does not biodegrade, readily precipitate, volatilize, or bioaccumulate. As well, it has been often noted that chloride does not adsorb readily onto mineral surfaces and therefore concentrations remain high in surface water and sediment pore water, and low in sediment (Mayer *et al.,* 1999; Evans and Frick, 2001; WHO, 2003). Overall, inorganic chloride is generally considered to be a hydrologically and chemically inert substance and is often used as a tracer for pollution (e.g. increasing urbanization in a watershed).

Research by Oberg and Sanden (2005) has indicated that chloride participates in complex biogeochemical cycling whereby the terrestrial environment plays a key role. Large amounts of naturally produced organic chloride are present in soils. A large portion of inorganic chloride that is deposited in terrestrial environments is transformed to organic chloride in soil or vegetation. The majority tends to become chlorinated organic matter. The process is not fully understood, but it is believed that the transformation is driven by biotic processes, with abiotic processes also playing a role. Recent research by Oberg (2006) suggests that inorganic chloride present in surface water to a large extent originates from decomposing organic matter, which may be years, decades or thousands of years old. Oberg (2006) states that it is not yet well understood how the application of road salt influences the biogeochemical cycling, whether it enhances or diminishes the natural formation of chlorinated compounds.

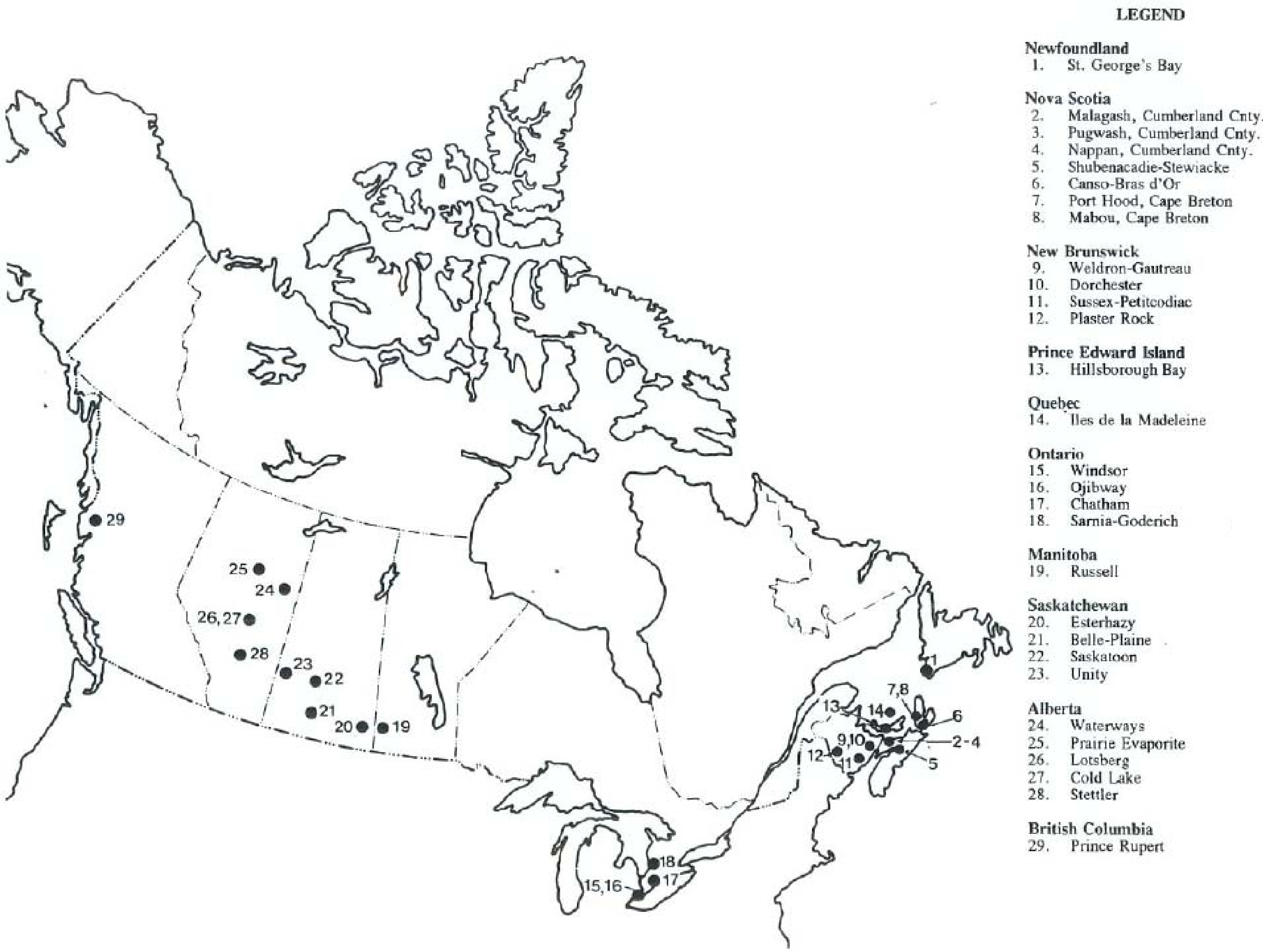

Figure 4.1 Principal salt deposits of Canada (CANMET 1991).

4.2 Anthropogenic Sources

Chloride compounds from anthropogenic sources enter the aquatic ecosystem through various pathways, such as stream inflow, road or overland runoff, groundwater inputs, and leaching from contaminated soils (Evans and Frick, 2001).

The application and storage of road salts for snow and ice control in the winter is a major non-industrial anthropogenic source of chloride to the aquatic environment, especially in densely populated regions such as southern Ontario and Quebec (Chapra *et al*., 2009). During the 1997 to 1998 winter, Morin and Perchanok (2000) approximated that 2,950,728 tonnes of chloride was released into the Canadian environment as a result of road salt (as NaCl) and dust suppressant (as $CaCl_2$) application. The provinces where the most chloride was used on roadways was Ontario (1,148,570 tonnes) and Quebec (950,444 tonnes), however Nova Scotia was found to have the highest loading per unit area of province (230,182 tonnes) (Morin and Perchanok, 2000) (Table 4.1).

Table 4.1 Total chloride loadings based on road salt (as NaCl) and dust suppressant (as $CaCl_2$) application. Loadings are based on total NaCl loadings during the 1997-98 winter season and the estimated use of $CaCl_2$ in a typical year (Morin and Perchanok, 2000).

Total Chloride Loadings	
Province / Territory	**Total Chloride Loading – Tonnes**
Yukon Territory	2,139
Northwest Territories	2,989
British Columbia	93,900
Alberta	114,641
Saskatchewan	33,642
Manitoba	46,880
Ontario	1,148,570
Quebec	950,444
New Brunswick	173,896
Nova Scotia	230,182
Prince Edward Island	18,061
Newfoundland and Labrador	135,384
Total Chloride	**2,950,728**
Ontario (MWWTP loading for 2008)[1]	*175,000*

[1]Estimated discharge of chloride in effluent from municipal wastewater treatment plants (MWWTPs) into Ontario waters in 2008 was 175,000 tonnes (M.Manoharan, Ontario Ministry of the Environment, 2009, pers.comm.).

In terms of total consumption of chloride-based dust suppressants in Canada, it has been estimated that in the year 2000, approximately 103 kt (on a 100% basis) was used. The majority of this consumption is calcium chloride, which is used across Canada (Environment Canada, 2005) (Table 4.2).

Table 4.2 Consumption of Chloride-Based Dust Suppressants in Canada, Year 2000 (kilotonnes - 100% basis) (Environment Canada, 2005).

Province / Territory	**Calcium Chloride**	**Magnesium Chloride**	**Total**
British Columbia	11	3	14
Alberta	6	<1	6
Saskatchewan	4	<1	4
Manitoba	3	2	5
Ontario	41	<1	41
Quebec	22	<1	22
New Brunswick	3	0	3
Nova Scotia	2	0	2
Prince Edward Island	1	0	1
Newfoundland and Labrador	1	0	1
Territories	4	0	4
Total	98	5	103

Elevated concentrations of chloride associated with deicing have been documented in groundwater, wetlands, streams, and ponds adjacent to snow dumps and salt-storage areas, and also those draining major roadways and urban areas in Canada (Evans and Frick, 2001; Nagpal *et al*., 2003). In the Northeastern United States, aquatic chloride is found to be at concentrations that are threatening to freshwater ecosystems, and are occurring as a result of deicing associated with increased coverage by roadways and urban development (Kaushal *et al*., 2005). In many semi-arid regions of the world, land clearing and over irrigation are causing increased salinization of freshwater (Hassell *et al*., 2006).

Another anthropogenic source of chloride includes municipal wastewater treatment plant (MWWTP) effluent. The estimated discharge of chloride into Ontario waters in 2008 was 175,000 tonnes (M.Manoharan, Ontario Ministry of the Environment, 2009, pers.comm.). In general, the concentration of chloride in untreated municipal wastewater effluent ranges from 20 to 160 mg/L (Chapra *et al.*, 2009). The addition of ferric chloride ($FeCl_3$) to enhance phosphorus removal was implemented by many MWWTPs in the Great Lakes basin, however the use of this chloride salt would only attribute to an approximate increase of 10 mg Cl^-/L in effluent discharges (Chapra *et al.*, 2009).

Canada's oilsands industry is an example of an industrial anthropogenic source of chloride to the environment (also a source of naphthenic acids which are likely of higher concern with respect to aquatic ecosystem impacts compared to chloride) (Allen, 2008).

Freshwater imported by oilsands mines is used to separate bitumen from sand and clay using hot water extraction. This process water (high in alkalinity with a pH of 8.0-8.4, slightly brackish with total dissolved solids ranging from 2,000 to 2,500 mg/L, acutely toxic to aquatic biota due to high levels of organic acids) cannot be released to the environment, and must be stored in tailings ponds. When mining ceases operation, process water and tailings are to be reintegrated into the landscape using a variety of land and aquatic system reclamation processes (Allen, 2008). Tailings pond waters are dominated by the following issolved solids: sodium (500 to 700 mg/L), bicarbonate, chloride (75 to 550 mg/L) and sulphate (200 to 300 mg/L) (Allen, 2008), and are more concentrated than in local surface waters, with chloride concentrations exceeding Athabasca river values by up to 90-fold (Allen, 2008) (Table 4.3).

Table 4.3 Chloride concentrations measured in oil sands process water, the Athabasca river and regional lakes (Allen, 2008).

Variable (mg/L)	Syncrude MLSB (2003)	Syncrude demonstration ponds (1997)	Suncor TPW (2000)	Suncor CT release water (1996-97)	Suncor CT Pond seepage (1996-97)	Athabasca River (2001)	Regional Lakes (2001)
Chloride	540	40-258	80	52	18	6	<1-2

Note: MLSB: Mildred Lake Settling Basin; TPW: tailings pond water; CT: consolidated tailings; data represent mean values from samples collected during the year indicated; ranges indicate mean values for multiple sites.

Additional chloride concentrations in various industrial oilsands operation wastewaters within the oil sands region were characterized (CEMA, 2003). The highest chloride concentrations were measured in basal water (saline aquifer water), consolidated or composite tailings release waters resulting from the management of mature fine tailings, as well as general tailings waters. Respective median chloride concentrations (and range) in these 3 wastewaters was 4,304 (95-21,603 mg/L), 360 (60-719 mg/L) and 388 (45-615 mg/L) (CEMA, 2003).

Canadian diamond mining is also a potential source of chloride to aquatic environments, and mines are found in the Northwest Territories, Nunavut and Ontario. Pit dewatering often occurs with the addition of saline groundwater (e.g. Victor Diamond mine in Ontario). The salinity of the groundwater is due to the natural geology of the area and may also be attributed to proximity to ocean water (e.g. Victor Diamond mine is in close proximity to James Bay). With respect to the Victor Diamond mine, the aquifer that is used for de-watering is an artifact of incursion of the Tyrell Sea some 8,000 years ago. The salinity is due to chlorides. Chloride concentrations will depend on the aquifer being de-watered (several different geological units). An average concentration of 1,100 mg/L was expected at the time of approval (2006). Chloride values as high as 6,600 mg/L were reported in some of the early and deep test holes (T. Kondrat, Surface Water Specialist, Northern Region, Ontario Ministry of the Environment, pers.comm.).

Other anthropogenic sources of chloride include industrial effluent. Chloride (from inorganic salts) is not tracked by Environment Canada's National Pollutant Release Inventory. Ontario does track chloride releases from a small number of sectors (e.g. electric power generation, industrial minerals, inorganic chemical, metal mining) covered under MISA (Municipal/Industrial Strategy for Abatement) regulations, but not from other industries (Chapra *et al* 2009). For the reporting years 1996 – 2010, 2 companies in 2 different sectors (industrial minerals and inorganic chemicals) reported average chloride daily loadings under MISA, on a monthly basis. The reported loadings are presented in Tables 4.4 and 4.5.

Table 4.4 Daily chloride loadings reported under MISA for one company in the industrial minerals sector. Loadings are reported as monthly averages of daily loading (kg/d).

Average daily loading (kg/d) averaged monthly												
Reporting Year & Month	**Jan**	**Feb**	**Mar**	**Apr**	**May**	**Jun**	**Jul**	**Aug**	**Sep**	**Oct**	**Nov**	**Dec**
1997	76	78	94	na	80	87	76	82	70	71	66	57
1998	71	76	75	76	68	80	89	68	74	96	79	72
1999	80	80	74	85	73	69	69	93	119	95	90	43
2000	88	77	81	73	71	68	82	73	58	74	70	50
2001	82	68	109	76	68	47	64	66	93	45	86	95
2002	87	71	83	69	64	74	67	64	68	54	70	81
2003	88	75	117	65	65	62	61	63	65	76	42	62
2004	62	65	71	59	38	54	69	62	54	56	54	70
2005	35	54	49	54	56	70	60	72	73	57	48	87
2006	70	63	56	58	53	60	61	42	74	55	55	68
2007	78	72	50	59	68	66	59	51	57	60	84	54
2008	81	58	70	61	75	64	68	54	66	67	61	54
2009	72	37	53	61	66	48	72	71	98	69	73	84
2010	81	107	75	105	91	76	93	96	92	66	77	73

na = Data not available.

Anthropogenic sources of chloride also include water softners, domestic sewage, refuse leachates, and irrigation drainage (Schneider, 1970; Little, 1971; Pettyjohn, 1971; 1972; NAQUADAT, 1985; CEMA 2003). The use of conditioning salts (water softening agents) and their subsequent release into septic and municipal wastewater treatment systems has also been shown to be a source of chloride to the aquatic environment (Kuntz and McBride, 1993). Other anthropogenic sources can also include mines, facilities producing deionized waters (especially reverse-osmosis operations, including water production facilities, ethanol plants, other foods and beverage producers), and food processors such as pickling facilities (Bill Dimond, Michigan Department of Environmental Quality, 2009, pers.comm.).

Table 4.5 Daily chloride loadings reported under MISA for one company in the inorganic chemicals sector. Loadings are reported as monthly averages of daily loading (kg/d).

Average daily loading (kg/d) averaged monthly												
Reporting Year & Month	**Jan**	**Feb**	**Mar**	**Apr**	**May**	**Jun**	**Jul**	**Aug**	**Sep**	**Oct**	**Nov**	**Dec**
1996	28478	7793	10668	10992	9250	9771	4225	4546	5495	6743	8894	6755
1997	na	na	na	7864	5293	5988	7286	7584	15536	11247	7353	14199
1998	6680	8081	9335	5292	9942	9332	9009	8761	6492	na	na	na
1999	9174	18685	11793	15415	12867	19286	6077	10024	15142	10070	10362	11184
2000	13898	13992	16015	7188	12028	5943	7120	6391	7057	6527	6582	7840
2001	8511	7564	8182	13750	10428	13651	6709	12133	6180	24122	23370	37930
2002	19084	31809	21604	9986	17498	15931	10320	8430	5957	10814	7282	6246
2003	8421	7221	8332	12715	5897	4784	6182	4209	7457	17399	5565	17452
2004	7234	7266	7767	10224	8134	4432	10884	4354	6878	26433	7588	6255
2005	7831	7763	6691	24556	5261	5810	4800	16687	10685	7187	4187	3368
2006	9283	7291	9309	na	na	na	na	na	na	na	na	na
2007	na	na	na	na	na	na	na	na	na	na	na	na
2008	na	na	na	na	na	na	na	na	na	na	na	na
2009	656	1358	431	1510	215	176	380	159	353	215	237	591
2010	447	2006	477	314	293	222	209	193	210	211	188	130

na = Data not available.

Evaporation and dilution are thought to be the main processes that cause the change of chloride concentrations in water (Mayer *et al.*, 1999). Once in the aquatic environment, chloride ions are easily transported, and those in groundwater can be expected to reach surface water, thus potentially affecting aquatic ecosystems (Nagpal *et al.*, 2003). The retention time of chloride tends to be longer in lakes and wetlands, hence the release of small quantities annually may result in increased levels over the course of several years. Evaporation affects wetlands more so than lakes, and can lead to increased salinity. Streams and other flowing water systems have short retention times, varying based on the physical properties of the system. However, continual releases from road salt leaching or groundwater inputs can increase retention time and chloride impacts (Evans and Frick, 2001).

4.3 Impacts of Increased Salt on Baseflow and Meromixis

The cumulative use of road salts every winter season is resulting in an increase in the concentration of chloride in urban rivers and streams during baseflow conditions (Perera *et al.*, 2009). Studies have shown that 35 to 50% of applied road salt is removed annually from catchments via overland flow, and ends up in rivers, creeks or lakes. The remainder accumulates in soil and shallow groundwater, and eventually makes its way to deep groundwater until steady state concentrations are attained (Howard and Haynes 1993; Ruth 2003). This chloride in groundwater then gets released to urban creeks and streams over the course of the year, resulting in increased chloride concentrations during baseflow conditions. A chloride mass balance conducted for an urban Lake Ontario watershed lagoon (Frenchman's Bay) crossed by a large transport route (Hwy 401) in south central Ontario (City of Pickering) indicated the total chloride delivery to be 3,700 tonnes annually, with up to 48% of the total load delivered by baseflow (Meriano *et al*., 2009). Increased chloride in baseflow has also been well documented in a study conducted by Perera *et al*., (2009). Continuous hourly monitoring of water quality in an urban creek (Highland Creek in Toronto) was collected. The data indicated that chloride concentrations in Highland Creek measured immediately following the winter period (which spans from November to March) during low-flow (dry weather conditions) were high, reflecting the direct impact of road salt application and spring snow melt. The baseflow chloride concentrations measured at this time, between April and May (2005 to 2008), ranged from approximately 310 to 590 mg Cl^-/L. The baseflow chloride concentrations then continued to decrease over the summer months, with lowest levels measured between September and October (2005 to 2008), ranging from 220 to 320 mg Cl^-/L. The start of the winter season (November to March) resulted in another cycle of increased chloride levels. The high chloride levels measured during the non-winter period were attributed to the chloride stored in soils and groundwater, which continuously released to surface waters (Perera *et al.*, 2009). The long-term baseflow chloride concentration for Highland Creek (Morningside subwatershed) was estimated to be 275 mg Cl^-/L (Perera *et al*., 2009), compared to the concentration of 150 mg/L reported in 1972 (Meriano *et al*., 2009). The current chloride baseflow of 275 mg/L exceeds the CCME CWQG of 118 mg Cl^-/L as well as both the British Columbia (150 mg Cl^-/L) and US EPA (230 mg Cl^-/L) chloride long-term guidelines for the protection of aquatic life. The subsurface storage of road salts in soils and groundwater has been shown to have long-term impacts on surface water quality. Monitoring of a rural stream in southeastern New York state showed an annual increase of 1.5 mg Cl^-/L per year over a 19-year period since 1986, even though road salt use ceased to increase over this same time period (Meriano *et al*., 2009).

Not only does the use of road salt impact baseflow chloride concentrations, it has also been shown to be a factor impacting the vertical mixing of surface waters by way of changing the density gradient in lakes. This phenomenon is referred to as meromixis, with the formation of meromictic lakes. A meromictic lake has layers of water that do not intermix. This differs from lakes that are holomictic (physical mixing occurs between deep and surface water layers at least once a year), monomictic (mixing occurs once a year), dimictic (mixing occurs twice a year, usually in spring and fall), and polymictic (mixing occurs several times a year). In the case of meromictic lakes, one of the outcomes of this stable stratification of deep and surface water layers is that the deep layer (monimolimnion) can become quite depleted of oxygen. The concentration of dissolved oxygen in the monimolimnion of a meromictic lake has less than 1 mg/L, while the surface layer (mixolimnion) may have concentrations of 10 mg/L or higher (Lampert *et al*., 1997). Very few organisms can survive in the low oxygen environment of the bottom lake layer. Variables known to play a role in lake stability, the capacity to resist vertical mixing, include:

- lake morphometry (ratio surface:volume, where lakes with a deep steep-sided basin and small surface area resist mixing)
- water residence time

- watershed topography and wind fetch
- watershed area and the ratio drainage area: lake area, as well as
- chemical (saline) input, leading to a chemical-based density stratification between lower and upper water layers (Wetzel, 2001).

Meromixis can also occur naturally in coastal lakes receiving saline intrusions and in water bodies supplied with subsurface saline springs (Wetzel, 2001). Road salts that enter into lakes through surface flow (overland runoff, ditches, streams) or groundwater discharge (seeps, springs) have the potential to induce meromixis. However, current practice salt management levels have greatly reduced the potential for meromixis (M. Satin, Canadian Salt Institute, pers.comm.).

The examples of meromictic lakes provided in the sections to follow will be focusing on examples of lakes impacted by road salt intrusion. There are several examples of the formation of meromictic conditions associated with road salt applications available in the scientific literature for lakes located within Ontario (Free & Mulamoottil, 1983; Smol *et al*., 1983), New York state (Bubeck *et al*., 1971; Bubeck *et al*., 1995) and Michigan state (Judd, 1970). Strong evidence related to the influence of increased salt loadings on meromixis has been provided in the literature. There are several studies that document the re-establishment of dimictic conditions in lakes following a decrease in salt use on roads located within their watersheds.

One example is First Sister Lake in Ann Arbor Michigan (Judd 1970) (Table 4.6). The lake characteristics include a surface area of 0.013 km^2, a catchment area of 0.348 km^2, a volume of 12,600 km^2, and a maximum depth of 7.2 m. Road salt application increased from April 1966 (3,300 tons/winter) to April 1967 (8,250 tons/winter), where the lake went from experiencing overturn (1966) to meromixis (1967). Chloride concentrations measured in surface (82.8 mg/L) and bottom (85.3 mg/L) layers in 1966 were fairly equal, whereas surface chloride concentrations in 1967 were half (64.5 mg/L) of what was measured in the bottom layer (129 mg/L). The difference in the densities between surface and bottom waters was found to be attributable to both the salinity and thermal gradients. Judd (1970) explains the reason the surface concentration of chloride decreased from 82.8 mg/L to 64.5 mg/L in one year. A spring melt occurred, whereby the water flushing out from the storm sewer outfall (which serviced a subdivision surrounding First Sister Lake) actually had a lower density (1.00008 gcm^3) when compared to the density of the lake surface water (1.00013 gcm^3), due to large amounts of melting snow diluting the salty run-off water. This water inflowing into the lake moved over the lake water of higher density, forming a new dilute surface layer on the lake.

Ides Cove in Rochester New York is another similar example (Bubeck *et al*., 1995) (Table 4.6). This lake has a surface area of 0.0118 km^2 and a maximum depth of 8.8 m. When road salt use was high (1970-1972), and there was no occurrence of lake overturn, the difference between surface and bottom layer chloride concentrations ranged from 80-160 mg/L. When a reduction in road salt use was implemented (1980-1982), and overturn was re-established, the difference between surface and bottom chloride concentrations ranged from 0-90 mg/L. As with First Sister Lake in New York, the difference in density between surface and bottom waters was found to be attributable to both the salinity and thermal gradients.

Table 4.6 Studies documenting lake meromixis associated with road salt applications.

Period before overturn	Salt appl Tons/ winter	Lake overturn	[Cl] surface (mg L^{-1})[a]	[Cl] bottom (mg L^{-1})[a]	Δ density surface-bottom (10^{-4} g cm^{-3}) [b]	Comment
			Ides Cove, Rochester NY (Bubeck *et al*., 1995)			
spring 70-72	---	N	Δ 80-160 between surface and bottom		1.5-2.7 (S) -0.6 - -0.2 (T°)	Surface area: 0.0118 km^2; max depth: 8.8 m. Decrease in road salt use after 1974 associated with re-establishment of dimictic condition in the early 1980s.
spring 80-82	---	Y	Δ 0-90 between surface and bottom		0-1.6 (S) -0.4-0 (T°)	
			Irondequoit Bay, Rochester NY (Bubeck *et al*., 1971)			
March 70	77 000	N	160	400	3.6 (S) 0 (T°)	Surface area: 6.7 km^2; catchment area: 435 km$^{2;}$ Volume: 46 x 10^6 m$^{3;}$ max depth: 23 m
			First Sister Lake, Ann Arbor, Michigan (Judd 1970)			
April 66 [c]	3 300	Y	82.8	85.3	0.4 (S) 1.56 (T°)	Surface area: 0.013 km^2; catchment area: 0.348 km^2; volume: 12 600 m^3; max depth: 7.2 m
April 67	8 250	N	64.5	129	1.0 (S) 4.6 (T°)	Lake experienced intermittent meromixis during the years 1965-1967.

[a] Chloride concentrations in surface and bottom waters.
[b] Differences between densities of surface and bottom waters; a part is attributable to the salinity gradient (S) and the other to the thermal gradient (T°).
[c] Values obtained following spring overturn; surface waters were warming up contributing to the strengthening of the thermal density gradient.

Irondequoit Bay in Rochester New York experienced no overturn in March 1970 when road salt application was 77,000 tons/winter (Bubeck *et al*., 1971) (Table 4.6). Characteristics of this Bay include a surface area of 6.7 km^2, a catchment area of 435 km^2, a volume of 46x10^6 m^3, and a maximum depth of 7.2m. Surface and bottom chloride concentrations were measured to be 160 and 400 mg/L, respectively.

Threshold conditions below which meromixis does not occur are suggested by studies providing both evidence of saline inputs but absence of meromixis (Tables 4.6 and 4.7). Lake Carré is a small lake in the Laurentians in the province of Québec, which is surrounded by roads and cottages. Even though this lake is impacted by road salt inputs, monitoring data (oxygen levels and conductivity with depth) indicate that the lake undergoes complete circulation in the spring and fall (Table 4.6). The way in which to determine if a lake is in fact impacted by road salt inputs is to calculate the molar ratio of Na:Cl and Na:Ca (Legendre *et al*., 1980). If the molar ratio of Na:Cl is nearly equal to 1, and the molar ratio of Na:Ca is greater than 0.55, then this is evidence that there are salt inputs into a lake (Legendre *et al*., 1980). Table 4.8 provides the concentrations of major cations and anions measured in Lake Carré, as well as in 12 lakes in the surrounding area which are not impacted by road salt inputs (St-Cyr, 2000).

Table 4.7 Depth profiles of oxygen concentrations and conductivity for Lake Carré, a small lake in the Laurentians, Québec.*

Depth (m)	$[O_2]$ (mg/L) - 2000		Conductivity	Comment
	30 April	26 August	(µS/cm)	
0.5	12.6	10.6	225	Surface area: <<1 km^2 Max. depth: 8.8 m Lake completely surrounded by roads and cottages.
1	12.8	10.6	225	
2	13.2	10.7	220	
3	14.1	10.5	225	
4	13.0	14.6	240	
5	12.3	10.3	320	
6	11.2	0.3	355	
7	---	0.2	---	
8	---	0.2	---	

* This lake is impacted by road salt inputs from roads surrounding it. These data constitute evidence that the lake undergoes complete circulation in spring and fall (data not shown for fall; source: St-Cyr, 2000).

Table 4.8 Concentrations of major cations and anions in Lake Carré, and in lakes in the surrounding area not known to be impacted by salt inputs (source: St-Cyr, 2000). *

	Na	K	Mg	Ca (mg/L)	Cl	NO_3	SO_4
L. Carré	23.13	1.16	3.07	16.30	40.12	ND	4.13
Lakes (N=12) in the surroundings	4.08	0.30	1.01	4.90	7.13	0.04	1.63

***** A molar ratio Na:Cl nearly equal to 1 (1.0056: 1.1316), and a molar ratio Na:Ca larger than 0.55 (Legendre *et al*., 1980) constitute evidence that there are saline inputs (NaCl) in Lake Carré.

5.0 AMBIENT CONCENTRATIONS IN CANADIAN WATERS, SEDIMENT AND SOIL

As was completed by Mayer *et al*., (1999) and EC/HC (2001b), an overview of background concentrations in Canadian surface waters will be presented based on geographic region. In general, the concentration of chloride in Canadian inland waters is low, but is dependant on location (e.g. proximity to urbanized areas, areas naturally high in salts, or influence of ocean water). It was stated in EC/HC (2001b) that overall, surface waters most susceptible to the influence of road salt loadings are lentic (standing) waters which include small urban lakes and ponds with long residence times, as well as wetlands. Also impacted are small streams in developed/urban areas.

5.1 Lakes and Rivers of the Atlantic Region (Newfoundland and Labrador, Nova Scotia, New Brunswick and Prince Edward Island)

Both Mayer *et al*., (1999) and EC/HC (2001b) provide water quality monitoring data compiled by Jeffries (1997) collected from lakes located in largely unimpacted areas. Data were collected from 38 lakes in Labrador, 63 lakes in Newfoundland and Labrador, 150 lakes in Nova Scotia, and 166 lakes in New Brunswick. The median chloride concentration in these lakes ranged from 0.3 to 4.5 mg/L (collected 1985 and later). Overall, Atlantic region chloride ion values were found to be highest in coastal regions, although there were areas where high chloride concentrations were linked to winter use of de-icing agents as well as to bedrock lithology (Clair *et al.*, 2007). Sites in and around Goose Bay in Labrador have been shown to have elevated chloride levels due to road salt (Clair *et al.*, 2007). High chloride levels measured in central Nova Scotia, where a major highway exists, are likely a result of de-icing (Clair *et al.*, 2007). Marine evaporite beds located between Saint John and Moncton in southern New Brunswick and to the east of the Annapolis Valley in Nova Scotia is likely responsible for elevated Cl^- in these areas (Clair *et al.*, 2007).

Several lakes and rivers were also monitored for chloride concentrations at Kejimkujik National Park in Nova Scotia, which were found to be influenced by sea salt. Chloride concentrations were found to range from 3.6 to 5.4 mg/L (Kerekes *et al*., 1989; Kerekes and Freedman 1989; Freedman *et al*., 1989; In: EC/HC 2001b). Monitoring of rain and measurable dry deposition between 1985 and 2000 at the Canadian Air Precipitation Monitoring (CAPMoN) site, located in Kejimkujik National Park 60 km from the Bay of Fundy and 80 km from the Atlantic Ocean, reported an average deposition of 15 kg Cl^-/ha/yr (Clair *et al*., 2007). However, Yanni *et al* (200a In: Clair *et al.*, 2007) reported that more than double the amount of Cl^- measured by CAPMoN was being exported from local catchments, indicating that the true influence of sea salt is likely greater than that measured by CAPMoN (e.g. fog containing Cl^- intercepted by forest canopies was not being measured by conventional atmospheric precipitation sampling equipment).

Studies on lakes in developed watersheds were found to have elevated chloride concentrations compared to lakes located in rural areas (EC/HC 2001b). Chloride concentrations in urban-located lakes were found to vary on a temporal basis as well (both seasonally and over time). For example, a small (82.7 ha) shallow (mean depth 3.9m, maximum depth 12.2m) lake in Nova Scotia, Chocolate Lake, was found to have highly elevated concentrations of chloride between April and August 1975 (where the estimated salt load during the winter of 1974-75 was 35,409 kg of chloride) (Kelly *et al*., 1976 In: EC/HC 2001b). The average summer chloride concentration was 207.5 mg/L, which is well above the background chloride concentration of 15-20 mg/L in non-impacted lakes. Chloride concentrations in Chocolate Lake have also been shown to vary with depth. This salt gradient has prevented complete vertical mixing (meromixis) resulting in deep anoxic waters in the summer. Measured concentrations of chloride at the surface ranged from 199 to 224 mg/L, at a depth of 6.1m the concentration ranged from 189 to 217 mg/L, and at a depth of 12.1m, the range was 225 to 330 mg/L. Spring (April) surface water samples were collected in the middle of 51 lakes in the Halifax/Dartmouth metropolitan area (Keizer *et al*., 2007). Of these 51 lakes, 49 were sampled in 1980, 48 were sampled in 1991 and all 51 were sampled in 2000 (Keizer *et al*., 2007). The mean chloride concentration measured in all lakes in 1980 (34.9 mg/L) was almost half that measured in 1991 (61.2 mg/L). There was a moderate increase in the mean chloride concentration measured between 1991 (61.2 mg/L) and 2000 (64.9 mg/L). The majority of lakes sampled in both 1991 and 2000 had average chloride concentrations greater than those measured in 1980. A few lakes with the highest chloride measurements in 1991 did provide lower measurements in 2000, depicting a potential improvement. In 1980 the maximum measured chloride in the 49 monitored lakes was 125 mg/L (Frog Pond). 9 of the 49 lakes had chloride concentrations >50 mg/L, and 3 lakes had concentrations >100 mg/L. In 1991, the maximum measured chloride in the 49 monitored lakes was 197 mg/L (Whimsical), with 10 lakes measuring >50 mg/L, 8 lakes measuring >100 mg/L, and 4 lakes measuring >150 mg/L (EC/HC 2001b). In 2000, the maximum measured chloride in the 51 monitored lakes was 160 mg/L (Whimsical), with 13 lakes measuring >50 mg/L, 12 lakes measuring >100 mg/L, and 1 lake measuring >150 mg/L (Keizer *et al*., 2007). It is important to point out that surface water samples were collected in the middle of each lake, therefore chloride concentrations at the sediment water interface may indeed be higher. In contrast, 19 rural lakes sampled by Keizer *et al*., (1993 In: EC/HC 2001b) in 1980 and 1991 had measured chloride below 20 mg/L.

Mean chloride concentrations measured in 25 river/stream sites in Nova Scotia for the 2006-2008 sampling period ranged from 3.4 to 99.4 mg/L (D. Parent, Environment Canada, pers. comm.). For sites with a drainage area population density less than 10 people per square kilometre, mean chloride concentrations (2006-2008) ranged from 3.4 to 47.1 mg/L with higher concentrations typically related to underlying evaporite deposits (e.g. salt and gypsum) present in the watershed. For sites with a drainage area population density greater than 10 people per square kilometre, chloride concentrations (2006-2008) ranged from 24.4 to 99.4 mg/L with higher concentrations typically related to road salt application in urban areas. Recent chloride trend analysis (e.g. last 10-15 years) is not available for the Nova Scotia monitoring network as the routine monitoring was only re-instated in 2006 after several years of dormancy (D. Parent, Environment Canada, pers. comm.).

In general, for the Atlantic region, chloride concentrations of <10 mg/L are normally observed in inland lakes, with concentrations as high as 20 mg/L in lakes located closer to coastal areas. Sea water salt concentrations are approximately 35,000 mg/L of which approximately 55% is chloride, which equates to 19,250 mg chloride/L. Salt spray would influence lakes located close to coastal areas.

5.2 Lakes and Rivers of the Central Region (Ontario and Quebec)

In the case of Ontario's creeks and rivers, the Provincial Water Quality Monitoring Network (PWQMN) database contains chloride (mg/L) readings taken from 2,164 stations across Ontario from 1964 to 2005 (MOE, 2005). The data are summarized for the following years: 1964 to 1975; 1976 to 1985; 1986 to 1995; and, 1996 to 2005. Each data set contains between 23,000 and 75,000 chloride measurements. Descriptive statistics are included in Table 5.1. The number of chloride measurements in each time period is as follows: 1964 to 1975 = 59,869; 1976 to 1985 = 74,611; 1986 to 1995 = 58,510; 1996 to 2005 = 23,732.

The 25th, 50th (median), and 75th percentiles ranged from 6.0 to 25.1 mg/L, 13.5 to 25.8 mg/L, and 32 to 59 mg/L, respectively. The means for each data set are between 31 and 52 mg/L. In the past 40 years, there were over 300 chloride measurements exceeding 1,000 mg/L (Table 5.2). The number of chloride measurements greater than 100 but less than 1,000 mg/L, greater than 10 but less than 100 mg/L, and less than 10 mg/L for each period are listed in Table 5.2.

There was an increase in the number of chloride measurements greater than 1,000 mg/L from 1964 to 1975 compared to 1986 to 1995, followed by a decrease in the 1996 to 2005 dataset (this assessment was based on percentage of samples exceeding 1,000 mg chloride/L since number of chloride measurements differed). There was a slight decrease in the number of stations that measured greater than100 but less than 1,000 mg chloride/L from the 1964 to 1975 dataset compared to the 1976 to 1985 dataset, followed by an increase up to the 1996 to 2005 dataset. As well, there was an increase in the number of stations with chloride measurements between 10 and 100 mg/L in the 1964 to 1974 data set compared to the 1990 to 2005 data set (Table 5.2). The majority of the stations that exceeded 1,000 mg chloride/L were located in southwestern Ontario (Montgomery Creek, Sheridan Creek, Black Creek) and around the Greater Toronto Area (Humber River, Don River, Etobicoke Creek, Mimico Creek) (Table 5.2). Eastern Ontario had relatively few stations measuring greater than 1,000 mg chloride/L, almost half of which were in Montgomery Creek. Compared to the other regions, Northern Ontario has the least number of stations exceeding 1,000 mg chloride/L.

Table 5.1 Summary of chloride[1] (mg/L) monitoring data from Ontario creek and river stations from 1964 to 2005.

Statistics	Year			
	1964-1975	**1976-1985**	**1986-1995**	**1996-2005**
n (number of chloride measurements)	59,869	74,611	58,510	23,732
mean	31	34	46	52
SD	79	91	130	100
median (50th percentile)	13.5	15.6	21.7	25.8
25th percentile	25.1	6.0	8.8	12
75th percentile	32	34	49	59
min	<0.2	<0.2	<0.2	<0.2
max	8,700	4,800	14,200	5,080
n >1,000 mg/L	43 (0.07%)[2]	107 (0.14%)	134 (0.23%)	29 (0.12%)
n between >100-1,000 mg/L	3,794 (6.3%)	4,195 (5.6%)	5,160 (8.8%)	3,082 (13%)
n between 10-100 mg/L	31,844 (53%)	42,518 (57%)	36,653 (63%)	15,848 (67%)
n<10 mg/L	24,188 (40%)	27,791 (37%)	16,563 (28%)	4,773 (20%)

[1]The Ontario Ministry of the Environment minimum reporting value for chloride in surface waters is 0.2 mg/L, and the method detection limit is five times that at 1.0 mg/L (see section on Laboratory Detection Limits).
[2]Percentage of total chloride measurements.

The Provincial Water Quality Monitoring Network of the Ontario Ministry of the Environment recently provided surface water chloride data for four representative watersheds found within the province (MOE, 2009). These include the Skootamotta River near Actinolite (undeveloped, Canadian shield, sparse road network), the Sydenham River near Owen Sound (agricultural, rural residential), Fletcher’s Creek at Brampton (rapidly urbanizing watershed, dense road network), and Sheridan Creek (fully developed, urban residential/industrial, dense road network). In the case of the Skootamotta River, monthly samples collected from pre-1980 to 2007 had a measured median chloride concentration of 2 mg/L, with a minimum of 0.5 and a maximum of 36 mg/L. The Sydenham River measured median chloride concentration was 10 mg/L, with a minimum of 3 and a maximum of 330 mg/L. In the case of the more developed watersheds, Fletcher’s Creek had a measured median chloride concentration of 131 mg/L, with a minimum and maximum of 13.5 and 4,150 mg/L, respectively. The minimum concentration was detected in the fall (September 1984), whereas the maximum concentration was detected in the winter (February 2007). Sheridan Cree had the highest measured chloride, with a median value of 292 mg/L, and min and max values of 14.5 and 5,320 mg/L, respectively. As with Fletcher’s Creek, the lowest chloride measurement in Sheridan Creek was detected in the fall (October 1980) and the highest

Table 5.2 Summary of Ontario creek and river sampling station locations in excess of 1,000 mg/L chloride from 1964 to 2005. (Number in brackets denotes the number of sampling stations along a creek or river that exceed 1,000 mg/L).

1964-1975	1976-1985	1986-1995	1996-2005
Northern Ontario			
Creek (near Stanrock) (1)	Lake Nipissing (3)	Fort Creek (1)	
Garden River (1)	Garden River (1)		
Eastern Ontario			
Little Cataraqui Creek (1)	Bear Brook (2)	Little Cataraqui Creek (2)	Butlers Creek (1)
Rideau River (1)	Pringle Creek (1)	Montgomery Creek[2] (9)	
Wilmot Creek (1)		Oshawa Creek (1)	
Ditch (Scotch River) (1)		Pringle Creek (1)	
Riviere de la Petite Nat. (1)		Drainage Canal, Holland Marsh (1)	
Southwestern Ontario			
Bear Creek (2)	Bear Creek (1)	Alder Creek (1)	Dingman Creek (1)
Sunfish Creek (5)	Black Creek (22)	Bear Creek (2)	Fletchers Creek (3)
Thames River (3)	Boyne River (2)	Belle River (1)	Sheridan Creek (8)
Twenty Mile Creek (7)	Centre Creek (2)	Big Creek (11)	
Larder Lake (1)	Big Creek (1)	Black Creek (13)	
Talfourd Creek (1)	Montgomery Creek[3] (24)	Boyle Drain Ditch, Milverton (1)	
Manning Drain (1)	Schneider Creek (3)	Canard River (1)	
Black Creek (1)	Six Mile Creek (2)	Credit River (1)	
	Fletchers Creek (1)	Fletchers Creek (3)	
	Twenty Mile Creek (3)	Fourteen Mile Creek (1)	
		Saugeen River (1)	
		Sheridan Creek (7)	
		Turkey Creek (1)	
Greater Toronto Area (GTA)			
Don River (4)	Don River (5)	Don River (31)	Don River (9)
Humber River Tributary (2)	Don River West (1)	Etobicoke Creek (14)	Etobicoke Creek (2)
Mimico Creek (4)	Etobicoke Creek (8)	Highland Creek (6)	Humber River (3)
Montgomery Creek[3] (2)	Etobicoke Creek	Mimico Creek (11)	Mimico Creek (2)

[2]There are two creeks named "Montgomery Creek" that have been monitored as part of the PWQMN (since 1964). One station at Harbour Road in Oshawa was monitored between 1966 and 2007. Results from this station are responsible for the Eastern Ontario and GTA entries in the Table 5.2. The other station at Vanier Drive in Kitchener was monitored between 1976 and 1978. Results from this station are responsible for the Southwestern Ontario entry in the table.

1964-1975	1976-1985	1986-1995	1996-2005
	West (1)		
Etobicoke Creek (1)	Highland Creek (6)	Don River West (1)	
German Mills Creek (1)	Mimico Creek (7)	Etobicoke Creek West (1)	
Don River (1)	Sheridan Creek (11)	Farewell Creek (2)	
		Humber River (9)	

measurement was detected in the winter (February 2007). For both Fletcher's Creek and Sheridan Creek, the majority of samples exceeding 1,000 mg/L were collected in the winter months (January, February, March), as well as one sample collected in late fall (November) at Fletcher's Creek. These maximum measured k concentrations are most likely associated with increased application of road salt or they may be associated with a thawing period resulting in runoff. Spikes in surface water concentrations of chloride in other water bodies have been measured in late summer which has been associated with decreased water levels due to evaporation (Russell and Collins, 2009). Another cause of increased chloride in surface water can also be related to chloride-contaminated-groundwater discharges into surface water. A study conducted by Meriano *et al*., (2009) assessed the fate of road salt applied in a densely urbanized watershed in the city of Pickering on the north shore of Lake Ontario (Frenchman's Bay). This watershed is traversed by Hwy 401, a 12-lane transport route. It was determined that 50% of the road salt applied in this watershed enters Frenchman's Bay lagoon via overland flow, while the remaining 50% enters the subsurface as aquifer recharge and enters Frenchman's Bay via chloride-contaminated groundwater. Surface water quality is continuously degraded year-round due to influx of salt from both surface-runoff (during winter road salt applications) and groundwater (where groundwater concentrations have been measured to exceed 1,600 mg/L). Chloride concentrations throughout Frenchman's Bay watershed continuously exceed Ontario's Drinking Water Aesthetic Objective for chloride of 250 mg/L. As well, several studies have indicated that even when road salt application decreases, surface water monitoring of chloride concentrations does not show an associated decrease in chloride concentration (Meriano *et al*., 2009; Kilgour *et al*., 2009). This is attributed to subsurface storage of chloride and the lag effect of chloride entering surface water systems. In any case, sensitive species located in surface waters within rapidly urbanizing watersheds or fully developed watersheds are at risk of being adversely impacted by chloride (from the application of road salt). Other factors that may be contributing to elevated chloride concentrations in Ontario surface waters include road density, presence of salt stockpiles and the locations of snow removal deposits.

Mayer *et al*., (1998) measured chloride in highway runoff along three roadways of varying density (2-lane and 4-lane highways) in Burlington, Ontario from 1997 to 1998. Chloride concentrations ranged from 45 to 10,960 mg/L, with the greatest concentrations recorded along the 4-lane highway with the greatest automobile density (Skyway Bridge). Foster and Maun (1978) documented chloride concentrations in snow near London, Ontario, ranging from 133 to 4,128 mg/L at the pavement edge. Lower levels were found 8 m from the highway edge, ranging from 9 to 79 mg/L. In urban snow dumps in the Ottawa-Carleton region, Droste and Johnson (1993) measured chloride concentrations between 454 to 1,018 mg/L. Stormwater ponds accumulate road salt *via* inputs of contaminated snowmelt runoff. Mayer *et al*., (1996) reported chloride concentrations ranging from 22 to 1,201 mg/L in several stormwater ponds in residential and industrial areas of Toronto.

The influence of road salt sources is supported by information summarized in Evans and Frick (2001). Evans and Frick (2001) documented chloride concentrations in various aquatic sources (streams, rivers, ponds, lakes, snow dumps, highway runoff, stormwater ponds) in Ontario which were directly attributed to road salt application. Generally these were in excess of background aquatic concentrations, which range from 10 to 25 mg/L of chloride (Evans and Frick, 2001).

The highest chloride concentrations in streams and rivers are found in those running through densely populated areas that utilized great quantities of road salt. Between 1990 and 1996, four creeks in the Toronto watershed located in highly developed areas near roads or highways were monitored. Observations from these locations (Etobicoke Creek, n=152; Mimico Creek, n=37; Black Creek, n=38; and Highland Creek, n=55) yielded maximum chloride concentrations ranging from 1,390 to 4,310 mg/L, and mean chloride concentrations ranging from 278 to 553 mg/L (MOE, 1999). The highest concentrations occurred during the winter months (December to March). Scott (1980a) measured chloride concentrations in Black Creek located in Metropolitan Toronto in 1974 and 1975 when an estimated 580 tonnes of chloride entered the creek as a result of road salt application. The highest chloride concentrations (250 mg/L), documented in the winter and early spring, were generally reduced with increased water flow in the spring, and were between 50 to 100 mg/L in the summer. In a creek passing through Waterloo, Ontario (Laurel Creek), Crowther and Hynes (1977) reported peak chloride concentrations of 680 mg/L in 1974 and 1,770 mg/L in 1975 at sampling locations near a major road. Williams *et al*., (1999) attributed elevated chloride concentrations in 20 Ontario springs (8.1 to 1,149 mg/L) to groundwater contaminated by road salt. Real-time monitoring of a Lake Ontario tributary (Cooksville Creek) in a highly urbanized watershed (Mississauga, Ontario) showed chloride levels exceeding that of seawater, with measurements made in February 2011 reporting chloride as high as 20,000 mg/L (K. vander Linden, Credit Valley Conservation Authority, pers. comm.).

In the Don River located in Metropolitan Toronto, minimum, maximum and mean chloride concentrations sampled from three roadside locations in the city (n = 543 total) ranged from 1 to 290 mg/L; 960 to 2,610 mg/L; and, 158 to 287 mg/L, respectively, between 1990 and 1996. The greatest concentrations were measured in winter and early spring (Scott, 1980a). At three urban roadside sampling locations beside the Humber River northwest of Toronto (n = 491), minimum, maximum and mean chloride concentrations ranged from 0.2 to 31 mg/L; 96 to 1,680 mg/L; and, 46 to 1,775 mg/L, respectively between 1990 and 1996. The Rouge River had lower chloride levels than the Don and Humber Rivers between 1990 to 1996, with concentrations ranging from 27 to 650 mg/L (mean 81 mg/L) (MOE, 1999). Seasonal fluxes of chloride in the Rideau River, increasing from 9 up to 57 mg/L in winter after ice storms, have been attributed to road salt (Oliver *et al*., 1974). Slight seasonal increases of chloride in the Niagara River in 1975 (11.2 to 12.5 mg/L February; *versus* 10.1 to 10.7 mg/L August to May) were also attributed to road salt (Chan and Clignett, 1978).

Small lakes and ponds are not as strongly impacted as rivers and streams by road salts (Evans and Frick, 2001). Within the Humber River watershed, Scanton (1999) reported chloride concentrations at eight small lakes in 1995 that ranged between 10.6 mg/L (Lake St. George) and 408.9 mg/L (Grenedier Pond). Watson (2000) documented lower chloride concentrations in ponds located in Southern Ontario near 2-lane roads as compared with 6-lane roads, reporting mean concentrations of 95 and 952 mg/L, and maximum concentrations of 368 and 3,950 mg/L, respectively. In 1973, the mean chloride concentration of 109 lakes in the Experimental Lakes Area in northwestern Ontario was 0.8 mg/L (Beamish *et al*., 1977).

Meromictic conditions in lakes (a lack in vertical mixing that results in anoxic conditions at depth) can be caused by road salts. Little Round Lake in Ontario is an example of this, and elevated chloride concentrations at the bottom layer of the lake (monimolimnion) were measured at 104 mg/L (Smol *et al.*, 1983). Another meromictic lake located in Mississauga (Lake Wabekayne) showed increases in chloride concentrations in winter (282 mg/L) compared to summer (50 mg/L) at the monimolimnion in 1972, which were attributed to road salt (Free and Mulamoottil, 1983). The anoxic conditions associated with lake meromixis, caused by large increases in chloride from road salt runoff, cause various metals to be more readily released from sediments (Wetzel, 1983). Wang *et al*., (1991) documented an enhanced release of mercury from sediments caused by elevated chloride levels, and MacLeod *et al*., (1996) found that chloride enhanced mercury mobilization from soils. Conversely, Smot *et al*., (1983) reported a benefit to the formation of meromictic conditions in Little Round Lake. Originally oligotrophic, the lake became eutrophic due to increased settlement within the watershed. With increased settlement came the construction of roads, which in turn introduced road salt loading into the lake. The lake returned to original oligotrophic conditions due to the lack of vertical mixing, whereby nutrients stored in the bottom lake layers were prevented from mixing with surface waters. Another noted benefit of road salt loading comes from a study conducted by Celis et al. (2009), whereby it has been shown that increased loading of sodium chloride to some Sudbury-area lakes is actually reducing metal toxicity for planktonic organisms

Larger lakes are not as strongly impacted by road salts as smaller lakes and ponds. Chloride inputs come from many sources in addition to road salts, such as sewage and industrial wastes (Evans and Frick, 2001). Chloride levels were monitored in Lake Simcoe and associated tributaries (Winter *et al.*, 2011). Lake Simcoe is the largest inland lake in southern Ontario, excluding the Laurentian Great Lakes. Surrounding land use is predominantly agricultural, however urban development is rapidly increasing. The lake also receives treated effluent from 15 municipal sewage treatment plants. Currently, only 12 % of the Lake Simcoe watershed drains urban land and roads, yet evidence of road salt application can already be seen. Current concentrations of chloride measured in the lake are between 36 to 40 mg/L, which is a greater than three-fold exceedance of chloride measured at the lake's outflow in 1971 (Winter *et al.*, 2011). Meassurements made in 8 Lake Simcoe tributaries indicated that chloride concentrations increased significantly from 1993 to 2007, with highest levels detected in rivers draining the greatest percentage of urban land and roads. Examples include Lovers Creek and East Holland River, where the respective annual rate of increase in chloride concentration was 5.2 and 10.4 mg Cl^-/L/yr (based on measurements taken from 1993-2007) (Winter *et al.*, 2011). One river in close proximity to a major highway (North Schomberg River) displayed an annual rate of increase in chloride concentration of 3.9 mg Cl^-/L/yr (Winter *et al*., 2011). The cumulative chloride load estimated at the mouths of 7 rivers flowing into Lake Simcoe ranged from 11,563 to 32,107 tonnes/yr from 1998 to 2007, and increased significantly over this period (Winter *et al*., 2011).

The Laurentian Great Lakes are even less impacted compared to larger inland lakes, although monitoring does show an increasing trend in chloride levels. Fraser (1981) estimated that road salt contributed 1 and 1.5 million metric tonnes per year of chloride into Lake Ontario and Lake Erie, respectively, accounting for 20% of the total chloride loading into the lakes. In the 1960s, chloride concentrations in Lake Ontario and Lake Erie were in exceedance of 25 mg/L. Over time concentrations in Lake Erie have declined (20 mg/L in 1990) while those in Lake Ontario have not, and this is attributed to the longer retention time of water in Lake Ontario. Chloride concentrations in Lake Superior and Lake Huron in the 1960s were estimated at 1 and 7 mg/L, respectively (Moll *et al.*, 1992). Higher concentrations (20 mg/L) present in the Great Lakes and the St-Lawrence River are attributed to industrial activity (NRC, 1977). A recent publication by Chapra *et al*., (2009) employs chloride surveillance data from the Great Lakes over the past 150 years in order to identify trends in chloride concentrations. Estimates of pre-settlement (background runoff and atmospheric input) chloride concentrations in the Great Lakes were 0.93 mg/L for Lake Superior, 1.58 mg/L for Lake Huron, 1.75 mg/L for Lake Erie, and 1.87 mg/L for Lake Ontario. Chloride levels measured in 2006 were 1.4 mg/L for Lake Superior, 6.6 mg/L for Lake Huron, 18.4 mg/L for Lake Erie and 22.3 mg/L for Lake Ontario. Results of the study indicated that (with the exception of Lake Superior) chloride concentrations have been increasing over the past 100 years, with loadings peaking from 1965 to 1975. Implementation of industrial load reductions resulted in a decrease in chloride levels in the 1980s, but recent trends indicate that chloride levels are increasing again. Possible reasons for the increase are that 1) despite load reductions, the lake systems are not at steady state, or 2) loadings from non-industrial sources (e.g. road salt, municipal discharges) are increasing, with potential introduction of new industrial inputs. Chloride is not tracked by Environment Canada's National Pollutant Release Inventory and Ontario tracks chloride releases from only a small number of inorganic chemical plants, but not all releases (Chapra *et al*., 2009).

The Drinking Water Surveillance Program (DWSP) database reports open lake raw water data from intake locations into numerous Ontario lakes (Table 5.3) (MOE, 2002). The greatest chloride concentrations were from the Lake Ontario intakes (South Peel, Kingston, R.L. Clark, Grimsby, and Cobourg), with mean, maximum, and minimum concentrations ranging from 20.8 to 25.7 mg/L; 26 to 58.5 mg/L; and, 1.2 to 19.4 mg/L, respectively. The Brockville intake at the St. Lawrence River had a mean concentration of 21.2 mg/L, and the Dunville intake at Lake Erie had a mean concentration of 20.2 mg/L, while the other two intakes at Lake Erie (Union, Elgin) had mean concentrations that were lower, at 12.8 and 15.8 mg/L, respectively. The mean chloride concentrations at the intakes located at the Detroit River, Niagara River, and Bay of Quinte were between 11.6 and 14.2 mg/L, and the intakes located at Lake Huron, Lake St. Clair and Lake Superior were the lowest, with concentrations <10 mg/L.

Table 5.3 Ontario's Drinking Water Surveillance Program open lake chloride monitoring data 1996 to 2006 (mg/L).

	L. Huron		**St. Clair R.**	**Detroit R.**	**L. Erie**		
Intake	Goderich	Grand Bend	Lambton	Amherstburg	Union	Elgin	Dunnville
n	486	249	519	522	456	521	496
mean	8.9	7.1	7.3	11.6	12.8	15.8	20.2
SD	2.5	0.7	2.0	4.0	2.6	1.7	4.6
min	6.0	6.0	2.2	7.2	7.5	11	15.6
max	29.6	11.4	25.6	38.2	22.4	38.1	47
	L.Ontario					**Niagara R.**	**St. Lawrence R.**
Intake	South Peel	Kingston	R.L. Clark	Grimsby	Cobourg	Rosehill	Brockville
n	517	518	499	515	516	517	467
mean	25.7	20.8	22.4	23.2	21.9	17.0	21.2
SD	4.96	1.36	2.54	2.16	0.95	1.40	1.24
min	11.8	1.2	7.2	19.4	14.8	14.6	7.2
max	58.5	26	42.6	41.2	27.6	34.2	35.8
	Bay of Quinte	**L. Superior**					
Intake	Belleville	Terrace Bay	Bare Point				
n	528	494	521				
mean	14.2	1.9	1.7				
SD	3.4	0.3	0.9				
min	7.2	0.8	0.6				
max	30.7	3.8	21.4				

Data collected by the Ministère du Développement durable, de l'Environnement et des Parcs (MDDEP) in Quebec surface waters from 1979 to 2004 indicate that the median and 98th percentile chloride concentrations are 5 and 66 mg/L, respectively. The minimum measured value was at the detection limit of 0.1 mg/L (collected in 1979), and the maximum measured value was 1,650 mg/L (from a sample collected in June 1997) (M. Bérubé, MDDEP 2009, pers.comm.). Other data presented for Quebec was investigated by the Ministère des Transports du Quebec (1980,1999) for Lac à la Truite, near Sainte-Agathe-des-Monts (EC/HC 2001b). The drainage area for this lake is affected by a 7 km stretch of highway, where at some places it is located as close as 250 m from the highway. The average chloride concentration measured in 1972 was 12 mg/L. A maximum chloride concentration of 150 mg/L was measured in 1979. Road salts were replaced with abrasives, and chloride average concentrations have decreased to 45 mg/L, as measured in 1990.

St-Cyr (2000) determined major ion concentrations in Lake Carré, which is impacted by road salt applications in its immediate watershed; results are provided in Table 4.5. Chloride concentrations in this lake were 40.12 mg/L. The mean chloride concentration of 12 lakes in the immediate vicinity of Lake Carré, but which are not impacted by road salt, was determined to be 7.13 mg/L (St-Cyr, 2000).

Pinel-Alloul *et al.,* (2002) measured major ion concentrations in 17 undisturbed lakes of the boreal Canadian Shield of central Québec (Haute-Mauricie). Chloride concentrations varied between 0.1 and 0.2 mg Cl^-/L from 1996 to 1998.

Fortin *et al.*, (2010) determined trace metal and major ion concentrations in littoral waters of 16 lakes of the Rouyn-Noranda mining area in Abitibi-Témiscamingue. Water samples were obtained by in situ dialysis during summers of 1998 and 1999, and major anions were measured by ion chromatography. Chloride concentrations ranged from 0.2 to 15.7 mg/L (Table 5.4), and were not related to contamination by metal mining as 3 lakes impacted by acid mine drainage (Turcotte, Dufault and Dasserat) did not have especially higher chloride levels than those of the other lakes of this study. The high chloride concentration of Lake Renaud is likely caused by road salt application as two sides of this lake are in close proximity of a major national highway.

Table 5.4 Chloride concentrations in lakes of the Rouyn-Noranda mining area in Abitibi. Mean and standard deviations are calculated from values obtained at 3 occasions during summers 1998-1999 and from variable numbers of sampling sites per lake.

Lake name	Number of sampling stations	Cl^- concentration (mg/L)
Vaudray	3	0.20±0.02
Caron	3	2.4±0.53
Bousquet	4	1.2±0.21
Turcotte	1	0.21±0.00
Bouzan	1	1.1
Moore	1	3.8±3.6
Joannès	3	0.27±0.02
Héva	1	1.01±0.04
Dufay	3	0.17±0.02
Renaud	1	15.7±1.3
Évain	2	4.6±2.6
Despériers	1	0.19
Ollier	2	5.3±3.7
Opasatica	4	3.06±0.09
Dufault	4	4.8±0.11
Dasserat	2 to 4	0.22±0.12

Overall, unimpacted lakes on the Canadian Shield have measured chloride concentrations of <1 to 7 mg/L, with higher concentrations (10 to 30 mg/L) measured in the lower Great Lakes and the St. Lawrence River. Chloride concentrations above background are commonly detected in densely populated areas (e.g. small urban watersheds) where road densities are high.

5.3 Lakes and Rivers of the Prairie Region (Manitoba, Saskatchewan and Alberta)

In addition to freshwater lakes, numerous naturally saline lakes are located in the Prairie Region, with Saskatchewan having the greatest number, followed by Alberta, and then Manitoba with the least. Unlike chloride-rich saline lakes in many other parts of the world (e.g. Australia, western United States, South Africa), these saline lakes are predominantly SO_4^{2-} ($NaSO_4$ or $Mg/NaSO_4$) and CO_3^{2-} ($CaCO_3$, $MgCO_3$ or $CaMg(CO_3)_2$) salt dominated, and make up over 95% of the total lakes (Last and Ginn, 2005). These lakes display a wide range in ionic composition and concentration, and range in salinities from relatively dilute water (less than 0.1 g/L total dissolved solids) to greater than seawater (nearly 400 g/L total dissolved solids) (Last and Ginn, 2005). Table 5.5 provides the mean ionic composition of saline and hypersaline lakes located in the northern Great Plains, indicating a strong predominance of Na and SO_4 in these lakes.

Locations of these saline lakes are largely localized to the southern portion of the provinces (the saline lake region) although these lakes have been found to occur as far north as Edmonton in Alberta (EC/HC 2001b). The southern portion of the provinces is underlain by Cretaceous bedrock, composed mainly of shales, silts and sandstones (Hammer 1994). Examples of meromictic lakes (resulting from naturally high saline conditions) known to exist in the Prairie Region include Waldsea Lake and Deadmoose Lake (discovered in the early 1970s) as well as Arthur Lake, Marie Lake and Sayer lake (discovered in 1985) (Hammer 1994). Other examples include Freefight Lake, Basin Lake, and Middle Lake (J.M. Davies, Saskatchewan Watershed Authority, pers.comm.). Lake Winnipegosis in Manitoba was studied by McKillop *et al*., (1992 In: EC/HC 2001b). Twenty-three naturally saline (sodium chloride dominated sites) were sampled along the western shore of the lake where chloride concentrations were found to range from 861 to 33,750 mg/L. Low chloride concentrations (<5 mg/L) are reported in lakes located in the northern portions of Saskatchewan and Alberta, outside of the Interior Plains Region. Some of the saline lakes in the Prairie region undergo significant changes in water levels, and in turn, ion concentrations and ratios, on a seasonal basis. One example, Ceylon Lake which is located in southern Saskatchewan, displays a wide range in total dissolved solids on an annual basis, ranging from 30,000 g/L to 300,000 g/L (Last and Ginn, 2005). In the spring, Ceylon Lake is dominated by (Mg)-SO_4-HCO_3, whereas in the fall, the lake is dominated by Mg-(Na)-Cl-SO_4 (Last and Ginn, 2005).

The biological species composition in these saline lakes is found to be comparable between lakes of low salinity and freshwater lakes. However, it is evident that as salinity increases, species diversity does decline. Lakes exhibiting extremely high salinity contain a very low diversity of species, and are dominated by halotolerant (saline tolerant) organisms (Herbst 2001).

Table 5.5 Mean ionic composition and total dissolved solids (TDS) of saline and hypersaline lakes located in Canada's northern Great Plains (Last and Ginn, 2005).

Geographic Area	Ca		Mg		Na		K	
	mmol/L	mg/L	mmol/L	mg/L	mmol/L	mg/L	mmol/L	mg/L
Eastern Prairies	4	160	24	583	4	92	1	39
Central Saskatchewan	19	761	149	3,621	193	4,437	5	195
SW Saskatchewan / SE Alberta	12	481	93	2,260	1,088	25,012	4	156
West-central Saskatchewan and east-central Alberta	3	120	144	3,500	1,362	31,311	10	391

Geographic Area	HCO_3		CO_3		Cl		SO_4		TDS
	mmol/L	mg/L	mmol/L	mg/L	mmol/L	mg/L	mmol/L	mg/L	g/L
Eastern Prairies	6	3,661	1	60	2	71	24	2,305	3
Central Saskatchewan	7	427	3	180	54	1,914	251	24,111	22
SW Saskatchewan / SE Alberta	96	5,858	36	2,160	29	1,028	1073	103,073	80
West-central Saskatchewan and east-central Alberta	268	16,352	44	2,640	107	3,793	1125	108,069	102

In northern Alberta, water quality monitoring data was collected from 1976-2000 by Alberta Environment from monitoring stations located upstream of Fort McMurray, and further north near Old Fort (CEMA, 2003) for assessment of ambient water quality. Chloride levels measured upstream of Fort McMurray ranged from non-detect to a maximum of 19 mg/L (median of 2.9 mg/L). Measurements taken at the Old Fort monitoring station showed chloride ranging from 1.2 to 65 mg/L (median of 17.9 mg/L), where these higher levels were attributed to the influence of the Clearwater River and other tributaries (CEMA, 2003).

5.4 Lakes and Rivers of the Pacific Region (British Columbia)

The Pacific region is also an area of naturally occurring saline lakes, and these are typically small and shallow (Topping and Scudder 1977 In: EC/HC 2001b). Chloride concentrations have been reported to range from 5.1 to 800 mg/L (Northcote and Halsey 1969 In: EC/HC 2001b). Six lakes located in a non-saline region in the southwest of British Columbia had a reported median chloride concentration of 2.5 mg/L (Phippen *et al*., 1996; Jeffries 1997; In: EC/HC 2001b).

The Serpentine River in the Lower Fraser Valley of British Columbia was monitored for conductivity every 15 minutes to assess the impacts of road salts on the receiving water. Conductivity was shown to increase 3-fold over 10- to 20-hour periods during times of thaw following a cold period when roads were salted (Whitfield and Wade, 1992 In: EC/HC 2001b). Aquatic organisms living in streams during winter months have the potential to be impacted by these fluctuations in salt concentrations (EC/HC 2001b).

Overall, the chloride concentration in unimpacted water bodies is <5 mgL; however, several lakes in the southern interior plateau of British Columbia had measured chloride concentrations >100 mg/L.

5.5 Lakes and Rivers of the Yukon, Northwest Territories and Nunavut

Surface water chloride monitoring data was available from nine monitoring stations in the Yukon. The data was retrieved from Environment Canada's Pacific and Yukon Water Quality Monitoring & Surveillance Program website (Environment Canada, 2009). Dissolved chloride measurements were found to be low, ranging from 0.1 to 4.6 mg/L.

No chloride monitoring data were presented for the Northwest Territories or Nunavut in the Priority Substances List Assessment Report for Road Salts (EC/HC 2001b).

5.6 Chloride in Benthic Sediments

Chloride salts are highly soluble and do not have a binding affinity for sediments (EC/HC 2001b). Mayer *et al.*, (1999) studied an urban pond and observed that chloride concentrations in sediment pore water were in equilibrium with overlying water. High chloride concentrations in sediment pore water can lead to osmotic stress in aquatic receptors. A high chloride concentration can also augment the concentration of dissolved metal by forming metal-chloride complexes, for example, with cadmium (EC/HC 2001b).

5.7 Chloride in Soil Near Salt Sources

Scott (1980b) documented chloride concentrations in soils close to major roads in Metropolitan Toronto in 1974 and 1975, within the watersheds of Black Creek and the Don River. Chloride concentrations in surface soils near Black Creek taken 0.5 m and 1 m from the pavement ranged from approximately 100 to 2,300 mg/kg (ppm) and 50 to 1,400 mg/kg (ppm), respectively. Generally, chloride concentrations were elevated in samples up to 15 m from the pavement/road. At 45 m from the road, the average chloride concentration was 8.7 mg/kg (ppm). A sample of sand taken from a paved median strip of Highway 7 contained 10,800 mg/kg (ppm). Samples were taken up to 60 cm from the surface, and showed evidence of chloride leaching to this depth, as concentrations were similar or greater than those taken at the surface in the same location (Scott, 1980b). Foster and Maun (1987) documented concentrations in soil up to 8 m from the highway edge in London, Ontario ranging from 110 to 380 mg/kg.

In the case of road salt application, high levels of sodium accumulated in soils can have serious implications for soil structure. Clay particles are negatively charged and have a tendency to bind calcium cations. The calcium binds closely to the surface of the clay particles, leading to a neutralization of the negative charge of the clay particles and allowing formation of soil aggregates. When sodium cations increase in concentration, the sodium displaces calcium and in turn the sodium cations bind to the clay particles. Hydrated sodium ions are larger than the calcium ions, and do not bind as closely to the clay particles as does calcium. The negative charge of the clay particles is not neutralized, and they in fact repel one another resulting in dispersion. Soils that are dispersed have impeded drainage, resulting in puddling and erosion (Bright and Addison, 2002).

5.8 Summary

Chloride concentrations in Canadian inland waters are generally low, but higher concentrations have been measured in highly urbanized areas, areas naturally high in salts (e.g. prairie saline lakes), and areas in close proximity to the influence of ocean water. Chloride does not have a strong binding affinity to sediment, and therefore chloride concentrations remain low in sediment, but high in sediment pore water. Chloride concentrations in soil near salt sources can become elevated, and chloride ions can easily mobilize to groundwater, which can ultimately lead to the discharge of high levels of chloride into surface waters.

6.0 TOXICITY OF CHLORIDE TO AQUATIC LIFE

6.1 Influence of Various Chloride Salts on Toxicity

The toxicity of chloride salts to aquatic life, on a chloride basis, can differ substantially depending on the cation present. Chloride toxicity tests have been conducted through the addition of chloride salts such as sodium chloride, calcium chloride, magnesium chloride and potassium chloride and the interactions of these different cations with chloride have been shown to affect toxicity. Results of tests with potassium and magnesium chloride suggest toxic effects observed are due to the potassium and magnesium cation, rather than the chloride anion. Conversely, it has been observed that the effects of calcium chloride and sodium chloride are likely due to the chloride anion. Generally speaking, the approximate order of chloride salt toxicity to freshwater organisms is $KCl > MgCl_2 > CaCl_2 > NaCl$ (Mount *et al.,* 1997) (Table 6.1). Based on these observations, chloride toxicity to freshwater organisms was only evaluated using tests with $CaCl_2$ and NaCl. As well, sources of $CaCl_2$ (e.g. dust suppressants) and NaCl (e.g. road salt) are one of the most significant anthropogenic non-industrial sources of chloride to the aquatic environment, specifically in densely populated regions of Canada (Evans and Frick, 2001; Chapra *et al*., 2009).

Table 6.1 Relative toxicity of potassium, magnesium, calcium and sodium chloride salts to freshwater organisms, assessed on a chloride ion basis.

Organism	Duration (h)	Endpoint	$[Cl^-]$ (mg Cl^-/L)				Reference
			K^+ Salt	Mg^{2+} Salt	Ca^{2+} Salt	Na^+ Salt	
Pimephales promelas (fathead minnow)	96	LC50	419	1,579	2,958	3,876	Mount *et al*., 1997
Daphnia magna (water flea)	48	LC50	314	990	1,770	2,893	Mount *et al*., 1997
Ceriodaphnia dubia (water flea)	48	LC50	300	655	1,169	1,189	Mount *et al*., 1997

Khangarot (1991) tested the toxicity of Ca^{2+}, Na^{2+}, and K^+ as chloride salts to the tubificid worm (*Tubifex tubifex)*, and the effect concentrations varied among the different compounds. Based on the toxicity of these cations in combination with chloride, 96 hour EC_{50}s for immobilization for Cl^- were 498, 1,204, and 737 mg/L, respectively. In the diatom (*Nitzschia linearis*), effect concentrations were observed at varying concentrations of chloride depending on the cation combination. A 50% reduction in the number of cells over a 120 hour exposure period was observed at 637 mg Cl^-/L (1,338 mg KCl/L), 1,474 mg Cl^-/L (2,430 mg NaCl/L), and 2,000 mg Cl^-/L (3,130 mg $CaCl_2$/L) (Patrick *et al*., 1968). From a comprehensive review of literature on chloride toxicity to aquatic organisms, Evans and Frick (2001) concluded that the most toxic salt is KCl, followed by $MgCl_2$, $CaCl_2$, and NaCl. Waller *et al*., *(*1996) demonstrated this trend in yellow perch (*Perca flavescens*) in a series of 24 hour exposures at 17ºC, where 2,500 mg/L KCl (1,189 mg Cl^-/L) caused 80% mortality, 10,000 mg/L $CaCl_2$ (6,389 mg Cl^-/L) caused 83% mortality, and 10,000 mg/L NaCl (6,066 mg Cl^-/L) resulted in 0% mortality. Jones *et al*., (1940; 1941) demonstrated that survival of the flatworm *Polycelis nigra* was most sensitive to KCl (1,259 mg/L or 599 mg Cl^-/L), followed by $MgCl_2$ (3,798 mg/L or 2,828 mg Cl^-/L), and NaCl (11,109 mg/L or 6,739 mg Cl^-/L) after a 48 hour exposure between 15ºC and 18ºC.

6.2 Mode of Action

Freshwater organisms are generally hyperosmotic, meaning they contain a higher internal concentration of salts compared to the surrounding water (Holland et al., 2010).

Increasing chloride in surface waters results in increased salinity, thereby affecting the ability of organisms to effectively osmoregulate, which could in turn affect endocrine balance, oxygen consumption following chronic exposures, and overall changes in physiological processes (Holland et al., 2010). In both invertebrates and fish, the main site of osmoregulation is the gill, which is also the site of active uptake of lost solutes. The sodium pump (Na^{+}+K^{+}-ATPase) is the main mechanism for moving ions across gills in aquatic animals. The mechanism of osmoregulation used is dependent on the life stage of the organism, for example pre-larval fish osmoregulate largely through the skin, whereas larval stages regulate through the gills (Varsamos and Charmantier, 2005). Insects possess a network of Malpighian tubules lined with secretory cells extending throughout much of the body cavity, which is involved in the reabsorption of ions (Dettner and Peters, 1999). In the case of spotted salamander (*Ambystoma maculatum*) egg clutches, disruption in osmoregulation has not been determined but is likely related to chemical changes in the egg capsule (perivitelline) membrane, as has been documented in egg clutches exposed to highly acidic conditions (Karraker and Gibbs, 2011). As with exposure to acid, high chloride may result in making the egg capsule membrane more rigid, reducing permeability, and therefore impacting the ability for water uptake (Karraker and Gibbs, 2011).

6.3 Short-Term (Acute) Toxicity

Criteria used for classifying available toxicity data as either primary, secondary, or unacceptable are described in the Protocol for the Derivation of Water Quality Guidelines for the Protection of Aquatic Life (CCME 2007). In general, primary toxicity studies involve acceptable test procedures, conditions, and controls, measured toxicant concentrations, and flow-through or renewal exposure conditions. Secondary toxicity studies usually involve unmeasured toxicant concentrations, static bioassay conditions and unsatisfactory reporting of experimental data. Unacceptable data are deemed not suitable for guideline development (e.g. no reporting of controls, test temperature too high to be relevant to Canadian surface waters, test organism not representative of a temperate species, etc.).

Short-term (acute) toxicity studies generally involved test durations of 96 hours or less for vertebrates and invertebrates. The following information provides a general overview of the toxicity data points used for short-term benchmark concentration derivation. The full suite of chloride toxicity data obtained from the scientific literature is presented at the end of this report in Appendix I.

6.3.1 Vertebrates

With respect to the fish studies selected for inclusion in guideline development, all were 96 hour LC50 endpoints. Overall, fish species were found to be quite tolerant of high chloride exposures for short (acute) durations. The fish exhibiting the greatest sensitivity was the fathead minnow, *Pimephales promelas*, with a 96h LC50 of 3,386 mg Cl^{-}/L (Mount *et al.*, 1997). The most chloride tolerant fish species was the threespine stickleback *Gasterosteus aculeatus* with a 96h LC50 of 10,200 mg Cl^-/L (Garibay and Hall 2004). Short-term chloride toxicity data for a total of 35 fish species was collected and is presented in Appendix I. Of these 35 species, studies for only 6 species were deemed acceptable for inclusion in the dataset for short-term guideline derivation. The short-term dataset for chloride does include data for the the rainbow trout (*Oncorhynchus mykiss*), a fish species that is considered to be euryhaline (able to adapt to a range of salinities). The rainbow trout was found to be one of the most chloride tolerant species in the short-term dataset with a 96h LC50 of 8,634 mg Cl^-/L (Elphick *et al*., 2010; Vosyliene *et al*., 2006). The rainbow trout is considered to be euryhaline because its life cycle involves migration between freshwater and saltwater environments. The physiology of euryhaline fish differs from that of stenohaline fish, which can only survive within a narrow range of salinities. One of the first responses in euryhaline fish such as rainbow trout, once exposure to saltwater is initiated, is increased drinking thought to be initiated by osmoreceptors in the oral region of the fish, with an associated decrease in urine ouput and decreased plasma water content (Bath and Eddy, 1979). After some time, drinking is reduced and salt excreting mechanisms are stimulated (increase in ionic effluxes) (Bath and Eddy, 1979). Eventually, the concentrations of ions (Na+ and Cl-) in plasma is reduced to the levels observed in freshwater exposures, with an increased concentration of ions (Na+ and Cl-) in muscle cells. Other physiological changes include increases in the number of mitochondrion rich cells with an associated increase in levels of Na+-K+ ATPase in the gills (Bath and Eddy, 1979). Sea water salt concentrations are approximately 35,000 mg/L of which approximately 55% is chloride, which equates to 19,250 mg chloride/L. Therefore, the rainbow trout should be able to tolerate at least the concentration of chloride in seawater, arguably under ideal exposure scenarios of gradually increasing salinities (the juvenile life stage was used in the derivation of the 96h LC50). However, the effect concentrations presented in this dataset fall below the chloride concentrations measured in seaweater. In the case of the rainbow trout, Vosyliene *et al*., (2006) observed that exposure to chloride induced significant changes in the morphological and haematological parameters studied, suggesting that sudden short-term exposures to increased chloride (e.g. similar to what would occur following a spike in chloride concentration in surface waters following a spring melt) may not provide the time necessary for physiological adaptation. However, if a field environment, fish are able to avoid areas of high salinity.

Short-term (acute) exposures were also obtained from the scientific literature for 11 amphibian species (see Appendix I), 9 of which were included in the short-term SSD dataset. Early life stages of amphibians were found to be generally more sensitive to acute chloride exposures, when compared to fish. The most sensitive species was the spotted salamander *Ambystoma maculatum*, with a 96h LC50 of 1,178 mg Cl^-/L (Collins and Russell 2009). The next two most sensitive species were the chorus frog *Pseudacris triseriata feriarum* and the wood frog *Lithibates sylvatica* (previously *Rana sylvatica*), with respective 96h LC50 concentrations of 2,320 mg Cl^-/L (Garibay and Hall 2004) and 2,716 mg Cl^-/L (Collins and Russell 2009; Sanzo and Hecnar 2006; Jackman 2010). The bullfrog, *Rana catesbeiana*, was the most tolerant, with a 96h LC50 of 5,846 mg $Cl^{-/}$L (Environ 2009). All 9 of these species have aquatic larval stages, and adults use wetlands, ephemeral sites and lakes for breeding, foraging and hibernation. As was discussed earlier, these types of water bodies are readily influenced by road salt application, and therefore it is important to include toxicity data for amphibians when available. Early amphibian physiological studies have provided indication that most amphibians cannot tolerate long-term exposures to 30% seawater (which would be equivalent to 5,775 mg Cl^-/L) due to osmotic dehydration and diffusional uptake of salt (Sanzo and Hecnar 2006). Discussions were had with experts with respect to whether or not acute toxicity test endpoints for 5 amphibian species from Collins and Russell (2009) could be used for the derivation of the short-term benchmark concentration. The issue or concern was that Collins and Russell (2009) removed “distressed” animals from exposure containers prior to end of test exposure. The removal of these “distressed” animals was thought to have the potential to result in a bias towards lower survival. The authors were contacted and it was verified that the statement that "animals were removed when distressed" was wording required by the local animal care committee. The tests were conducted as standard toxicity assessments with an end point of mortality. “Distressed” in this instance can be equated to mortality, ensuring that there were no testing artifacts favouring increased mortality. The decision made was to include data for the following 5 species in derivation of the short-term benchmark concentration: spotted salamander (*Ambystoma maculatum*), wood frog (*Rana sylvatica*), spring peeper (*Pseudacris crucifer*), green frog (*Rana clamitans*), and the American toad (*Bufo americanus*) (Collins and Russell 2009).

6.3.2 Invertebrates

Exposure durations of 24, 48 and 96h were reported for tests utilizing invertebrates. In general, invertebrates were found to be more sensitive to acute chloride exposures when compared to vertebrates. Some of the most sensitive species were found to be freshwater mussels as well as a freshwater clam. Four species of freshwater mussels (all tested at the glochidia life-stage, with one mussel designated as COSEWIC endangered and a second as COSEWIC special concern) and 1 species of freshwater clam (juvenile life-stage) were found to be more sensitive to short-term chloride exposures when compared to a daphnid species (neonate life-stage). Gillis (2011) observed a 24h EC50 (glochidia viability based on the ability to close valves) at 244 mg/L for the endangered northern riffleshell mussel, *Epioblasma torulosa rangiana*. The next two highest effect concentrations were for the fatmucket mussel, *Lampsilis siliquoidea*, and the COSEWIC special concern wavy-rayed lampmussel, *Lampsilis fasciola*, with 24h EC50s of 709 mg/L (Bringolf *et al*., 2007; Gillis 2011) and 746 mg/L (Valenti *et al.*, 2007; Bringolf *et al*., 2007; Gillis 2011), respectively. The 24h glochidia EC50 for the plain pocketbook *Lampsilis cardium* and the 96h LC50 for the fingernail clam *Sphaerium simile* was 817 and 902 mg/L, respectively. The waterflea *Ceriodaphnia dubia* (neonate lifestage), traditionally thought to be the most sensitive of test species, had a 48h LC50 of 1,080 mg/L (Valenti *et al*., 2007; Hoke *et al*., 1992; Mount *et al*., 1997; GLEC & INHS 2008; Elphick *et al*., 2010; Cowgill & Milazzo 1990), while the most sensitive daphnid was found to be Daphnia magna with a 48h EC50 (immobilization) of 621 mg/L (Khangarot and Ray, 1989). In addition to *Ceriodaphnia dubia*, data for four other species of water fleas was available. Effect concentrations ranged from a 48h EC50 (immobilization) of 1,213 mg/L for

Daphnia ambigua (Harmon *et al*., 2003), to a 48h LC50 of 5,308 for *Daphnia hyalina* (Baudouin & Scoppa 1974).

The encystement of freshwater mussel glochidia (valve closure) onto the host fish gill is in fact stimulated by the high salt content of the host fish gills. Therefore, if salt concentrations in surface waters are high enough, this could trigger glochidia to prematurely close their valves, which would inhibit them from encysting onto a host fish (Gillis, 2009, pers.comm.), thereby making this endpoint ecologically relevant since it is by this mechanism that glochidia are able to encyst onto host fish gills.

For some mussel species, both 24h and 48h LC50 data was provided (Appendix I). It was decided to limit freshwater mussel glochidia data used for guideline derivation to 24h exposures only (due to lack of species-specific knowledge on length of time between release into water column and attachment to fish host), ensuring only studies with >90% control survival were used (as per ASTM 2006 protocol for toxicity testing with freshwater mussels).

Glochidia are thought to survive for only a few days after release unless they are able to attach to a suitable host (Cope *et al*., 2008). Glochidia viability curves have been published for at least 35 species of mussels (ASTM 2006; Cope *et al*., 2008). The curves have shown that glochidia free-living in water can remain viable from at least 24 hours up to 10 days post-release, regardless of their host fish infection strategy and glochidia release mechanisms (Barnhart *et al*., 2008). Toxicity tests utilizing the glochidia from three species of freshwater mussels, one of which is endangered and the second designated as special concern (as designated by the Committee on the Status of Endangered Wildlife in Canada) provided the lowest effect concentrations for the entire short-term/acute exposure dataset.

The northern riffleshell mussel (*Epioblasma torulosa rangiana)* has been designated as endangered (under both COSEWIC and SARA) because this species has undergone a drastic range reduction and significant population decline throughout its range. In Canada, it is now restricted to short segments of two rivers in southern Ontario (Ausable River and Sydenham River) where it occurs at low densities and is threatened by siltation, highway and agricultural runoff and other pollutants in the water. Only four populations in the world, including the two in Canada, show signs of recruitment (COSEWIC, 2010b). With respect to the wavy-rayed lampmussel (*Lampsilis fasciola*), this species has been designated as special concern by COSEWIC and as endangered by SARA because it is confined to 4 river systems (Maitland River, Thames River, Grand River, Ausable River) and the Lake St. Clair delta in southern Ontario. All of the wavy-rayed lampmussel populations are in areas of intense agriculture and urban and industrial development, subject to degradation, siltation, and pollution. Invasive mussels continue to threaten the Lake St. Clair delta population and could be a threat to populations in the Grand and Thames rivers if they invade upstream reservoirs (COSEWIC, 2010a).

Data for 4 other species of endangered freshwater mussel glochidia was provided. Valenti *et al*., (2007) provided data for the cumberlandian combshell (*Epioblasma* *capsaeformis*) and the oyster mussel (*Epioblasma brevidens*). These 2 species are only found in the US states of Kentucky, Alabama, Tennessee and Virginia (Williams *et al*., 1992), but are considered endangered, and so were added in to the dataset, as these may be representative of other untested mussel species found in Canadian waters. The 24h EC50s for these two species (glochidia life-stage) was 1,626 and 1,644 mg Cl^-/L, respectively (Valenti *et al*., 2007). Data for another COSEWIC endangered freshwater mussel (glochidia life-stage) was provided by Gillis (2011) for the kidneyshell mussel (*Ptychobranchus fasciolaris*) (24h EC50 of 3,416 mg Cl^-/L). Wang and Ingersoll (2010) also provided data for the juvenile lifestage (≤2 months old) for the COSEWIC endangered rainbow mussel (*Villosa iris*) (96h EC50 of 1,815 mg Cl^-/L). Other species of mussel glochidia (although not considered endangered) also displayed sensitivity to chloride. Glochidia of *Lampsilis siliquoidea* were sensitive to acute chloride exposures, with a 24h EC50 of 709 mg Cl^-/L (Valenti et al., 2007; Bringolf *et al*., 2007; Gillis 2011). Juvenile (≤2 months old) mussel data was also obtained from Bringolf *et al*., (2007), with 96h EC50 (survival based on movement inside or outside of the shell) values of 2,414, 2,766 and 3,173 mg Cl^-/L for *Lampsilis fasciola*, *Lampsilis siliquoidea* and *Villosa delumbis*, respectively.

The US EPA recently updated the aquatic life ambient water quality criteria for ammonia in freshwater with the addition of new data for freshwater mussels (US EPA, 2009). Many states in the continental USA are known to have mussel species present in at least some of the surface waters. Mussel populations are on the decline, one quarter of species in the USA are listed as endangered, threatened or of special concern, and ammonia has been shown to be particularly toxic to freshwater mussels, including unionid mussels. As a result, the US EPA updated both the acute and chronic criteria to ensure that the values are protective of unionids. It was decided by the US EPA to only include juvenile mussel data, and to disclude glochidia data from guideline derivation. The full rationale for this decision is presented in US EPA (2009), but the major issue driving this decision was based on the argument that although there is a standard method for testing with glochidia (ASTM 2006), there are still a lot of uncertainties related to glochidia life history (e.g. for species of mussel that broadcast glochidia, there is no certainty related to the duration of time the glochidia remain viable from time of release to time of host encystment). The CCME Water Quality Task Group has deliberated on this issue, and has decided to include toxicity data using the glochidia life stage (24h EC50 values) for short-term benchmark concentration development. The policies of CCME are precautionary, and including data from high quality studies for the most sensitive life stage of a species is recommended in order to ensure that maximum protection is afforded to all aquatic species.

Jacobson *et al*., (1997) observed that released glochidia (in the water column) were more sensitive to copper than encysted (attached to a fish host) glochidia. A comparison of released glochidia and juveniles indicated that the two life stages had similar tolerances to copper (the rainbow mussel *Villosa iris* and the papershell *Pyganodon grandis*). Other studies have also been conducted that provide indication that the glochidia life stage is more, or just as, sensitive as the juvenile life stage (Valenti *et al.*, 2007). Augspurger *et al*., (2003) observed lower ammonia LC50 values for the glochidia of paper pondshell mussels (*Utterbackia imbecillus*), pheasantshell mussels (*Actinonaias pectorosa*) and rainbow mussel (*V. iris*) when compared to the juvenile life stage of the same species. Glochidia of the pondshell (*U. imbecillus*), little spectaclecase (*Villosa lienosa*), and downy rainbow mussel (*Villosa villosa*) were all substantially more sensitive to malathion when compared to the juvenile life stage (Keller and Ruessler, 1997).

Studies that have indicated that glochidia are more or just as sensitive to substances when compared with standard freshwater test organisms (*C. dubia*, *D. magna*, fathead minnow, *O. mykiss*) are listed in Valenti *et al*., (2007).

One species of fingernail clam was also found to be sensitive to acute chloride exposures, with a 96h LC50 of 902 mg Cl^-/L for *Sphaerium simile* (GLEC and INHS 2008). The oligochaete *Tubifex tubifex* was found to be immobilized (EC50) at a concentration of 1,204 mg Cl^-/L following a 96h exposure (Khangarot 1991), but this study was discluded due to a high test temperature. The most tolerant invertebrates were found to be the copepod (*Cyclops abyssorum prealpinus*) with a 48h LC50 of 12,385 mg/L (Baudouin and Scoppa 1974).

6.4 Long-Term (Chronic) Toxicity

Criteria used for classifying available toxicity data as either primary, secondary, or unacceptable are described in the Protocol for the Derivation of Water Quality Guidelines for the Protection of Aquatic Life (CCME 2007). In general, primary toxicity studies involve acceptable test procedures, conditions, and controls, measured toxicant concentrations, and flow-through or renewal exposure conditions. Secondary toxicity studies usually involve unmeasured toxicant concentrations, static bioassay conditions and unsatisfactory reporting of experimental data. Unacceptable data are deemed not suitable for guideline development (e.g. no reporting of controls, test temperature too high to be relevant to Canadian surface waters, test organism not representative of a temperate species, etc.).

Long-term (chronic) toxicity data studies include complete life cycle tests and partial life cycle tests involving early life stages. The following information provides a general overview of the toxicity data collected for long-term guideline derivation. The full suite of chloride toxicity data obtained from the scientific literature is presented at the end of this report in Appendix I.

6.4.1 Vertebrates

Data for three species of fish were utilized in the derivation of the long-term guideline for chloride. Effect concentrations for these three species were: a 33d LC10 of 598 mg Cl^-/L for the fathead minnow (*Pimephales promelas*) (Elphick *et al*., 2010; Birge *et al*., 1985), an 8d NOEC (survival) of 607 mg Cl^-/L for the brown trout (*Salmo trutta fario*) (Camargo and Tarazona 1991) and a 7d EC25 (embryo viability) of 989 mg Cl^-/L for the rainbow trout (*Oncorhynchus mykiss*) (Beak 1999). As with the acute dataset, chronic data for euryhaline fish species was also added to the long-term dataset. This includes the data for the brown trout (*Salmo trutta fario*) and for the rainbow trout (*Oncorhynchus mykiss*). With respect to the brown trout study (Camargo and Tarazona 1991), the authors examined the toxicity of the fluoride ion (F-) (NaF) to brown trout (and rainbow trout), and also conducted exposures using NaCl to determine if any effects observed were actually due to the F- ion. No mortality occurred during the 8 d exposures to the single high concentration of NaCl (brown trout NOEC = 606 mg Cl^-/L, RBT NOEC = 485 mg Cl^-/L), however fingerlings did show symptoms of hyperexcitability and hyperventilation at first, returning to their normal state after approximately 10 hours. Sublethal effects were not observed (hypoexcitability, darkened backs, decreased respiration). The 8d NOEC of 607 mg Cl^-/L for *Salmo trutta* was included in the long-term dataset based on guidance provided in the 2007 CCME protocol for derivation of CWQGs: "The use of toxicity data from a test where an insufficient concentration range on the higher end has been tested (i.e., where the results are expressed as "toxic concentration is greater than x"), are generally acceptable, as they will not result in an under-protective guideline". This is the only NOEC in the entire dataset, and the effect concentration for the brown trout was located in the middle of the SSD, thereby likely not having a large effect on the final guideline value. As well, the CCME (2007) protocol calls for exposure periods ≥21d for testing on juvenile and adult fish. The Camargo and Tarazona (1991) study tested fingerlings. Discussion with CCME Water Quality Task Group members resulted in agreement for the inclusion of this data point in the long term curve, which allows this species to be represented.

Since both the brown trout and the rainbow trout are considered euryhaline species, both species should be able to tolerate at least the concentration of chloride in seawater, arguably under ideal exposure scenarios of gradually increasing salinities. The no- and low-effect concentrations presented in this chronic dataset do fall considerably below the chloride concentrations measured in seaweater (19,250 mg Cl^-/L), and thus are appropriate for inclusion in the long-term dataset for setting of a water quality criteria for chloride for freshwater environments. The physiological adaptations of euryhaline fish going from a freshwater to salt water environment are explained in the section above titled "Acute toxicity: vertebrates".

Long-term exposure data was also obtained for 7 species of amphibians, 2 of which are represented in the long-term SSD. The study by Beak (1999) provided the lowest effect concentration for the amphibian dataset, with a 7d LC10 of 1,307 mg Cl^-/L for the African clawed frog *Xenopus laevis*. The most tolerant of the 3 amphibian species was the northern leopard frog (*Rana pipiens*) with a 108-d MATC (survival) of 3,431 mg Cl^-/L (Doe 2010).

A study by Sanzo and Hecnar (2006) with the wood frog *Rana sylvatica* reported the results for a 90d exposure; however, >50% mortality was observed with the controls. At day 10 of the exposure, control mortality was <20%. At day 10 of the exposure is where the significant decrease in survival is observed in the highest chloride treatment (mortality is <20% for the control, low and medium treatment exposures). From 10d to test end (90d), the rate of decrease in survival is equal for all exposure groups (control, low, medium and high). This data was discluded from the final long-term SSD dataset.

6.4.2 Invertebrates

As with the acute data, invertebrates were found to be more sensitive to chronic chloride exposures when compared to fish. Two species of freshwater mussels (glochidia lifestage), a fingernail clam and a daphnid were the most sensitive species. The respective 24h EC10 values for the COSEWIC special concern wavy rayed lampmussel (*Lampsilis fasciola*) and the COSEWIC endangered northern riffleshell mussel (*Epioblasma torulosa rangiana*) was 24 (Bringolf *et al.*, 2007) and 42 (Gillis, 2010) mg Cl^-/L. The 60-80d LOEC (reduced natality[3]) for the fingernail clam *Musculim securis* was 121 mg Cl^-/L (Mackie 1978). The 10d EC10 for the daphnid *Daphnia ambigua* was 259 mg Cl^-/L (Harmon *et al*., 2003). This again indicates that daphnids may not be the most sensitive organisms, as traditionally thought. The most tolerant invertebrate was found to be the chironomid (*Chironomus tentans*) with a 20d growth IC10 of 2,316 mg Cl^-/L (Elphick *et al.*, 2010).

High chloride concentrations in sediment pore water can lead to osmotic stress in aquatic receptors, particularly for benthic organisms residing near the sediment-water interface. A high chloride concentration can also augment the concentration of dissolved metal by forming metal-chloride complexes, for example, with cadmium (EC/HC 2001b). A study conducted by Mayer *et al*., (2008) tested the toxic impacts sediment pore-water (collected from a salt-impacted stormwater management pond) on the amphipod *Hyalella azteca*. The sediment pond pore-water was found to be toxic to the amphipod, with toxicity being related to an increased mobilization of cadmium as a result of increased chloride concentrations. The measured cadmium concentration in the sediment pore-water was 8 ug/L, while the 7d cadmium LC50 was 4.41 ug/L (Mayer *et al*., 2008).

6.4.3 Plants and Algae

Data for one aquatic plant (*Lemna minor*) and three species of algae (*Chlorella minutissimo, Chlorella zofingiensis*, and *Chlorella emersonii*) were used in long-term guideline derivation. The aquatic plant was found to be most sensitive, with a 96h growth MATC of 1,171 mg Cl^-/L (Taraldson and Norberg-King 1990). The algae were found to be as tolerant as some of the fish species to chronic chloride exposures. The 28d growth MATC for *C. minutissimo* and *C. zofingiensis* was 6,066 mg Cl^-/L for both (Kessler 1974). The 8-14d growth inhibition MATC for *C. emersonii* was 6,824 mg Cl^-/L (Setter *et al*., 1982). Kessler (1974) provided data for 8 other species of algae, all being more tolerant of chloride when compared to *C. minutissimo* and *C. zofingiensis.* This additional data was not included in the long-term guideline dataset because the Kessler (1974) paper was found to be related to taxonomy, and not toxicology, with the purpose of developing a method for identification of different algal species based on salt tolerance.

[3] Natality, as defined in Mackie (1978), is used as an index for assessment of water quality and is "a measure of population increase under an actual specific environmental condition varying with the size and composition of the population and the physical environmental conditions". Organisms selected for assessment of natality should commonly be found in aquatic systems (e.g. oligochaetes, chironomids, sphaeriids), and should bear living young (e.g. be ovoviviparous, like the sphaerid *Musculium securis*).

6.5 Summary of Toxicity Data

In the case of the acute dataset, the glochida lifestage of 1 species of freshwater mussel (*Epioblasma torulosa rangiana*) was found to be more sensitive to chloride when compared to daphnids. In the case of the chronic dataset, the glochidia lifestage of 2 species of freshwater mussels (*Lampsilis fasciola*, *Epioblasma torulosa rangiana*) and 1 fingernail clam (*Musculim securis*) were found to be more sensitive than daphnids as well. Toxicity testing with non-traditional bioassay organisms has indicated that daphnids may not be the most sensitive species to both short-term and long-term chloride exposures, as traditionally thought.

7.0 EFFECTS OF WATER QUALITY PARAMETERS ON TOXICITY

7.1 Oxygen

Aquatic species are more tolerant of salts in water with high oxygen concentrations (Evans and Frick, 2001). Toxicity thresholds for the water flea (*Daphnia magna)* exposed to NaCl with two different concentrations of dissolved oxygen (1.48 and 6.4 mg/L) were 3,170 and 5,093 mg/L, respectively (Fairchild, 1955). The low oxygen concentration was below the criterion set by the US EPA for protection of aquatic life (5 mg/L) (US EPA, 1976). Therefore, the low oxygen may be responsible for the observed toxicity. In addition, the formation of meromictic lakes associated with excess chloride from deicing salt runoff causes anoxic conditions. This can create stress to the ecosystem, adversely affecting aquatic species (Smol *et al*., 1983). Stagnant bodies of water, such as stormwater ponds or wetlands, which are impacted by increased chloride concentrations as a result of road salt runoff, often display a vertical gradient in chloride concentration. Highest chloride levels are measured at the sediment-water interface, while lowest concentrations are measured at the surface (Marsalek, 2003). This is of concern for amphibians, mussels and all other organisms dwelling at the sediment-water interface. A microcosm study conducted by Snodgrass *et al*., (2008) investigated the toxicity of stormwater pond sediment to embryos and larvae of the wood frog (*Rana sylvatica*). Stormwater sediment and overlying water metal concentrations (Cr, Cu, Ni, Zn, As, Cd, Pb) were elevated above controls (clean sand) but were found to not exceed US EPA Water Quality Criteria or consensus-based sediment quality guidelines. The source of increased metals in stormwater management ponds is linked with deterioration of car parts (brakes, tires) where metals accumulate on road surfaces and enter stormwater ponds via runoff. Chloride levels were found to be elevated in the stormwater pond sediment overlying water, with concentrations ranging from 224-1,055 mg/L. Hatchlings (Gosner stage 20-24) of *R. sylvatica* displayed 100% mortality after 13 days exposed to chloride concentrations in the range of 224 to 243 mg Cl^-/L. These toxic concentrations for chloride are lower than those reported by Sanzo and Hecnar (2006), where chronic sublethal effects were observed at 625 mg Cl^-/L for larval *R. sylvatica* (Gosner stage 25 and greater). Since Snodgrass *et al*., (2008) collected water samples from mid-depth of the microcosms, and eggs were placed on the bottom, it is possible that eggs experienced exposure to higher chloride concentrations than indicated by the water samples. Stormwater management ponds containing high levels of chloride often display a gradient in chloride concentrations, with lowest measurements at the surface, and highest measurements at the sediment-water interface. In this case, toxicity could have been due to exposure to higher chloride concentrations at the sediment-water interface, as well as exposure to polycyclic aromatic hydrocarbons and potential interaction among pollutants (Snodgrass *et al*., 2008). Environmental monitoring should take samples from the sediment-water interface, where chloride concentrations are highest.

7.2 Temperature

The effects of temperature on chloride toxicity are inconsistent, and few studies have systematically evaluated the influence of temperature on chloride toxicity. Some studies have shown that species are more tolerant to chloride at higher temperatures. For example, Kanygina and Lebedeva (1957) demonstrated that *Daphnia magna* had a greater tolerance to NaCl toxicity at 20ºC (maximum tolerance concentration 800 mg/L) than at 3ºC (maximum tolerance concentration 200 mg/L). In a series of acute exposure experiments at a concentration of 4,756 mg Cl^-/L, Waller *et al*., (1996) reported 22.1 and 93.3% mortality in the rainbow trout (*Oncorhynchus mykiss*) at 12ºC and 17ºC, respectively. However, the same study also reported 0% mortality at 4,756 mg Cl^-/L in the yellow perch (*Perca flavescens*) at both 12 and 17ºC.

7.3 Chloride and the toxicity of other compounds

Chloride affects the toxicity of other compounds. Soucek and Kennedy (2005) found that sulphate toxicity to *Hyalella azteca* decreased with increasing levels of chloride. Yanbo *et al*., (2006) reported evidence of a protective effect of chloride against nitrite toxicity in juvenile tilapias (*Oreochromis niloticus)*, where increasing chloride concentrations nearly doubled the 96 hour LC50 of nitrite. Higher chloride concentrations tend to reduce nitrite toxicity to fishes, as the chloride ion will bind competitively with chloride cells (the primary site of nitrite uptake), thereby limiting the amount of nitrate entering the blood stream (Wedemeyer and Yasutake 1978; Russo *et al*., 1981; Lewis and Morris 1986). These same chloride interactions however, do not appear to reduce the toxicity of nitrate to salmonids. For chinook salmon and rainbow trout exposed to nitrate in both freshwater and 15‰ salinity salt water, nitrate was more toxic (p <0.05) in saltwater by a factor of up to 1.4 (Westin 1974). However, no explanation was provided for the increased toxicity in trials with greater salinity. Increasing chloride concentrations also reduced percent methaemoglobin (MHb) in blood (increased MHb causes oxygen depletion, followed by anaemia and tissue hypoxia) in nitrite-exposed tilapia. Brauner *et al*., (2003) found that high chloride levels in water partially protected rainbow trout *(Oncorhynchus mykiss)* embryos and larvae from ionoregulatory disturbance and mortality caused by silver toxicity.

7.4 Hardness

A number of studies have shown that chloride toxicity is counteracted by calcium chloride in solution. Garrey (1916) reported that the toxicity of a chloride in solution (KCl, $MgCl_2$, and NaCl) to minnows (*Notropis* sp.) was reduced by the addition of 20 to 40 mg/L of $CaCl_2$. In a 24 hour exposure, Grizzle and Mauldin (1995) found that the addition $CaCl_2$ reduced the toxicity of NaCl to juvenile striped bass (*Morone saxatilis),* resulting in 50% mortality increasing from 1,400 to 18,200 mg/L NaCl as calcium concentrations increased from 3 to 100 mg/L. This effect was accounted for by the reduction of the Na^+: Ca^{2+} ratio (Evans and Frick, 2001).

Water hardness has been shown to ameliorate chloride toxicity. Lasier *et al*., (2006) documented more severe reproductive effects in the water flea *Ceriodaphnia dubia* with a reduction in water hardness. At an effect concentration of 342 mg Cl^-/L, reproduction was reduced by 12.8% with a water hardness of 100 mg/L, compared to a reduction of 37.8% with a water hardness of 46 mg/L. In the same study, alkalinity did not exert any consistent effects on chloride toxicity. The main objective of the Lasier *et al*., (2006) study was to show that organisms cultured in moderately hard water show increased stress when exposed to soft bioassay water, whereas this stress is reduced or absent when organisms cultured in soft water are exposed to very soft water. In soft water, Naumann (1934) demonstrated weakening and immobilization in *Daphnia magna* from $CaCl_2$ and KCl exposure, respectively, while no effects were observed at the same test concentrations in hard water.

Water hardness refers to the concentration of calcium (Ca^{2+}) and magnesium (Mg^{2+}) ions in water and comes mainly from the dissolution of $CaCO_3$ in calcareous soils and sediments. Alkalinity refers to the buffering capacity of water (ability to neutralize acid) (Welsh, 1996). It is primarily a measure of carbonate (CO_3^{2-}) and bicarbonate (HCO_3^-) concentrations in exposure water (Welsh, 1996). It is well known that both water hardness and alkalinity ameliorate the toxicity of metals to aquatic organisms. With respect to hardness, the mechanism behind metal toxicity mitigation involves competition between the hardness cations and metal cations for binding sites at cellular surfaces (e.g. fish gills) (Paquin *et al*., 2002). Of the two hardness cations, Ca^{2+} has been identified as the primary cation involved in protecting against metal uptake and toxicity in both fish (Part *et al*., 1985; Carrol *et al*., 1979) and invertebrates (Heijerick *et al*., 2002; Jackson *et al*., 2000; Wright 1980). The reason Ca^{2+} may exert a more protective effect is because the molar concentration of Ca^{2+} is typically twice that of Mg^{2+} in surface waters (Everall *et al*., 1989). Alkalinity reduces metal toxicity by decreasing the number of free metal ions by forming metal-CO_3^{2-} or metal-HCO_3^- complexes (Welsh, 1996). In order to be able to determine whether or not hardness alone has the ability to ameliorate toxicity, one would need to isolate for true hardness, for example, by adding Ca^{2+} in the form of $CaSO_4$ or $CaCl_2$ to exposure water. Tests that add in $CaCO_3$ salts to the exposure solutions will actually confound the effects of hardness with alkalinity (Charles *et al*., 2002).

Recently, the US EPA worked with the state of Iowa to update the state's water quality criteria for chloride (for the protection of aquatic life). A literature review of current data provided indication that water hardness may in fact be ameliorating the toxicity of chloride to aquatic receptors. The mechanism behind chloride toxicity amelioration would differ when compared to metals. Due to the negative charge of the chloride ion (Cl^-), the hardness cations Ca^{2+} and Mg^{2+} would form complexes with the Cl^- ion inhibiting Cl^- ion uptake by the aquatic receptor. In order to determine whether or not a hardness adjustment for chloride criteria development was warranted, additional testing for the US EPA was conducted by two laboratories, the Great Lakes Environmental Centre and the Illinois National History Survey, using the cladoceran *Ceriodaphnia dubia*, the fingernail clam *Sphaerium simile*, the tubificid worm *Tubifex tubifex*, and the planorbid snail *Gyraulus parvus* (GLEC and INHS, 2008).

Ceriodaphnia dubia (water flea) 48h LC50 data was collected from exposures to waters of varying hardness (25, 50, 100, 200, 400, 600, 800 mg/L as $CaCO_3$) and a constant sulphate concentration (65 mg/L). The 48h LC50 values approximately <u>doubled</u> when comparing results using soft exposure water, with a hardness of 25 mg/L (48h LC50 = 947 mg/L for GLEC, 48h LC50 = 1007 mg/L for INHS), with results using extremely hard exposure water, where the hardness was 800 mg/L (48h LC50 = 1764 mg/L for GLEC, 48h LC50 = 1909 mg/L for INHS). It was concluded that the relationship between chloride LC50 and water hardness was strong, with an R-squared of 0.78 on untransformed arithmetic data, or with an R-squared of 0.78 or 0.82 on semi-log or log-log transformation. The slope was approximately 1.2 (mg Cl^-/L per mg/L as $CaCO_3$) and intercept was approximately 1,000 mg Cl^-/L (GLEC & INHS, 2008). Results from this study are presented in Table 7.1 (for comparative purposes, hardness and sulphate concentrations in the geographic regions of Canada are presented in Table 11.1). This exposure with *Ceriodaphnia dubia* was conducted in a manner that isolated for the effects of true hardness. The exposure medium was prepared using chloride salts of K^+, Mg^{2+}, and Ca^{2+}, and sulphate salt of Na^+, plus addition of $NaHCO_3$ (GLEC & INHS, 2008). The addition of KCl, $NaHCO_3$ and Na_2SO_4 salts remained constant while the addition of $CaCl_2$ and $MgCl_2$ increased in order to increase hardness (Ca^{2+} and Mg^{2+} cation) levels in the exposure medium (GLEC & INHS, 2008). The Ca^{2+}:Mg^{2+} ratio was maintained at 2.25 over the varying hardness concentrations (25, 50, 100, 200, 400, 600, 800 mg/L as $CaCO_3$), similar to the ratio found in natural surface waters. All other ions, with the exception of Cl^-, remained constant over the varying hardness concentrations, including K^+, Na^+, SO_4^{2-}, and HCO_3^-. <u>When taking into consideration the reasonable extremes of water hardness values for Canadian surface waters (5 mg/L to 240 mg/L as $CaCO_3$) (CCME, 1987; NRCAN, 1978), there appears to only be a minor effect of hardness on chloride toxicity to *C. dubia* (Table 7.1).</u>

Table 7.1 Summary of studies that investigated hardness as a toxicity modifying factor.

Taxa/organism	Short-term or long-term	Tox. Endpoint	Effective Concentration (Cl mg/L)	Hardness (as mg/L $CaCO_3$)	Effect of hardness on toxicity[1]	Comments	Reference
Fish (short-term)							
Fathead minnow *Pimephales promelas*	short-term (96h)	LC50	2,790 2,123	39.2	No apparent effect of hardness on toxicity.	Two effect concentrations were provided for the exposure conducted in soft reconstituted water.	US EPA 1991[4]
		LC50	2,244	339			
Fathead minnow *Pimephales promelas*	short-term (96h)	LC50	4,167	81.4	No apparent effect of hardness on toxicity.		WISLOH 2007[4]
		LC50	4,127	169.5			
Invertebrates (short-term and long-term)							
Wavy-rayed lampmussel *Lampsilis siliquoidea*	Short-term (24h)	EC50	763	47	Substantial effect of hardness on toxicity. A 6.85-fold increase in hardness (763 to 1870 mg/L) results in a ~ 2.45-fold decrease in toxicity.		Gillis 2011
		EC50	1430	99			
		EC50	1962	172			
		EC50	1870	322			
Water flea *Ceriodaphnia dubia*	short-term (48h)	LC50[2]	977	25	Minor effect of hardness on toxicity. A 32-fold increase in hardness results in a ~ 1.9-fold decrease in toxicity. In the hardness range relevant to Canadian surface waters (5 mg/L to 240 mg/L as $CaCO_3$), there is still only a minor effect of hardness on toxicity. An 8-fold increase in hardness results in a ~ 1.4-fold decrease in toxicity.	Alkalinity ranged from 60-68 mg/L as $CaCO_3$, and pH ranged from 7.9-8.2. The calcium to magnesium ratio was maintained at approximately 2.25 for all levels of total hardness.	GLEC and INHS 2008[3,5]
		LC50	861	50			
		LC50	1,250	100			
		LC50	1,402	200			
		LC50	1,589	400			
		LC50	1,779	600			
		LC50	1,836	800			
Water flea *Ceriodaphnia dubia*	short-term (48h)	LC50	1,395 1,638 1,274 1,395	39.2	No apparent effect of hardness on toxicity.	Four effect concentrations provided for the exposure conducted in soft reconstituted water.	US EPA 1991[4]

Taxa/organism	Short-term or long-term	Tox. Endpoint	Effective Concentration (Cl mg/L)	Hardness (as mg/L $CaCO_3$)	Effect of hardness on toxicity[1]	Comments	Reference
		LC50	1,698	339			
Water flea *Ceriodaphnia dubia*	short-term (96h)	LC50	1,677	81.4	No apparent effect of hardness on toxicity.		WISLOH 2007[4]
		LC50	1,499	169.5			
Water flea *Ceriodaphnia dubia*	long-term (7d reproduction)	IC25	147	44 (45 mg/L alkalinity)	Substantial effect of hardness on toxicity. A 2.1-fold increase in hardness (44 to 93 mg/L) results in a ~ 2.6-fold decrease in toxicity.	The Ca^{2+}:Mg^{2+} ratio was consistent at 1.15 for all of the exposure waters. This is lower than the typical ratio found in natural waters, where Ca^{2+} is typically double that of Mg^{2+}. Chloride toxicity was also reduced in water with moderate alkalinity compared to low alkalinity water (when measured at the same hardness of 44 mg/L). The authors conclude that the reduction in chloride toxicity was due to the increase in Na^+ rather than the increase in alkalinity (alkalinity provided by additions of $NaHCO_3$).	Lasier and Hardin 2009
		IC25	340	44 (101 mg/L alkalinity)			
		IC25	379	93 (66 mg/L alkalinity)			
Water flea *Ceriodaphnia dubia*	long-term (7d reproduction)	IC50	342	44 (45 mg/L alkalinity)	Substantial effect of hardness on toxicity. A 2.1-fold increase in hardness (44 to 93 mg/L) results in a ~ 1.9-fold decrease in toxicity.	The Ca^{2+}:Mg^{2+} ratio was consistent at 1.15 for all of the exposure waters. This is lower than the typical ratio found in natural waters, where Ca^{2+} is typically double	Lasier and Hardin 2009
		IC50	563	44 (101 mg/L alkalinity)			
		IC50	653	93 (66 mg/L alkalinity)			

Taxa/organism	Short-term or long-term	Tox. Endpoint	Effective Concentration (Cl mg/L)	Hardness (as mg/L $CaCO_3$)	Effect of hardness on toxicity[1]	Comments	Reference
						that of Mg^{2+}. Chloride toxicity was also reduced in water with moderate alkalinity compared to low alkalinity water (when measured at the same hardness of 44 mg/L). The authors conclude that the reduction in chloride toxicity was due to the increase in Na^+ rather than the increase in alkalinity (alkalinity provided by additions of $NaHCO_3$).	
Water flea *Ceriodaphnia dubia*	long-term (7d reproduction)	IC25	117	10	Minor effect of hardness on toxicity. A 32-fold increase in hardness results in a ~ 4.5-fold decrease in toxicity.		Elphick *et al.*, 2010
		IC25	264	20			
		IC25	146	40			
		IC25	454	80			
		IC25	580	160			
		IC25	521	320			
Water flea *Ceriodaphnia dubia*	long-term (7d reproduction)	IC50	161	10	Minor effect of hardness on toxicity. A 32-fold increase in hardness results in a ~ 4.4-fold decrease in toxicity.		Elphick *et al.*, 2010
		IC50	301	20			
		IC50	481	40			
		IC50	697	80			
		IC50	895	160			
		IC50	700	320			
Water flea *Ceriodaphnia dubia*	long-term (7d survival)	LC50	132	10	Substantial effect of hardness on toxicity. A 32-fold increase in hardness results in a ~ 9.9-fold decrease in toxicity.		Elphick *et al.*, 2010
		LC50	316	20			
		LC50	540	40			
		LC50	1134	80			
		LC50	1,240	160			
		LC50	1,303	320			

Taxa/organism	Short-term or long-term	Tox. Endpoint	Effective Concentration (Cl mg/L)	Hardness (as mg/L $CaCO_3$)	Effect of hardness on toxicity[1]	Comments	Reference
Fingernail clam *Sphaerium simile*	short-term (96h)	LC50	740	50	Substantial effect of hardness on toxicity. A 4-fold increase in hardness results in a ~ 1.5-fold decrease in toxicity.	At low hardness, alkalinity was 64 mg/L as $CaCO_3$ and pH was 7.8. At high hardness, alkalinity was 61 mg/L as $CaCO_3$ and pH was 7.9. The calcium to magnesium ratio was maintained at approximately 2.25 for all levels of total hardness.	GLEC and INHS 2008[3,5]
		LC50	1,100	200			
Tubificid worm *Tubifex tubifex*	short-term (96h)	LC50	4,278	50	Substantial effect of hardness on toxicity. A 4-fold increase in hardness results in a ~ 1.4-fold decrease in toxicity.	At low hardness, alkalinity was 60 mg/L as $CaCO_3$ and pH was 7.6. At high hardness, alkalinity was 56 mg/L as $CaCO_3$ and pH was 7.7. The calcium to magnesium ratio was maintained at approximately 2.25 for all levels of total hardness.	GLEC and INHS 2008[3,5]
		LC50	6,008	200			
Planorbid snail *Gyraulus parvus*	short-term (96h)	LC50	3,078	50	No apparent effect of hardness on toxicity.	At both low and high hardness, alkalinity was 56 mg/L as $CaCO_3$ and pH was 7.7. The calcium to magnesium ratio was maintained at approximately 2.25 for all levels of total hardness.	GLEC and INHS 2008[3,5]
		LC50	3,009	200			
Fingernail clam *Sphaerium tenue*	short-term (96h)	LC50	698	20	No apparent effect of hardness on toxicity.		Wurtz and Bridges 1961[4]
		LC50	667	100			
Snail *Physa*	short-term	LC50	2,487	20	Substantial effect of hardness on toxicity for non-juveniles only.		Wurtz and Bridges 1961[4]
		LC50	3,094	100			

Taxa/organism	Short-term or long-term	Tox. Endpoint	Effective Concentration (Cl mg/L)	Hardness (as mg/L $CaCO_3$)	Effect of hardness on toxicity[1]	Comments	Reference
heterostropha	(96h)		3,761		A 5-fold increase in hardness results in a ~ 1.2- to 1.5-fold decrease in toxicity.		
Isopod *Asellus communis*	short-term (96h)	LC50	3,094	20	Substantial effect of hardness on toxicity. A 5-fold increase in hardness results in a ~ 1.6-fold decrease in toxicity.		Wurtz and Bridges 1961[4]
		LC50	5,004	100			
Damselfly *Argia* sp.	short-term (96h)	LC50	13,952	20	Minor effect of hardness on toxicity. A 5-fold increase in hardness results in a ~ 1.0-fold decrease in toxicity.		Wurtz and Bridges 1961[4]
		LC50	14,558	100			
Plants, including algae							
NA							

[1]For the purposes of a simple trend analysis, results were compared on a mg/L basis; however, a molar comparison would be more appropriate, since hardness is believed to ameliorate toxicity through competition at the site of uptake. The qualitative terms of "no apparent effect", "minor effect" and "substantial effect" are subjectively assigned, but consistent among studies. "No apparent effect" was assigned if there was no consistent decrease in toxicity with increasing hardness. "Substantial effect" was assigned if the ratio of decrease in toxicity to increase in hardness was greater than or equal to 0.21. For example, in the fourth entry under invertebrates (*Ceriodaphnia dubia*), this ratio is 2.6/2.1 = 1.2; hence, this would be classified as substantial effect. The 0.21 cut-off is derived from the subjective estimate of the reasonable extremes of water hardness values (5 mg/L to 240 mg/L as $CaCO_3$, or 48-fold [NRCAN, 1978; see Section 11.0]), and an arbitrary decrease in toxicity (10-fold decrease, a common safety factor used). Hence, 10-fold/48-fold = 0.21. "Minor effect" was assigned if the ratio was less than 0.21.

[2]The LC50 data presented is the mean LC50 value from the two separate laboratories, GLEC and INHS.

[3]Exposures were conducted using a constant sulphate concentration of 65 mg/L.

[4] Unknown if Ca was added in as $CaCO_3$ (where true hardness is confounded by alkalinity) or as $CaSO_4$.

[5] $MgCl_2$ and $CaCl_2$ (anhydrous) were used to manipulate water hardness to the desired level. These salts were selected over $MgSO_4$ and $CaSO_4$ to manipulate hardness in order to maintain the sulphate level near 65 mg/L. The calcium to magnesium ratio was maintained at approximately 2.25 for all levels of total hardness. The sulphate concentration was maintained at approximately 65 mg/L through the addition of Na2SO4, alkalinity was maintained between 60-70 mg/L with the addition of NaHCO3, and potassium was maintained at about 2 mg/L with the addition of KCl.

Acute (96h) toxicity tests were also conducted using the juvenile fingernail clam *Sphaerium simile*, mixed ages of the planorbid snail *Gyraulus parvus*, and mixed ages of the tubificid worm *Tubifex tubifex* (GLEC and INHS, 2008). Toxicity tests were conducted using exposure water of varying hardness (50 and 200 mg/L as $CaCO_3$) and constant sulphate concentration (65 mg/L). Hardness appears to ameliorate toxicity for both *S. simile* and *T. tubifex,* but not for *G. parvus*. Results from this study are presented in Table 13. This exposure, conducted in a similar manner to the one noted above using *Ceriodaphnia dubia,* was conducted in order to isolate for the effects of true hardness, with a Ca^{2+}:Mg^{2+} ratio maintained at 2.25 over the varying hardness concentrations (50 and 200 mg/L as $CaCO_3$) and an unchanging concentration of the ions K^+, Na^+, SO_4^{2-}, and HCO_3^-. When taking into consideration the reasonable extremes of water hardness values for Canadian surface waters (5 mg/L to 240 mg/L as $CaCO_3$) (CCME, 1987; NRCAN, 1978), there appears to be a substantial effect of hardness on chloride toxicity to both *S. simile* and *Tubifex tubifex*, and no apparent effect of hardness on chloride toxicity to *G. parvus* (Table 7.1).

Other studies reported in Table 7.1 that tested the effects of hardness on chloride toxicity include the following.

Gillis (2011) determined chloride 24h EC50 (survival of glochidia) values for the the wavy-rayed lampmussel *Lampsilis siliquoidea* exposed to soft (47 mg/L as $CaCO_3$), moderately hard (99 mg/L as $CaCO_3$), hard (172 mg/L as $CaCO_3$) and very hard (322 mg/L as $CaCO_3$) reconstituted waters. When taking into consideration the reasonable extremes of water hardness values for Canadian surface waters (5 mg/L to 240 mg/L as $CaCO_3$) (CCME, 1987; NRCAN, 1978), there appears to be a substantial effect of hardness on chloride toxicity to this species of freshwater mussel (Table 7.1).

Wurtz and Bridges (1961) determined 96h TLm values (Median Tolerance Limit) for four species (fingernail clam *Sphaerium tenue*, snail *Physa heterostropha*, isopod *Asellus communis*, damselfly *Argia* sp.) exposed to NaCl at two levels of water hardness, soft (20 mg/L total hardness) and hard (100 mg/L total hardness). Soft and hard dilution water chloride (6 mg/L for both soft and hard), alkalinity (20 mg/L for soft and 60 mg/L for hard), and pH (7.30 for soft and 7.85 for hard) characteristics were also reported. NaCl was then added to these soft and hard dilution waters and exposure concentrations were set up as a series of bisections of a logarithmic scale. The results do not indicate an effect of hardness on chloride toxicity amelioration, and this may be in part due to the lower range in hardness used (20 to 100 mg/L) compared to that of GLEC and INHS (2008) (25 to 800 mg/L). Results from Wurtz and Bridges (1961) are presented in Table 13. When taking into consideration the reasonable extremes of water hardness values for Canadian surface waters (5 mg/L to 240 mg/L as $CaCO_3$) (CCME, 1987; NRCAN, 1978), there appears to be a substantial effect of hardness on chloride toxicity to both *P. heterostropha* and *A. communis*, a minor effect of hardness on chloride toxicity to *Argia* sp., and no apparent effect of hardness on chloride toxicity to *S. tenue* (Table 7.1).

Data from a study conducted by the Environmental Research Laboratory (ERL) in Duluth was reported in USEPA (1991). The study examined the effects of hardness on the toxicity of NaCl to the fathead minnow (*Pimephales promelas*) and the water flea (*Ceriodaphnia dubia*) in both soft (39.2 mg/L hardness) and very hard (339 mg/L hardness) reconstituted water. The results did not indicate an effect of hardness on chloride toxicity amelioration, and are presented in Table 7.1. The original report generated by ERL-Duluth (cited in USEPA 1991) was not obtained, and so confirmation could not be made whether or not the exposure was measuring the effects of true hardness. When taking into consideration the reasonable extremes of water hardness values for Canadian surface waters (5 mg/L to 240 mg/L as $CaCO_3$) (CCME, 1987; NRCAN, 1978), there appears to be no apparent effect of hardness on chloride toxicity to both *P. promelas* and *C. dubia* (Table 7.1). This data from ERL-Duluth was not used for CCME WQG derivation because the original report was not obtained for review.

Data from a study conducted at the Wisconsin State Laboratory of Health (WISLOH 2007) was reported in USEPA (1991). The study examined the effects of hardness on the toxicity of NaCl to the fathead minnow (*Pimephales promelas*) and the water flea (*Ceriodaphnia dubia*) in both soft (81.4 mg/L hardness) and hard (169.5 mg/L hardness) water. The results do not indicate an effect of hardness on chloride toxicity amelioration, and are presented in Table 7.1. The original report generated by WISLOH (cited in USEPA 1991) was not obtained, and so confirmation could not be made whether or not the exposure was measuring the effects of true hardness. When taking into consideration the reasonable extremes of water hardness values for Canadian surface waters (5 mg/L to 240 mg/L as $CaCO_3$) (CCME, 1987; NRCAN, 1978), there appears to be no apparent effect of hardness on chloride toxicity to either *P. promelas* or *C. dubia* (Table 7.1). This data from WISLOH was not used for CCME WQG derivation because the original report was not obtained for review.

A study conducted by Elphick *et al*., 2010 assessed the potential effect of hardness on ameliorating chloride toxicity. Chronic (7 day) toxicity tests (reproduction and survival) were conducted using the water flea *C. dubia*. At a hardness range of 10 to 160 mg/L, a decrease in chloride toxicity (for both reproduction and survival) was observed with increasing hardness. Similar effect concentrations (for both reproduction and survival) were observed at both 160 and 320 mg/L total hardness, indicating that an additional reduction in toxicity is not provided by hardness >160 mg/L. Results are presented in Table 7.1. The study concluded that the relationship between chloride IC25, IC50 and LC50 for *C. dubia* and water hardness (10 to 160 mg/L) was strong, with R-square values of 0.8, 0.9 and 0.9, respectively, using semi-log transformation data (log hardness). When taking into consideration the reasonable extremes of water hardness values for Canadian surface waters (5 mg/L to 240 mg/L as $CaCO_3$) (CCME, 1987; NRCAN, 1978), the IC25 and IC50 reproduction data indicates only a minor effect of hardness, while the LC50 survival data indicates a substantial effect of hardness (Table 7.1). With respect to water chemistry, the author states that exposure waters were prepared by addition of reagent grade salts to deionized water in the ratio recommended by Environment Canada (1990) to achieve the target hardness concentrations.

A recently published study by Lasier and Hardin (2009) assessed the chronic toxicity of chloride to *C. dubia* in low- and moderate-hardness waters with a three-brood reproduction test. Chloride was found to be significantly less toxic in moderate-hardness water when compared to low-hardness water. Alkalinity was also shown to have an impact on decreasing chloride toxicity. Chloride toxicity was reduced in low-hardness (40 mg/L as $CaCO_3$) moderate-alkalinity (100 mg/L) water when compared to exposures in low-hardness (40 mg/L) low-alkalinity (40 mg/L) water (Table 7.1). When taking into consideration the reasonable extremes of water hardness values for Canadian surface waters (5 mg/L to 240 mg/L as $CaCO_3$) (CCME, 1987; NRCAN, 1978), a substantial effect of water hardness was observed, but this could be confounded by alkalinity (Table 7.1).

7.4.1 Discussion on Development of a Hardness-adjusted Guideline

With respect to the assessment of hardness-toxicity data for potential inclusion in the development of a hardness-adjusted short-term benchmark concentration or long-term CWQG, CCME follows the guidance provided by US EPA (2001). The guidance states that "in order for a species to be included, definitive acute / chronic values have to be available over a range of hardness such that the highest hardness is at least 3 times the lowest, and such that the highest hardness is at least 100 mg/L higher than the lowest". This guidance is also stated in the CCME scientific criteria documents for the development of CWQG values for cadmium (CCME 2010a) and zinc (CCME 2010b). With respect to this guidance, the only long-term study listed in Table 7.1 that would quality would be the data presented for *C. dubia* by Elphick *et al.*, (2010). In terms of the short-term studies listed in Table 7.1, studies for 6 species qualified which included *P. promelas* (US EPA 1991), *C. dubia* (GLEC & INHS, 2008; US EPA 1991), *L. siliquoidea* (Gillis, 2011); *S. simile* (GLEC & INHS, 2008), *T. tubifex* (GLEC & INHS, 2008) and *G. parvus* (GLEC & INHS, 2008). However, the original studies presented in US EPA 1991 were not obtainable, and so this data was excluded from consideration, resulting in no fish data for assessment of hardness-toxicity relationship for chloride. Of the long-term invertebrate studies that met the US EPA (2001) criteria, a substantial effect of hardness on chloride toxicity was observed for the freshwater mussel *L. siliquoidea* glochidia, the fingernail clam *S. simile* and the oligochaete *T. tubifex* (Table 7.1). Only a minor effect of hardness was observed for the water flea *C. dubia*, and no apparent effect of hardness was observed for the planorbid snail *G. parvus* (Table 7.1). This results in one long-term study for one species (*C. dubia*), and 3 short-term studies for 3 species (*L. siliquoidea, S. simile, T. tubifex*), that show a substantial effect of water hardness ameliorating chloride toxicity. No data was available for plants and algae.

For comparative purposes, both the draft short-term benchmark concentrations and long-term CWQG values for cadmium (CCME 2010a) and zinc (CCME 2010b) were adjusted for water hardness. In the case of cadmium, short-term data for 12 fish and invertebrate speices and long-term data for 1 fish and 2 invertebrate speices, met the criteria of US EPA (2001) and were used to calculate slopes for the hardness-toxicity relationship. In the case of zinc, short-term data for 12 fish and invertebrate species and long-term data for 2 fish, 2 invertebrates and 1 algae, met the criteria of US EPA (2001) and were used to calculate slopes for the hardness-toxicity relationship.

In theory, there may be sufficient short-term hardness-toxicity relathionships to adjust the short-term benchmark concentration for chloride for hardness effects. However, since the long-term CWQG cannot be adjusted for hardness based on only one study for one species, the CCME Water Quality Task Group decided that there would be no hardness adjustment of short-term benchmark concentration value at this time. Jurisdictions will have the option of adjusting for site-specific hardness conditions, if they so choose, with the development of site-specific water quality guidelines (or objectives).

One study that should be highlighted at this point is that of Mount *et al*., (1997) in which evidence is provided that a reduction in chloride toxicity is based on a multi-ion effect rather than a hardness effect. The study assessed the acute toxicity of major ions to three species of organisms, the daphnids *Ceriodaphnia dubia* and *Daphnia magna*, as well as the fathead minnow *Pimephales promelas*. The study findings were that for *C. dubia* and *D. magna*, the toxicity of the Cl^- ion was reduced in solutions containing more than one cation. This effect of multiple cations was not found to be an effect of hardness alone. One example is in the comparison of the *C. dubia* 48h LC50 values for NaCl and $CaCl_2$. When expressed on a Cl^- ion basis, the 48h LC50 values were almost identical (1,187 and 1,172 mg/L, respectively), even though the solutions had greatly different hardness (exact hardness values not provided). Another example is with the addition of NaCl to KCl, where the *C. dubia* 48h LC50 increased from 329 mg K/L for KCl to 458 mg K/L for a NaCl + KCl mix, even though hardness levels were the same in both solutions (exact hardness values not provided).

Decision: Insufficient data was available in order to develop a hardness relationship for chronic toxicity and thus, a hardness based national CWQG was not developed. CCME will re-visit the chloride guidelines when sufficient studies are available. Jurisdictions have the option of deriving site-specific hardness adjusted water quality criteria if they so choose.

8.0 OTHER EFFECTS OF CHLORIDE

8.1 Impact on Taste and Odour of Water and Fish Tainting

The Health Canada (1987) chloride Guideline for Canadian Drinking Water Quality is an aesthetic objective of $\leq$250 mg/L. This value was selected as chloride concentrations above 250 mg/L in drinking water may cause corrosion in the distribution system (Health Canada, 1987). The taste threshold for chloride, which is dependent on the associated cation, generally ranges from 200 to 300 mg/L (WHO, 2003). Chloride concentrations detected by taste in drinking water panels of greater than or equal to 18 people were 210, 310, and 222 mg/L, respectively, for sodium chloride, potassium chloride and calcium chloride (Lockhart *et al*., 1955). In addition, the taste of coffee was adversely affected at chloride concentrations of 200, 450, and 530 mg/L for sodium chloride, potassium chloride and calcium chloride, respectively (Lockhart *et al*., 1955). Increasing chloride concentrations in surface water and groundwater not only pose a hazard to aquatic biota, but also to drinking water systems. For example, the Municipality of Heffley Creek in British Columbia reported contamination of two municipal water supply wells in excess of 3,000 mg chloride/L (Canadian Drinking Water Quality Standard for chloride is 250 mg/L), where the source was leachate from adjacent abrasive and salt storage piles (CEPA, 2001). In Meriano *et al*., (2009) it is stated that “there are no major removal mechanisms for road salts from subsurface and surface waters, and as a result, their concentration can build up. For example, chloride concentrations throughout a highly urbanized watershed on the north shore of Lake Ontario in the city of Pickering (Frenchman's Bay), consistently exceed the Ontario Drinking Water Aesthetic Objective of 250 mg/L". The implementation of large-scale treatment systems to remove chloride from drinking water sources has not taken place due to the high use of energy as well as high cost of implementation.

A scientific literature search indicated that there were no data on the tainting of fish tissues for chloride.

8.2 Mutagenicity

A comprehensive scientific literature search indicated that there was no mutagenicity or genotoxicity information available for aquatic plants and animals exposed to chloride. For KCl, toxicity tests on laboratory animals did not produce adverse mutagenic effects (Myron L. Company, 2006). In general, chloride and its salts do not appear to be mutagenic. The only evidence of chloride mutagenicity was found from chronic exposure to NaCl tablets, which yielded mixed results in mouse lymphoma assay, and inconclusive results in an *in vitro* chromosome aberration assay (Eli Lilly and Company, 2001). NaCl did not induce chromosomal damage (sister chromatid exchanges) (Eli Lilly and Company, 2001).

8.3 Bioaccumulation

Bioaccumulation is the process whereby living organisms accumulate substances in their tissues from water and diet. Calculated log K_{ow} values for potassium chloride and sodium chloride of -0.42 and -3, respectively, have been reported (CCOHS, 1991; OECD, 2001). Chloride is highly soluble in water, and concentrations in water are not greatly affected by chemical reactions, and evaporation and dilution are the main processes that affect concentrations in water (Mayer *et al.,* 1999).

Some elements may be highly accumulated from the surrounding medium because of their nutritional essentiality (Schlekat *et al.*, 2007). This is the case of the chloride ion which is essential for plants and animals (Markert, 1994). For example, chloride is the main extracellular anion in the vertebrate body, maintaining proper osmotic pressure, water balance, and acid-base balance; it is an essential co-factor for plant photosynthesis (Health Canada, 1987).

A 'generic' bioaccumulation factor (BAF) of chloride for the human body can be calculated based on a typical chloride content of 105 g/70 kg body weight (Health Canada, 1987), and a world-average chloride concentration in freshwater streams of 8 mg/L (Reimann and de Caritat, 1998). The derived BAF of 187.5 L/kg is consistent with the idea that this ion is actively taken up by living organisms because of its essentiality. In other words, a high bioaccumulation potential for an essential substance does not bear at all the negative connotation of high bioaccumulation potential attributed to persistent organic pollutants (POPs).

The bioaccumulation potential of chloride may be also evaluated from the angle of dose-response relationships obtained in polluted environments. Kayama *et al.*, (2003) studied bioaccumulation of Na^+ and Cl^- in two spruce species planted along roadsides in Japan. Average chloride concentrations in needles were significantly higher in trees near roadsides than in ones from a control site at 30-32 m from the edge of the highway (Paired T-test based on age strata: species 1: 2840 vs 1970 µg Cl^-/g dry wt, P=0.06; species 2: 3760 vs 2260 µg Cl^-/g dry wt, P=0.002). The authors determined that chloride was a primary source of stress resulting in suppression of tree growth at the site impacted by road salts.

8.4 Other Effects

After a comprehensive literature search, no information on the protection of recreational water uses based on public health concerns, wildlife protection, toxicant interactions or sediment quality were identified for chloride.

8.5 Dermal Effects

No information on the protection of recreational water uses based on dermal exposure was identified. No dermal effects are expected.

9.0 CANADIAN WATER QUALITY GUIDELINES

9.1 Long-term Canadian Water Quality Guidelines and Short-term Benchmark Concentrations for the Protection of Freshwater and Marine Aquatic Life

Canadian Water Quality Guidelines for the Protection of Aquatic Life are nationally accepted threshold values for substances and other attributes (such as pH and temperature) in water. These values are determined such that no adverse toxic effects are expected in aquatic plants and animals. A CWQG for the protection of aquatic life can either be numerical or narrative and is developed using the most current scientific information available at the time of derivation. Data available from algae, macrophytes, invertebrates, and vertebrates are all considered. The development of a CWQG is based on the toxicity data. Implementation issues (e.g. technological and economic feasibility) are not taken into consideration. A CWQG is not a regulatory instrument, but can be used to derive Water-Quality-Based effluent limits, which are legally enforceable (e.g. Certificates of Approval for waste dischargers). A CWQG can be the basis for the derivation of site-specific guidelines (e.g. derived using site-specific aquatic receptors). The guidelines are management tools constructed to ensure that anthropogenic stresses, such as the introduction of toxic substances, do not result in the degradation of Canadian waters.

A CWQG is a maximum concentration of a substance that can be measured in an aquatic environment in order to be protective of all forms of aquatic life (all species, all life stages) for indefinite exposure periods. The development of a CWQG for chloride will assist environmental risk assessors and risk managers to better assess the potential impacts of chloride to aquatic ecosystems. A strong need to develop a CWQG for chloride exists for the following reason. The Priority Substances List Assessment Report for Road Salts was published on December 1, 2001. The report concluded that Road Salts that contain inorganic chloride salts with or without ferrocyanide salts have adverse impacts on the environment and are therefore toxic under subsections 64(*a*) and (*b*) of CEPA 1999. This has led to the development of a Code of Practice for the Environmental Management of Road Salts developed to manage risks posed to the environment by road salts (Environment Canada, 2004). As well, monitoring data (e.g. Ontario's Ministry of Environment Provincial Water Quality Monitoring Network) strongly indicates that chloride concentrations in surface waters are increasing, especially in small urban watersheds where road densities are high.

In 2007, the CCME established a new protocol for deriving water quality guidelines for the protection of aquatic life. Under the new protocol (CCME, 2007) there are currently three methods for the development of a CWQG, and each varies based on minimum data (quality and quantity) requirements. The three methods are:

1) Statistical approach (Type A or SSD approach),
2) Lowest endpoint approach using only primary data with a safety factor (Type B1),
3) Lowest endpoint approach using primary and/or secondary data with a safety factor (Type B2).

The minimum data requirements for each of these three methods are presented in Tables 1 and 3 in CCME (2007) and shown here as Tables 9.1 and 9.2.

Table 9.1 Minimum data set requirements for the generation of a short-term freshwater benchmark concentration and a long-term freshwater CWQG following the 2007 CCME guideline protocol (CCME 2007).

Derivation Method	**Minimum Toxicity Dataset**
Type A Guideline	Toxicity tests required for the generation of an SSD, broken out as follows: Fish: 3 studies on 3 different species including 1 salmonid, 1 non-salmonid. Invertebrates: 3 studies on 3 different species including 1 planktonic crustacean, 2 others. For semi-aquatic invertebrates, the life stages tested must be aquatic. It is desirable, but not necessary, that one of the aquatic invertebrate species be either a mayfly, caddisfly, or stonefly. Plant/Algae: For short-term guidance: none (for non-phytotoxic substances), 2 studies (for phytotoxic substances). For long-term guidance: At least one study on a freshwater vascular plant or freshwater algal species (for non-phytotoxic substances), 3 studies (for phytotoxic substances) Toxicity data for amphibians are highly desirable, but not necessary. Data must represent fully aquatic stages. Acceptable endpoints for short-term guidance: LC/EC50 (severe effects) Acceptable endpoints for long-term guidance: Most appropriate ECx/ICx representing a no-effects threshold > EC10/IC10 > EC11-25/IC11-25 > MATC > NOEC > LOEC > EC26-49/IC26-49 > nonlethal EC50/IC50. Note: Primary or secondary no- and low-effects data are acceptable to meet the minimum data requirements.

Derivation Method	Minimum Toxicity Dataset
Type B1 Guideline	Toxicity tests required for the generation of a Type B1 guideline, broken out as follows: Fish: 3 studies on 3 different species including 1 salmonid, 1 non-salmonid. Invertebrates: 3 studies on 3 different species including 1 planktonic crustacean, 2 others. For semi-aquatic invertebrates, the life stages tested must be aquatic. It is desirable, but not necessary, that one of the aquatic invertebrate species be a mayfly, caddisfly, or stonefly. Plant/Algae: For short-term guidance: none (for non-phytotoxic substances), 2 (for phytotoxic substances). For long-term guidance: At least one study on a freshwater vascular plant or freshwater algal species (for non-phytotoxic substances), 3 studies (for phytotoxic substances) Toxicity data for amphibians are highly desirable, but not necessary. Data must represent fully aquatic stages. Acceptable endpoints for short-term guidance: LC/EC50 (severe effects) Acceptable endpoints for long-term guidance: Most appropriate ECx/ICx representing a low-effects threshold > EC15-25/IC15-25 > LOEC > MATC > EC26-49/IC26-49 > nonlethal EC50/IC50 > LC50. Note: only primary data are acceptable. Only short-term studies for short-term guidance, and long-term for long-term.
Type B2 Guideline	Toxicity tests required for the generation of a Type B2 guideline, broken out as follows: Fish: 2 short-term or long-term studies on two or more fish species, including 1 salmonid, 1 non-salmonid. Invertebrates: 2 short-term or long-term studies on 2 or more invertebrate species from different classes, including 1 planktonic sp. Plants: For short-term guidance: none (for non-phytotoxic substances), 2 (for phytotoxic substances) For long-term guidance: none (for non-phytotoxic substances), 2 (for phytotoxic substances) Acceptable endpoints for short-term guidance: LC/EC50 (severe effects) Acceptable endpoints for long-term guidance: Most appropriate ECx/ICx representing a low-effects threshold > EC15-25/IC15-25 > LOEC > MATC > EC26-49/IC26-49 > nonlethal EC50/IC50 > LC50. Note: primary or secondary data are acceptable. Only short-term studies for short-term guidance, and short or long-term for long-term guidance.

Table 9.2 Minimum data set requirements for the generation of a short-term marine benchmark concentration and a long-term marine CWQG following the 2007 CCME guideline protocol (CCME 2007).

Derivation Method	Minimum Toxicity Dataset
Type A Guideline	Toxicity tests required for the generation of an SSD, broken out as follows: Fish: 3 studies on 3 different species including 1 temperate species. Invertebrates: 2 studies on 2 different species from different classes including 1 temperate species. Plant/Algae: For short-term guidance: 1 study on a temperate marine vascular plant or algal species (for non-phytotoxic substances), 2 studies (for phytotoxic substances). For long-term guidance: 1 study on a temperate marine vascular plant or algal species (for non-phytotoxic substances), 3 studies (for phytotoxic substances) Acceptable endpoints for short-term guidance: LC/EC50 (severe effects) Acceptable endpoints for long-term guidance: Most appropriate ECx/ICx representing a no-effects threshold > EC10/IC10 > EC11-25/IC11-25 > MATC > NOEC > LOEC > EC26-49/IC26-49 > nonlethal EC50/IC50. Note: Primary or secondary no- and low-effects data are acceptable to meet the minimum data requirements.
Type B1 Guideline	Toxicity tests required for the generation of a Type B1 guideline, broken out as follows: Fish: 3 studies on 3 different species including 1 temperate species. Invertebrates: 2 studies on 2 different species from different classes including 1 temperate species. Plant/Algae: 1 study on a temperate marine vascular plant or algal species (for non-phytotoxic substances), 2 studies (for phytotoxic substances). Acceptable endpoints for short-term guidance: LC/EC50 (severe effects) Acceptable endpoints for long-term guidance: Most appropriate ECx/ICx representing a low-effects threshold > EC15-25/IC15-25 > LOEC > MATC > EC26-49/IC26-49 > nonlethal EC50/IC50 > LC50. Note: only primary data are acceptable to meet the minimum data requirements. The value used to set the guideline must be primary. Only short-term studies for short-term guidance, and long-term for long-term.

Derivation Method	Minimum Toxicity Dataset
Type B2 Guideline	Toxicity tests required for the generation of a Type B2 guideline, broken out as follows: Fish: 2 studies on 2 different species including 1 temperate species. Invertebrates: 2 studies on 2 different species. Plants: For short-term guidance: data for marine plants desirable but not necessary (for non-phytotoxic substances), 2 studies (for phytotoxic substances) For long-term guidance: none (for non-phytotoxic substances), 2 studies (for phytotoxic substances) Acceptable endpoints for short-term guidance: LC/EC50 (severe effects) Acceptable endpoints for long-term guidance: Most appropriate ECx/ICx representing a low-effects threshold > EC15-25/IC15-25 > LOEC > MATC > EC26-49/IC26-49 > nonlethal EC50/IC50 > LC50. Note: primary or secondary data are acceptable. The value used to set the guideline must be secondary. Only short-term studies for short-term guidance, and short or long-term for long-term guidance.

The statistical approach (which is the preferable method if the minimum data requirements are attained) involves the use of species sensitivity distributions (SSDs) which represent the variation in sensitivity of species to a substance by a statistical or empirical distribution function of responses for a sample of species. The basic assumption of the SSD concept is that the sensitivities of a set of species can be described by some distribution, usually a parametric sigmoidal cumulative distribution function. The data points used in the SSD are most commonly those derived from laboratory-based studies. Emphasis is placed on plotting organism-level effects, such as survival, growth, and reproduction, which can be more confidently used to predict ecologically-significant consequences at the population level (Meador 2000; Forbes and Calow 1999; Suter *et al*., 2005). However, the CCME (2007) protocol does state that 'non-traditional' endpoints can be used (e.g., behaviour [predator avoidance, fitness, swimming speed, etc.], physiological changes), but only if the ecological relevance of these 'non-traditional' endpoints can be demonstrated. Therefore, another assumption of the SSD is that the distribution of sensitivities of laboratory species to a substance reflects the sensitivity of species in natural aquatic environments to that same substance. The SSD method involves modelling the cumulative SSD and estimating the 95% confidence interval. The guideline is defined as the intercept of the 5^{th} percentile of the species sensitivity distribution (CCME, 2007). CCME (2007) states that no effect (e.g. EC/IC10, NOEC) data are to be used primarily, with low effect (e.g. EC/IC25, LOEC) data being less preferable, but still acceptable if no-effect data is unavailable, for guideline derivation. By using mostly no- and some low-effect data, and setting the guideline value as the 5th percentile, this guideline is expected to maintain aquatic community structure and function. SSD derived guidelines are referred to as Type A guidelines. The use of SSDs has become common in ecological risk assessment. SSDs are also used in the development of environmental quality guidelines within the European Union, Australia and New Zealand as well as the USA. Each jurisdiction has developed its own protocol (policies) with respect to WQC development using an SSD (e.g. some use only no effect data, some apply safety factors to the HC5 value, some may plot multiple endpoints for one species, some only plot NOEC survival data, etc), and therefore the approaches used are not completely identical between jurisdictions. In the case of chloride, suitable short-term and long-term datasets were provided for the development of a Type A guideline. Freshwater SSDs for freshwater biota were derived for both exposure durations following the CCME Protocol for the Derivation of Water Quality Guidelines for the Protection of Aquatic Life (CCME, 2007).

To generate the short-term and long-term SSDs, only toxicity data classified as either primary or secondary were included; datapoints classified as unacceptable were excluded. When multiple data points for effects (*e.g.*, growth, mortality, reproduction) were available for the same species professional judgment was utilized to select a representative species effect concentration (*e.g.*, lowest value or geomean). Only one endpoint per species was plotted on the SSD. Using a customized Microsoft Excel-based software package, SSD Master Version 2.0 (Rodney *et al*., 2008), a total of five cumulative distribution functions (Normal, Logistic, Gompertz, Weibull, Fisher-Tippett) were fit to the data using regression techniques. Model fit was assessed using statistical and graphical techniques. The best model was selected based on goodness-of-fit and model feasibility. Model assumptions were verified graphically. The concentration of chloride in freshwater at which 5% of species are predicted to be affected was determined for both short-term and long-term scenarios with 95% confidence intervals on the mean (expected) value.

Each species for which appropriate toxicity data were available was ranked according to sensitivity (from lowest to highest value), and its centralized position on the SSD (Hazen plotting position) was determined using the following standard equation (Aldenberg *et al.*, 2002; Newman *et al.*, 2002):

$$\text{Hazen Plotting Position} = \frac{i - 0.5}{N}$$

where:

i = the species rank based on ascending toxicity values
N = the total number of species included in the SSD derivation

9.1.1 Summary of Existing Water Quality Guidelines for the Protection of Freshwater Aquatic Life

Currently there is no health-based guideline for chloride in drinking water in Canada. An aesthetic objective of ≤250 mg/L for chloride in drinking water has been established by the Federal–Provincial Subcommittee on Drinking Water based on Health Canada recommendations, and this is also endorsed by the World Health Organization (CCME 1999; Health Canada 1987; WHO 2003). Chloride concentrations above this objective can give rise to undesirable tastes in water and beverages prepared from water, and may cause corrosion in water distribution systems (Health Canada 1987). No guideline exists for chloride to protect recreational water use.

CCME (1999) recommends a quality guideline for irrigation ranging from 100 to 700 mg chloride/L, where 100 mg chloride/L is recommended for chloride-sensitive plants, and up to 700 mg/L for chloride-tolerant plants. The BC Ministry of the Environment adopted the lower of the two guidelines for crop irrigation (Nagpal *et al*., 2003).

The BC Ministry of the Environment adopted a water quality guideline of 600 mg chloride/L for livestock watering and for waters utilized by wildlife, assuming that wildlife species would not be more sensitive than livestock to the effects of chloride (Nagpal *et al*., 2003). This guideline was calculated based on a CCME (1999) threshold of 1,000 mg/L for total soluble salts in water for livestock watering, and assuming that chloride represents 60% by weight of total soluble salts.

The British Columbia Water Protection Section of the Ministry of Water, Land and Air Protection derived guidelines for freshwater aquatic life based on studies summarized by Evans and Frick (2001) and Bright and Addison (2002) (Nagpal *et al*., 2003). The maximum chloride concentration for acute exposures is 600 mg/L (as NaCl), and this is based on a 96 hour EC50 of 1,204 mg/L for the tubificid worm, *Tubifex tubifex* (Khangarot, 1991) with the application of a safety factor of 2 based on the relative strength of the acute dataset. This study had the lowest toxicity value in a 96 hour exposure among thirteen studies with fish, seven with cladocerans, and eight with other invertebrates. The BC MOE has recommended that the chloride concentration in freshwater not exceed 150 mg/L for the protection of aquatic life from chronic effects. This was based on the lowest LOEC from a chronic toxicity test selected from nine different taxa, reporting a 50% reduction in reproduction over 7 days at 735 mg/L for *Ceriodaphnia dubia* (DeGraeve *et al*., 1992), with the application of a safety factor of 5. The latter toxicity value was an average concentration based on 14 separate trials in the study, involving many different laboratories. The safety factor was selected based on a study by Diamond *et al*., (1992) that reported a LOEC/NOEC ratio of 3.75 for reproduction of *Ceriodaphnia dubia* in a 7 day exposure (NaCl). Also taken into consideration were LC50/LC0 and LC100/LC0 ratios of 3 and 4, respectively, from Hughes (1973), as well as LC50/NOEC ratios ranging from 1.0 to 6.9 in DeGraeve *et al*., (1992). Chronic data from the reviewed literature were scant, and Nagpal *et al*., (2003) selected the safety factor to provide additional protection for potentially sensitive species that have not yet been tested.

The US EPA has established acute (1 hour average) and chronic (4 day average) freshwater National Ambient Water Quality Criteria (NAWQC) for chloride of 860 and 230 mg/L, respectively, which are not to be exceeded more than once every three years (US EPA, 1988; 2006). The acute value was derived from the Final Acute Value (FAV) of 1,720 mg/L divided by 2. Insufficient long-term data were available to derive the chronic value directly from chronic data. The chronic criterion was derived by dividing the FAV by an Acute to Chronic Ratio (ACR) of 7.59. This ACR was based on the geometric mean of ACR values from tests with the rainbow trout (7.308), the fathead minnow (15.17) and the water flea *Daphnia pulex* (3.951). Applying an ACR of 7.59 indicates that the chronic criterion should be 7.59 times lower than the acute criterion.

Evans and Frick (2001) conducted a review of chloride toxicity to aquatic organisms as part of the *Canadian Environmental Protection Act*, 1999, (CEPA 1999) Priority Substances List assessment of road salts in order to evaluate the risk of chloride to aquatic communities. Acute toxicity test data was collected and converted to chronic values by applying an ACR of 7.59, as used by the US EPA in the 1988 chloride guideline derivation. This chronic data was fitted to an SSD, from which it was estimated that 10% of aquatic species would be impacted from long-term exposures to 240 mg chloride/L (with 95% confidence limits of <194 to 295 mg/L).

In Kentucky, recommendations to protect warm water species specified that average and maximum chloride concentrations may not exceed 600 mg/L and 1,200 mg/L, respectively, for any consecutive 3 day period, but concentrations may average between the latter two values for up to 48 hours (Birge *et al.*, 1985). The value of 1,200 mg/L was based on an assessment of benthic community structure and fish survivorship at 7 sites located downstream of a salt seepage. Benthic community diversity and fish survivorship was reduced at sites where chloride measured 1,000 and 3,160 mg/L, when compared to sites where chloride measured 100 mg/L.

A site-specific water quality objective has been proposed for the EKATI Diamond Mine in the Northwest Territories, Canada (Rescan, 2008). New chronic toxicological data was obtained for the purposes of guideline derivation (Rescan, 2007; Elphick *et al.*, 2010). An SSD was used to derive an HC5 value of 325 mg/L (95% CI 269-377) at a water hardness of 80 mg/L as $CaCO_3$. Seven day survival and reproduction tests with *Ceriodaphnia dubia* demonstrated a decrease in chloride toxicity with increasing hardness, when hardness ranged from 10 to 160 mg/L as $CaCO_3$. Additional reductions in toxicity were not observed with hardness exceeded 160 mg/L as $CaCO_3$. The final proposed site specific water quality objective is to be calculated using the following hardness adjustment equation, calibrated for hardness levels ranging from 10 to 160 mg/L as $CaCO_3$: WQO = 124 x ln(hardness) – 218. It must be noted that the top tail of the SSD is dominated with high threshold toxicity algal data published by Kessler (1974). The CCME Technical Secretariat (Environment Canada) was consulted on this paper (published in German) and it was decided that this paper is not a toxicology paper, but rather related to taxonomy, with the development of a method for identification of different algal species based on salt tolerance – only those algal species identified as most sensitive were included in the dataset for CWQG derivation.

The state of Iowa is in the process of developing a chloride water quality criteria for the protection of aquatic life, which will be adjusted for total hardness and sulphate. The Iowa Department of Natural Resources (DNR) updated the criteria for chloride based on new toxicity data collected since the derivation of the 1988 US EPA guideline values. Publicly available proposed criteria (from an update of March 2009 published on the Iowa DNR website) were normalized for a hardness value of 200 mg/L and a sulphate value of 63 mg/L.

The acute criteria value (CMC) for chloride proposed in the March 2009 update was:
Acute Criteria Value (mg/L) = $287.8(\text{Hardness})^{0.205797}(\text{Sulphate})^{-0.07452}$
Acute Criteria Value (mg/L) = 287.8(200 mg/L)0.205797(63 mg/L)-0.07452

Acute Criteria Value (mg/L) = 629 mg/L Chloride

The chronic criteria value (CCC) for chloride proposed in the March 2009 update was:
Chronic Criteria Value (mg/L) = $177.87(\text{Hardness})^{0.205797}(\text{Sulphate})^{-0.07452}$
Chronic Criteria Value (mg/L) = 177.87(200 mg/L)0.205797(63 mg/L)-0.07452
Chronic Criteria Value (mg/L) = 389 mg/L Chloride

Based on the March 2009, Tables 9.3 and 9.4 provide the proposed acute and chronic chloride criteria, respectively, at various concentrations of hardness and sulphate.

The Iowa Department of Natural Resources has provided an update to the proposed criteria as of May 2009 (C.Stephan, US EPA, 2009, pers.comm.). The May 2009 proposed equations to derive CMC and CCC WQC are the following. The CMC and CCC are for hardness = 300 mg/L and sulphate = 65 mg/L.

The resulting equations for the CMC and CCC are:

$$\text{CMC} = (682.0 \text{ mg chloride/L}) (\text{hardness}/300)^{0.205797} (\text{sulphate}/65)^{-0.07452}$$
$$= (287.8 \text{ mg chloride/L}) (\text{hardness})^{0.205797} (\text{sulphate})^{-0.07452}$$

At hardness = 300 mg/L and sulphate = 65 mg/L, CMC = 682.0 mg chloride/L.

CCC = (428.0 mg chloride/L) (hardness/300)0.205797 (sulphate/65)-0.07452
= (180.6 mg chloride/L) (hardness)0.205797 (sulphate)-0.07452

At hardness = 300 mg/L and sulphate = 65 mg/L, CCC = 428.0 mg chloride/L.

Table 9.3 Proposed acute chloride criteria for state of Iowa at varying hardness (mg/L) and sulphate (mg/L) concentrations.

Sulfate	Hardness (as CaCO3)												
	50	100	150	200	250	300	350	400	450	500	600	700	800
5	571	659	716	760	795	826	852	876	897	917	952	983	1010
10	542	625	680	721	755	784	809	832	852	871	904	933	959
15	526	607	660	700	733	761	785	807	827	845	877	906	931
20	515	594	646	685	717	745	769	790	809	827	859	886	911
25	506	584	635	674	705	732	756	777	796	813	845	872	896
50	481	555	603	640	670	695	718	738	756	773	802	828	851
100	457	527	573	608	636	660	682	701	718	734	762	786	808
150	443	511	556	589	617	641	661	680	697	712	739	763	784
200	434	500	544	577	604	627	647	665	682	697	723	747	767
250	427	492	535	567	594	617	637	654	671	685	711	734	755
300	421	485	528	560	586	609	628	646	661	676	702	724	745
350	416	480	522	553	579	602	621	638	654	668	694	716	736
400	412	475	516	548	574	596	615	632	647	662	687	709	729
450	408	471	512	543	569	590	609	626	642	656	681	703	722
500	405	467	508	539	564	586	605	622	637	651	676	697	717

Table 9.4 Proposed chronic chloride criteria for state of Iowa at varying hardness (mg/L) and sulphate (mg/L) concentrations.

Sulfate	Hardness (as CaCO3)												
	50	100	150	200	250	300	350	400	450	500	600	700	800
5	353	407	442	469	491	510	527	541	555	567	589	607	624
10	335	387	420	446	467	485	500	514	527	538	559	577	593
15	325	375	408	433	453	470	485	499	511	522	542	560	575
20	318	367	399	423	443	460	475	488	500	511	531	548	563
25	313	361	392	416	436	453	467	480	492	503	522	539	554
50	297	343	373	395	414	430	444	456	467	477	496	512	526
100	282	326	354	375	393	408	421	433	444	453	471	486	499
150	274	316	343	364	381	396	409	420	430	440	457	471	485
200	268	309	336	357	373	388	400	411	421	431	447	461	474
250	264	304	331	351	367	381	394	404	414	423	440	454	467
300	260	300	326	346	362	376	388	399	409	418	434	448	460
350	257	297	322	342	358	372	384	394	404	413	429	443	455
400	255	294	319	339	355	368	380	391	400	409	425	438	450
450	252	291	316	336	351	365	377	387	397	405	421	434	447
500	250	289	314	333	349	362	374	384	394	402	418	431	443

9.1.2 Evaluation of Toxicological Data

In accordance with the CCME protocol for the derivation of water quality guidelines for the protection of aquatic life, toxicity studies were classified as primary, secondary or unacceptable (CCME 2007). Primary and secondary studies were considered for guideline development. In general, primary toxicity studies involve acceptable test procedures, conditions, and controls, measured toxicant concentrations, and flow-through or renewal exposure conditions. Secondary toxicity studies usually involve unmeasured toxicant concentrations, static bioassay conditions and unsatisfactory reporting of experimental data. Unacceptable data are deemed not suitable for guideline development (e.g. no reporting of controls, test temperature too high to be relevant to Canadian surface waters, test organism not representative of a temperate species, etc.). Studies using distilled and/or deionized water to hold test organisms were not included due to potential ionic influences on survival. Studies using species resident to Canadian waters or temperate non-native species were preferentially included in the freshwater guideline derivation as per the CCME (2007) protocol. Only toxicity data for sodium chloride and calcium chloride were used in deriving the freshwater guidelines.

9.1.3 Freshwater Aquatic Life Guideline Derivation

The Protocol for the Deriviation of Canadian Water Quality Guidelines includes a guideline value for long-term exposure and a benchmark concentration for short-term exposure (CCME 2007). The long-term exposure guideline is designed to protect all species at all life stages over an indefinite exposure to a substance in water. Continuous releases may occur from point or non-point sources, gradual release from soils/sediments and gradual entry through groundwater/runoff, and long-range transport. The short-term benchmark concentration value does *not* provide guidance on protective levels of a substance in the aquatic environment, as short-term benchmark concentrations are levels which *do not* protect against adverse effects, but rather indicate the level where severe effects are likely to be observed.

While separate data sets are used to calculate short-term benchmark concentrations and long-term guidelines, both are derived using either a statistical approach without the application of a safey factor (Type A or Species Sensitivity Distribution), or one of two assessment factor approaches. The first assessment factor approach (Type B1) applies a safety factor to the lowest endpoint from a primary study, and the second approach (Type B2) applies a safety factor to the lowest endpoint from a primary and/or secondary study. The three approaches are detailed in CCME (2007).

All toxicity data for freshwater organisms can be found in appendix A. For the derivation of the short-term benchmark concentration and the long-term CWQG for the chloride ion, this list was pared down to include data only from studies classified as primary or secondary following CCME (2007).

9.1.4 Derivation of the Short-term Benchmark Concentration

Short-term benchmark concentrations are derived using severe effects data (such as lethality) of defined short-term (e.g. 24 or 96 hour) exposure periods (see CCME 2007 for exposure period definitions). These benchmark concentrations are estimators of severe effects to the aquatic ecosystem and are intended to give guidance on the impacts of severe, but transient, situations (e.g., spill events to aquatic receiving environments and infrequent releases of short-lived/nonpersistent substances). Short-term benchmark concentrations *do not* provide guidance on protective levels of a substance in the aquatic environment, as short-term benchmark concentrations are levels which *do not* protect against adverse effects, but rather indicate the level where severe effects are likely to be observed.

The minimum data requirements for the development of a short-term Type A (SSD-derived) benchmark concentration were met, and these are listed in Table 9.1.

A total of 51 data points (14 of which were EC50 values with the remainder being LC50 values) were used in the derivation of the short-term benchmark concentration (Table 9.5). These 51 data points were retrieved from toxicity studies meeting the requirements for primary or secondary data, according to CCME (2007) protocol. Intra-species variability was accounted for by taking the geometric mean of the studies considered to represent the most sensitive lifestage and endpoint. Each data point was ranked according to sensitivity, and its centralized distribution on the species sensitivity distribution (SSD) was determined using the Hazen plotting position (estimate of the cumulative probability of a data point). The plotting positions are treated as observed proportions of species affected. These positional rankings, along with their corresponding LC/EC50 values, were used to derive the SSDs.

The values reported in Table 9.5 range from a 24 hour EC50 of 244 mg/L for the glochidia life stage of the COSEWIC endangered Northern Riffleshell mussel, *Epioblasma torulosa rangiana (*Gillis 2011), to a 48 hour LC50 of 12,385 mg/L for the copepod *Cyclops abyssorum prealpinus* (Baudouin and Scoppa 1974). Multiple bioassay results for the same species should not be used in an SSD regression analysis. This is particularly important when there is a large amount of data available for very few test species. There are numerous methods that can be applied to account for multiple results for a single species (Duboudin et al., 2004). For the derivation of a short-term benchmark concentration for chloride, intra-species variability was accounted for by taking the geometric mean of the studies considered to represent the most sensitive life stage and endpoint. The geometric means, in these cases, where taken for like species, life stage and endpoint. Geometric mean values were calculated for *Lampsilis siliquoidea*, *Lampsilis fasciola* (COSEWIC special concern), *Sphaerium simile, Ceriodaphnia dubia*, *Daphnia pulex, Villosa iris* (COSEWIC endangered)*, Brachionus calyciflorus, Lithibates sylvatica* (previously *Rana sylvatica*)*, Gyraulus parvus, Baetis tricaudatus, Pimephales promelas, Lumbriculus variegates, Tubifex tubifex,* and *Oncorhynchus mykiss* (Table 9.6). Effect concentrations reported for the remaining species were taken from single studies.

Table 9.5 Short-term LC/EC50s for species exposed to chloride in freshwater. See Table 9.6 for grouped data.

Rank	Scientific Name	Common Name	Endpoint	LC/EC50 (mg Cl^- /L)	Data Quality	Hazen Plotting Position	Reference
1	*Epioblasma torulosa rangiana*[a]	Northern Riffleshell Mussel (glochidia)	24h EC50 (survival of glochidia)	244	S	0.01	Gillis 2011
2	*Daphnia magna*	Water flea (<24h old)	48h EC50 (immobilization)	621	S	0.03	Khangarot and Ray 1989
3	*Lampsilis siliquoidea*	Fatmucket mussel (glochidia)	24h EC50 (survival of glochidia)	709	S Grouped*	0.05	
4	*Lampsilis fasciola*[b]	Wavy-rayed Lampmussel (glochidia)	24h EC50 (survival of glochidia)	746	S Grouped*	0.07	
5	*Lampsilis cardium*	Plain pocketbook	24h EC50 (survival of glochidia)	817	S	0.09	Gillis 2011
6	*Sphaerium simile*	Fingernail clam (juveniles, 4.5-6.5 mm)	96h LC50	902	P Grouped*	0.11	
7	*Ceriodaphnia dubia*	Water flea (neonates, <24 hr old)	48h LC50	1,080	S/S/S/P /P/S Grouped*	0.13	
8	*Ambystoma maculatum*	Spotted salamander (larvae, Gosner stage 25)	96h LC50	1,178	S	0.15	Collins and Russell 2009
9	*Daphnia ambigua*	Water flea (neonates, <24 hr old)	48h EC50 (Immobilization)	1,213	S	0.17	Harmon *et al*., 2003
10	*Daphnia pulex*	Water flea	48h LC50	1,248	S Grouped*	0.19	
11	*Elliptio lanceolata*	Yellow lance mussel (10d old)	96h LC50	1,274	S	0.21	Wang and Ingersoll 2010
12	*Brachionus patulus*	Rotifer (neonate)	24h LC50	1,298	S	0.23	Peredo-Alvarez *et al*., 2003
13	*Hyalella azteca*	Amphipod (7-8d old)	96 h LC50	1,382	P	0.25	Elphick *et al*., 2010
14	*Elliptio complananta*	Freshwater mussel (glochidia)	24h EC50 (survival of glochidia)	1,620	S	0.26	Bringolf *et al*., 2007
15	*Epioblasma brevidens*	Cumberlandian combshell (endangered in	24h EC50 (survival of glochidia)	1,626	S	0.28	Valenti *et al*., 2007

Rank	Scientific Name	Common Name	Endpoint	LC/EC50 (mg Cl^- /L)	Data Quality	Hazen Plotting Position	Reference
		USA) (glochidia)					
16	*Epioblasma capsaeformis*	Oyster mussel (endangered in USA) (glochidia)	24h EC50 (survival of glochidia)	1,644	S	0.30	Valenti *et al*., 2007
17	*Villosa constricta*	Freshwater mussel (glochidia)	24h EC50 (survival of glochidia)	1,674	S	0.32	Bringolf *et al*., 2007
18	*Villosa iris*[a]	Rainbow mussel (2 months old)	96h EC50	1,815	S Grouped*	0.34	
19	*Musculium transversum*	Fingernail clam (juvenile)	96h LC50	1,930	S	0.36	US EPA 2010
20	*Villosa delumbis*	Freshwater mussel (glochidia)	24h EC50 (survival of glochidia)	2,008	S	0.38	Bringolf *et al*., 2007
21	*Brachionus calyciflorus*	Rotifer (neonate, <4h old)	24 h LC50	2,026	P Grouped*	0.40	
22	*Pseudacris triseriata feriarum*	Chorus frog (72h post hatch)	96h LC50	2,320	S	0.42	Garibay and Hall 2004
23	*Physa gyrina*	Snail	96h LC50	2,540	S	0.44	Birge *et al*., 1985
24	*Lithibates sylvatica* (previously *Rana sylvatica*)	Wood frog (Gosner stage 25)	96h LC50	2,716	S Grouped*	0.46	
25	*Pseudacris crucifer*	Spring peeper (Gosner stage 25)	96h LC50	2,830	S	0.48	Collins and Russell 2009
26	*Lirceus fontinalis*	Isopod	96h LC50	2,950	S	0.50	Birge *et al*., 1985
27	*Gyraulus parvus*	Snail (mixed ages, 3-5mm)	96h LC50	3,043	P Grouped*	0.52	
28	*Rana clamitans*	Green frog (Gosner stage 25)	96h LC50	3,109	S	0.54	Collins and Russell 2009
29	*Rana temporaria*	Common frog	96h LC47.6	3,140	S	0.56	Viertel 1999
30	*Baetis tricaudatus*	Mayfly	48h EC50 (Immobility)	3,266	S Grouped*	0.58	
31	*Lithibates pipiens* (previously *Rana pipiens*)	Leopard frog	96h LC50	3,385	S	0.60	Doe 2010
32	*Chironomus dilutus / tentans*	Chironomid	96h LC50	3,761	S	0.62	Wang and Ingersoll 2010
33	*Bufo americanus*	American toad	96h LC50	3,926	S	0.64	Collins and Russell 2009
34	*Lumbriculus*	Oligochaete	96h LC50	4,094	P	0.66	

Rank	Scientific Name	Common Name	Endpoint	LC/EC50 (mg Cl^- /L)	Data Quality	Hazen Plotting Position	Reference
	variegatus				Grouped*		
35	*Pimephales promelas*	Fathead minnow	96h LC50	4,223	S Grouped*	0.68	
36	*Nephelopsis obscura*	Leech	96h LC50	4,310	P	0.70	Environ 2009
37	*Hexagenia* spp.	Mayfly	48h LC50	4,671	S	0.72	Wang and Ingersoll 2010
38	*Chironomus attenatus*	Chironomid	48h LC50	4,850	S	0.74	Thornton and Sauer 1972
39	*Lepomis macrochirus*	Bluegill sunfish	96h LC50	5,272	S Grouped*	0.75	
40	*Daphnia hyalina*[c]	Water flea (adult avg length of 1.27 mm)	48h LC50	5,308	S	0.77	Baudouin and Scoppa 1974
41	*Rana catesbeiana*	Bullfrog	96h LC50	5,846	P	0.79	Environ 2009
42	*Lepidostoma* spp.	Caddisfly	96h LC50	6,000	S	0.81	Williams *et al*., 1999
43	*Cyprinella leedsi*	Bannerfin Shiner	96h LC50	6,070	P	0.83	Environ 2009
44	*Tubifex tubifex*	Oligochaete	96h LC50	6,119	P/S/P Grouped*	0.85	
45	*Chironomus riparius*	Chironomid	48h LC50	6,912	S	0.87	Wang and Ingersoll 2010
46	*Eudiaptomus padanus padanus*[c]	Copepod	48h LC50	7,077	S	0.89	Baudouin and Scoppa 1974
47	*Oncorhynchus mykiss*	Rainbow trout	96h LC50	8,634	P/S Grouped*	0.91	
48	*Gambusia affinis*	Mosquito-fish	96h LC50	9,099	S	0.93	Al-Daham and Bhatti 1977
49	*Gasterosteus aculeatus*	Threespine stickelback	96h EC50	10,200	S	0.95	Garibay and Hall 2004
50	*Cyclops abyssorum prealpinus*[c]	Copepod	48h LC50	12,385	S	0.97	Baudouin and Scoppa 1974
51	*Anguilla rostrata*	American eel	96h LC50	13,012	S	0.99	Hinton and Eversol 1979

[a]Status – Endangered - as designated by COSEWIC.
[b]Status - Special Concern - as designated by COSEWIC.
[c]Based on testing with $CaCl_2$ salt (all others based on testing with NaCl salt).
Data Quality:
S = Secondary; P = Primary
Grouped: Indicates that the geomean of multiple values was used to calculate the effect concentration
*value shown is the geometric mean of comparable values, individual values and references can be seen in Table 9.6.

Table 9.6 Studies used to derive geometric means for short-term data in Table 9.5.

Organism	Endpoint	Effect Concentration (mg/L)	Geometric Mean (mg/L)	Reference
Lampsilis siliquoidea (Fatmucket mussel)	24h EC50	**334**	709	Bringolf *et al*., 2007
	24h EC50 24h EC50 24h EC50 24h EC50	1,962 1,870 1,430 763 Mean[a] = **1,506**		Gillis 2011
Lampsilis fasciola (Wavy-rayed Lampmussel)	24h EC50	**1,868**	746	Valenti *et al*., 2007
	24h EC50	**1,116**		Bringolf *et al*., 2007
	24h EC50 24h EC50	113 285 Mean[a] = **199**		Gillis 2011
Sphaerium simile (fingernail clam)	96h LC50 96h LC50	740 1,100	902	GLEC and INHS 2008
Ceriodaphnia dubia (water flea)	48h LC50	**1,413**	1,080	Valenti *et al*., 2007
	48h LC50 48h LC50	507 447 Mean[a] = **477**		Hoke *et al*., 1992
	48h LC50	1,169[b]		Mount *et al*., 1997
	48h LC50 48h LC50 48h LC50 48h LC50 48h LC50 48h LC50 48h LC50 48h LC50 48h LC50 48h LC50 48h LC50 48h LC50 48h LC50 48h LC50 48h LC50 48h LC50 48h LC50 48h LC50	947 955 1,130 1,609 1,491 1,907 1,764 1,007 767 1,369 1,195 1,687 1,652 1,909 1,400 1,720 1,394 1,500		GLEC & INHS 2008

Organism	Endpoint	Effect Concentration (mg/L)	Geometric Mean (mg/L)	Reference
	48h LC50 48h LC50 48h LC50 48h LC50 48h LC50 48h LC50 48h LC50 48h LC50	1,109 1,206 1,311 1,258 1,240 1,214 1,199 1,179 Mean[a] = **1,351**		
	48h LC50	**1,068**		Elphick *et al*., 2011
	48h LC50	**1,395**		Cowgill and Milazzo 1990
Daphnia pulex (water flea)	48h LC50 48h LC50 48h LC50 48h LC50	1,159 1,775 1,805 2,242 Mean[a] = **1,745**	1,248	Palmer *et al*., 2004
	48h LC50	**892**		Birge *et al*., 1985
Villosa iris (Rainbow mussel)	96h EC50 96h EC50 96h EC50 96h EC50 96h EC50	1,517 1,638 2,244 1,820 1,941	1,815	Wang and Ingersoll 2010
Brachionus calyciflorus (rotifer)	24h LC50 24h LC50 24h LC50	1,645 2,275 2,223	2,026	Elphick *et al*., 2011; Peredo-Alvarez *et al*., 2003; Calleja *et al*., 1994
Lithibates sylvatica (previously *Rana sylvativa*) (wood frog)	96h LC50 96h LC50 96h LC50	1,721 3,099 3,755	2,716	Collins and Russell 2009; Sanzo and Hecnar 2006; Jackman 2010
Gyraulus parvus (snail)	96h LC50 96h LC50	3,078 3,009	3,043	GLEC and INHS 2008
Baetis tricaudatus (mayfly)	48h EC50 48h EC50	3,233 3,300	3,266	Lowell *et al*., 1995
Lumbriculus variegates (oligochaete)	96h LC50 96h LC50	3,100 5,408	4,094	Elphick *et al*., 2011; Environ 2009

Organism	Endpoint	Effect Concentration (mg/L)	Geometric Mean (mg/L)	Reference
Pimephales promelas (fathead minnow)	96h LC50 96h LC50 96h LC50	2,958[b] 3,876 6,570	4,223	Mount *et al* 1997; Mount *et al* 1997; Birge *et al* 1985
Lepomis macrochirus (bluegill sunfish)	96h LC50 96h LC50	3,543 7,846	5,272	Birge *et al* 1985; Trama 1954
Tubifex tubifex (oligochaete)	96h LC50	**5,648**	6,119	Elphick *et al* 2011
	96h LC50	**7,886**		Wang and Ingersoll 2010
	96h LC50 96h LC50	4,278 6,008 Mean[a] = **5,143**		GLEC and INHS 2008
Oncorhynchus mykiss (rainbow trout)	96h LC50 96h LC50	6,030 12,363	8,634	Elphick *et al* 2011; Vosyliene *et al*., 2006

[a]To reduce bias towards any one study reporting multiple LC50 effect concentrations, an average LC50 value has been calculated for each study reporting multiple LC50 values and is subsequently used to calculate the geometric mean for the organism.
[b]Based on testing with $CaCl_2$ salt (all others based on testing with NaCl salt).

One item to note in Table 9.6 is the range in 24h LC50 values reported for the COSEWIC designated special concern wavy-rayed lampmussel *Lampsilis fasciola*. Gillis (2011) collected gravid *Lampsilis fasciola* (wavy-rayed lampmussel) from the same site (Grand River, ON) on two different occasions (2008 & 2009), producing somewhat similar glochidia 24h EC50s of 113 (63-163) and 285 (163-451) mg Cl^-/L, respectively. The exposures were conducted in ASTM moderately hard reconstituted water (95-115 mg/L as $CaCO_3$). This provides indication that the low EC50 value is not an outlier. The ASTM standard guide for conducting laboratory toxicity tests with freshwater mussels was used. Bringolf *et al*., (2007) collected gravid *Lampsilis fasciola* from Little Tennessee River (North Carolina), with ASTM reconstituted hard water (160-180 mg/L as $CaCO_3$) used as dilution water for toxicity testing. The ASTM standard guide for conducting laboratory toxicity tests with freshwater mussels was used. Valenti *et al*., (1997) collected gravid *Lampsilis fasciola* from the Clinch River (Virginia). Moderately hard reconstituted water (100 mg/L as $CaCO_3$) was used for toxicity testing. The ASTM standard guide for conducting laboratory toxicity tests with freshwater mussels was not yet available, and so adhered to test design described in US EPA protocol (1993) for standard freshwater test organisms. One reason for the range in 24h EC50 values is that the organisms used for testing are not obtained from an established laboratory culture, but rather field-collected from various river systems. Prior exposure or even acquired tolerance may alter the response of glochidia to contaminants. Gillis (2011) observed a range in glochidia 24h EC50 values for a second freshwater mussel species, *Lampsilis siliquoidea*, collected from 2 separate water bodies. Testing conducted in ASTM moderately hard water (95-115 mg/L as $CaCO_3$) resulted in a glochidia 24h EC50 of 1,430 mg Cl^-/L for organisms collected from Maitland River (ON), whereas a glochidia 24h EC50 of 168 mg Cl^-/L resulted for organisms collected from Cox Creek (ON) [It is important to note here that for the exposure with Cox Creek collected organisms, glochidia control survival dropped by more than 10% from test start (0h) to test end (24h) and therefore did not meet control survival requirements as per ASTM (2006)]. As well, additonal testing conducted by Gillis (2011) with *L. fasciola* indicated that glochidia were significantly less sensitive to chloride when tested using natural waters (Sydenham River hardness = 292 mg $CaCO_3$/L, Grand River hardness = 278 mg $CaCO_3$/L, Maitland River hardness = 322 mg $CaCO_3$/L, Thames River hardness = 306 mg $CaCO_3$/L) versus moderately hard (95 to 115 mg $CaCO_3$/L) reconstituted water. Gillis (2011) indicated that in addition to elevated water hardness, other water chemistry factors contributed to the reduced toxicity of chloride in natural waters. A disadvantage of using data collected from toxicity tests conducted in reconstituted water is that the EC50s may not predict how an organism will respond to a particular contaminant in the natural environment, but it does allow for comparison of effects between tests.

Five cumulative distribution functions (normal, logistic, Gompertz, Weibull, and Fisher-Tippett) were fit to the data, both in arithmetic space (no transformation of LC50 values) and log space (log transformed LC50 values) using regression methods. Model fit was assessed using statistical and graphical techniques. The best model was selected based on consideration of goodness-of-fit test and model feasibility. Model assumptions were verified graphically and with the use of statistical tests.

Of the ten models tested, the log-Normal model fit the data best (Figure 9.1). The Anderson-Darling Goodness of Fit test statistic (A^2) was 0.183 (P-value >0.10). The equation of the fitted log-Normal model is:

$$f(x) = \frac{1}{2}\left(1 + erf\ \left(\frac{x-\mu}{\sigma\sqrt{2}}\right)\right)$$

Where, for the fitted model: x = log (concentration) of chloride (mg/L), y is the proportion of species affected, $\mu = 3.4390$, $\sigma = 0.3841$ and *erf* is the error function (a.k.a. the Gauss error function).

Summary statistics for the short-term SSD are presented in Table 9.7. The 5^{th} percentile on the short-term SSD is 640 mg/L which is essentially within the range of the data (to which the model was fit). Therefore the 5^{th} percentile and its fiducial limits (FL) (boundaries within which a parameter is considered to be located) are interpolations. The lower FL (5%) on the 5^{th} percentile is 605 mg/L, and the upper FL (95%) on the 5^{th} percentile is 680 mg/L. The short-term benchmark concentration is defined as the 5^{th} percentile on the SSD. **Therefore, the short-term exposure benchmark concentration indicating the potential for severe effects (e.g. lethality or immobilization) to sensitive freshwater life during transient events is 640 mg chloride/L.**

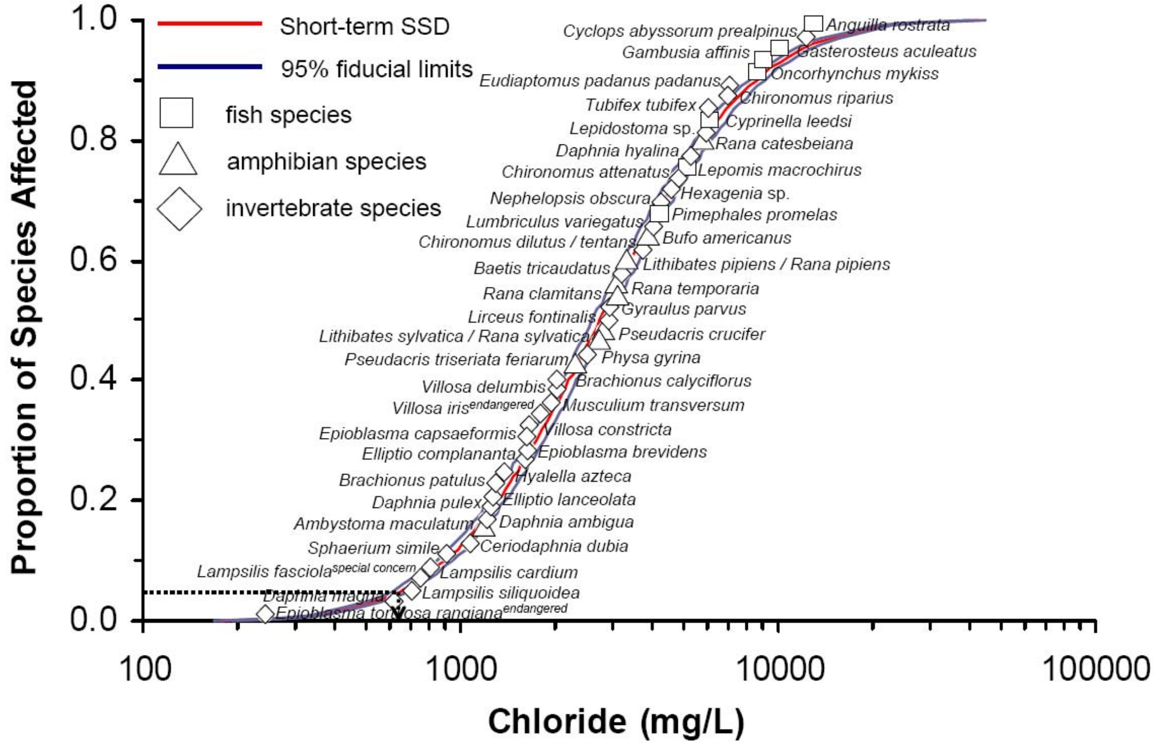

Figure 9.1 SSD of short-term L/EC50 toxicity data for the chloride ion in freshwater derived by fitting the Normal model to the logarithm of acceptable toxicity data for 51 aquatic species versus Hazen plotting position (proportion of species affected). The arrow at the bottom of the graph denotes the 5th percentile and the corresponding short-term benchmark concentration value.

Table 9.7 Short-term freshwater CWQG for the chloride ion using the SSD method.

	Concentration
SSD 5th percentile	**640 mg/L**
Lower 95% confidence limit	605 mg/L
Upper 95% confidence limit	680 mg/L

In general, the invertebrate species are grouped towards the lower end of the SSD, while the fish species are grouped towards the upper end of the SSD. This translates to invertebrates being more sensitive to acute chloride exposures when compared to fish. The amphibian species are generally grouped in the centre of the SSD, with the spotted salamander (*Ambystoma maculatum*) located closer to the lower end. Two data points fall below the short-term SSD HC5 value of 640 mg/L. These include the 24h EC50 of 244 mg Cl^-/L for the mantle lure spawning freshwater mussel (glochidia lifestage) *Epioblasma torulosa rangiana* (COSEWIC endangered) (Gillis, 2011), and the 48h EC50 (immobilization) of 621 mg Cl^-/L for the water flea *Daphnia magna* (Khangarot and Ray, 1989). Two other COSEWIC assessed species of freshwater mussels are also represented on the short-term SSD, with all data points above the 5th percentile value. This includes the glochidia 24h EC50 of 746 mg Cl^-/L for the COSEWIC special concern wavy-rayed lampmussel (*Lampsilis fasciola*) (Valenti *et al.*, 2007; Gillis, 2010; Bringolf *et al*., 2007), and the juvenile 24h EC50 of 1,815 mg Cl^-/L for the COSEWIC endangered rainbow mussel (*Villosa iris*) (Wang and Ingersoll, 2010). Both *L. fasciola* and *V. iris* are mantle lure spawners. The short-term benchmark concentration is intended for assessing the potential for severe effects following intermittent or short-lived periods of chloride exposure (e.g. spike or spill). Therefore, based on the short-term SSD, short-term exposures to levels of chloride exceeding the benchmark concentration of 640 mg Cl^-/L *may* pose the greatest hazard to the glochidia life stage of certain freshwater mussel species and to *Daphnia magna*. Note that meeting the proposed long-term guideline will protect from severe effects. Implementation of the Protection Clause does not apply in the case of short-term benchmark concentrations – it may only be applied to the long-term guideline.

It is worth noting that glochidia of the COSEWIC special concern mussel *Lampsilis fasciola* are significantly more sensitive when tested in reconstituted laboratory water compared to natural river waters. Two separate tests derived 24h EC50 values of 113 (63-163) and 285 (163-451) mg Cl^-/L for *L. fasciola* when conducted in moderately hard reconstituted water (99 mg/L as $CaCO_3$) (Gillis 2011). In comparison, the 24h EC50 values for *L. fasciola* tested in water collected from 4 different rivers in Ontario, Canada were 1,559 mg Cl^-/L (Grand River, hardness 278 mg/L as $CaCO_3$), 1,313 mg Cl^-/L (Sydenham River, hardness 292 mg/L as $CaCO_3$), 1,391 mg Cl^-/L (Maitland River, hardness 322 mg/L as $CaCO_3$) and 1,265 mg Cl^-/L (Thames River, hardness 306 mg L^{-1} as $CaCO_3$) (Gillis, 2011). The ameliorating effect of natural water chemistry was attributed to more than just a difference in water hardness. A separate test looking at the impact of water hardness on chloride toxicity was conducted with *Lampsilis siliquoidea* (Gillis, 2011). Resulting 24h EC50 values were 763, 1430, 1962 and 1870 mg Cl^-/L in soft (47 mg/L as $CaCO_3$), moderately hard (99 mg/L as $CaCO_3$), hard (172 mg/L as $CaCO_3$) and very hard (322 mg/L as $CaCO_3$) reconstituted water, respectively. The 4-fold difference in 24h EC50 values obtained for *L. fasciola* in natural river water, when compared to reconstituted water, is much larger than would be expected from hardness alone (as determined with *L. siliquoidea*), implying that other water chemistry variables are contributing to the reduction of chloride toxicity in natural waters. Short-term benchmark concentrations (as well as long-term CWQGs) are derived using laboratory-based studies which use reconstituted water to ensure consistency and the ability to compare results between studies. One disadvantage of using reconstituted waters is that results may not necessarily reflect organism responses in natural waters. Natural waters, on the other hand, contain variable water chemistry in addition to other potential contaminants resulting in variable toxic impacts to biotic receptors. Therefore, both short-term benchmark concentrations (as well as long-term CWQGs) are, by design, going to be conservative values.

Aside from the fact that the data utilized in the short-term SSD originates from laboratory studies (which are intrinsically conservative), it is also worth considering the life history of the 3 most sensitive species of freshwater mussel (*Epioblasma torulosa rangiana, Lampsilis siliquoidea, Lampsilis fasciola)*, The reproductive behaviour of these three species needs to be taken into consideration to get a better idea of the season and timing of glochidia release. Mussels are generally categorized as either tachytictic (short-term summer brooders), or bradytictic (longer-term winter brooders) (EPA, 2007). Tachytictic brooders have a shorter gestation period, where glochidia development and release occurs in April and August, respectively. Bradytictic brooders have a longer gestational period, whereby spawning occurs in the summer months, with subsequent release of glochidia occurring in late spring or early summer (EPA, 2007). With respect to the process of spawning, males discharge sperm into the water column. Females take up this sperm through their siphons during periods of feeding and respiration. The fertilized eggs are then contained within marsupia (specialized gills) that act as brood pouches for the glochidia (developing larvae). The mussel glochidia are ultimately released into the water column where they must attach to an appropriate host fish in order to develop into the juvenile life stage (EPA, 2007). Of the 3 aforementioned mussel species (*Epioblasma torulosa rangiana, Lampsilis siliquoidea, Lampsilis fasciola*), all are categorized as bradytictic longer-term winter brooders. For these 3 species, the breeding season occurs in the summer months, followed by a 10 month gestation period, with release of glochidia occurring the following spring (Michigan Natural Features Inventory 2004; Mulcrone R. 2006b; Mulcrone, R. 2006c) (Table 9.8).

Table 9.8 Overview of Life History of Species of Freshwater Mussels Included in the Short-Term Chloride Dataset.

Common Name	Scientific Name	Range of Occurrence	Conservation Status	Fish Host Species[2b]	Breeding Information	Toxicity Data Reference
Northern Riffleshell Mussel	*Epioblasma torulosa rangiana* (mantle lure spawner)	ON[1a,2]	COSEWIC Endangered[3]	Fish host species not conclusively known. Brown trout, Blackside darter, Logperch are all suspected hosts. Could also include banded darter (*Etheostoma zonale*), bluebreast darter (*E. camurum*), brown trout (*Salmo trutta*), and banded sculpin (*Cottus carolinae*) (EPA, 2007).	Gravid from late summer to the following spring, at which time the glochidia are released[6]	Gillis 2011
Fatmucket Mussel	*Lampsilis siliquoidea* (mantle lure spawner)	AB, MB, NT, ON, PQ, SK[2]	Currently stable[2]	Bass, perch, walleye, sturgeon[4]	Breeds once in the warmer months of the year. In Michigan breeding season is likely June to July, with 10 month gestation period (average)[11]	Bringolf *et al.,* 2007; Gillis 2011
Wavy-rayed Lampmussel	*Lampsilis fasciola* (mantle lure spawner)	ON[1b]	COSEWIC Special Concern[3]	Largemouth bass, smallmouth bass[5]	Breeds once in the warmer months of the year. In Michigan, breeding season is likely summer, with a 10 month gestation period (high)[12]	Bringolf *et al.,* 2007; Gillis 2011
Plain	*Lampsilis*	Upper	Currently stable	white	Spawns in	Gillis 2011

Common Name	Scientific Name	Range of Occurrence	Conservation Status	Fish Host Species[2b]	Breeding Information	Toxicity Data Reference
pocketbook	*cardium*	Mississippi and Ohio drainages; from Lake Superior to the Ottawa River and Lake Champlain[13]	(not listed)[14]	crappie, bluegill, largemouth bass, smallmouth bass, yellow perch and others[13]	late July; glochidia are relaeased the following early July[13]	
Yellow lance mussel	*Elliptio lanceolata*	US only (VA, NC and GA)[15]	Lower Risk/near threatened[16]	Has not been determined[17]	Little known about life history, gravid females have been found in spring and June[17]	Wang and Ingersoll 2010
Eastern Elliptio	*Elliptio complanata* (broadcast spawner)	NB, NS, ON, PQ[2]	Currently stable[2]	killifish, sunfish, bass, crappie, perch[8]	In Michigan, breeding season is mid-July to Auguest. Gestation period 10 months (average)[9]	Bringolf *et al.,* 2007
Notched Rainbow	*Villosa constricta* (mantle lure spawner)	US only (NC, VA)[2]	Special Concern[2]	Lab study indicated Fantail Darter served as best host; fantail darters are not present in all streams where *V. constricta* exist[10]	NA	Bringolf *et al.,* 2007
Rainbow mussel	*Villosa iris*	ON (Ausable, Bayfield, Detroit, Grand, Maitland, Moira, Niagara, Salmon, Saugeen, Sydenham, Thames and Trent rivers; Lakes Huron, Ontario, Erie and St. Clair)[18]	COSEWIC endangered[18]	Striped Shiner, Smallmouth Bass, Largemouth Bass, Green Sunfish, Greenside Darter, Rainbow Darter, Yellow Perch[18]	Spawn in late summer, release glochidia in early spring[18]	Wang and Ingersoll 2010
Cumberlandian combshell	*Epioblasma brevidens* (mantle lure spawner)	US only (AL, KY, TN, VA)	Endangered[2]	NA	NA	Valenti *et al.,* 2007

Common Name	Scientific Name	Range of Occurrence	Conservation Status	Fish Host Species[2b]	Breeding Information	Toxicity Data Reference
Oyster mussel	*Epioblasma capsaeformis* (mantle lure spawner)	US only (AL, KY, TN, VA)	Endangered[2]	NA	NA	Valenti *et al.,* 2007
Eastern Creekshell	*Villosa delumbis* (mantle lure spawner)	US only (GA, NC, SC)[2]	Currently stable[2]	5 sunfish species (bluegill, redbreast sunfish, green sunfish, warmouth, redear sunfish) all were viable hosts in the lab[10]	NA	Bringolf *et al.,* 2007

[1a] COSEWIC. 2010b.
[1b] COSEWIC. 2010a.
[2] Williams, JD, ML Warren Jr., KS Cummings, JL Harris, and RJ News. 1992. Conservation status of freshwater mussels of the United States and Canada. Fisheries. 18(9):6-22.
[2a] Morris TJ, DJ McGoldrick, JL Metcalfe-Smith, D Zanatta, and PL Gillis. 2008. Pre-COSEWIC assessment of the Wavyrayed Lampmussel (*Lampsilis fasciola*). Canadian Science Advisory Secretariat. Research Document 2008/83. (Successful reproduction will not occur in the absence of a suitable host fish)
[2b]Successful reproduction will not occur in the absence of a suitable host fish
[3]Species at risk as designated by the Committee on the Status of Endangered Wildlife in Canada (COSEWIC)
[4]Wikipedia. Accessed 13Jan10 http://en.wikipedia.org/wiki/Lampsilis_siliquoidea
[5]Morris *et al*., 2008 (Excert taken from this pre-COSEWIC assessment document: "Largemouth bass have been shown to aquire immunity to the glochidia of a closely related species, *Lampsilis siliquoidea*, after repeated infestations. This host-aquired resistance to glochidial infestation can extend across mussel genera. Individual largemouth or smallmouth bass may become less suitable as hosts with each repeated infestation, regardless of which species of mussel is first to infest them. In river reaches with severely diminished populations of largemouth and/or smallmouth bass, the competition for naïve hosts may be a significant factor limiting the reproduction of *L. fasciola*. Healthy and recruiting populations of largemouth and/or smallmouth bass are crucial "habitat" for the larval stage of *L. fasciola*.")
[6]Michigan Natural Features Inventory 2004. Status of Northern Riffleshell (*Epioblasma torulosa rangiana*). http://web4.msue.msu.edu/mnfi/abstracts/zoology/Epioblasma_torulosa_rangiana.pdf
[7]Department of Fisheries and Oceans Canada 2004 http://www.dfo-mpo.gc.ca/species-especes/species-especes/riffleshell-dysnomie-eng.htm
[8]Wisconsin Department of Natural Resources 2009 http://dnr.wi.gov/org/land/er/biodiversity/index.asp?mode=info&Grp=19&SpecCode=IMBIV14060
[9] Mulcrone, R. 2006a. "Elliptio complanata" (On-line), Animal Diversity Web. Accessed January 13, 2010 at http://animaldiversity.ummz.umich.edu/site/accounts/information/Elliptio_complanata.html
[10]North Carolina State University/ North Carolina Museum of Natural Sciences. 2007. Propagation of freshwater mussels for release into North Carolina waters. Submitted to North Carolina Department of Transportation (Project Number: HWY-2005-07) FHWA/NC/2006-37. May 2007. Accessed January 13, 2010 at http://www.ncdot.org/doh/preconstruct/tpb/research/download/2005-07FinalReport.pdf
[11] Mulcrone, R. 2006b. "Lampsilis siliquoidea" (On-line), Animal Diversity Web. Accessed January 13, 2010 at http://animaldiversity.ummz.umich.edu/site/accounts/information/Lampsilis_siliquoidea.html
[12] Mulcrone, R. 2006c. "Lampsilis fasciola" (On-line), Animal Diversity Web. Accessed January 13, 2010 at http://animaldiversity.ummz.umich.edu/site/accounts/information/Lampsilis_fasciola.html
EPA. 2007. Appendix C: Status and Life History of the Three Assessed Mussels. August 29, 2007.
[13] Amy Benson. 2011. *Lampsilis cardium*. USGS Nonindigenous Aquatic Species Database, Gainesville, FL. http://nas.er.usgs.gov/queries/factsheet.aspx?SpeciesID=2238 RevisionDate: 4/21/2004 Accessed February 28, 2011
[14] http://www.marietta.edu/~biol/mussels/planpock.html
[15] http://amylyne.myweb.uga.edu/fwmolluscs/Altamahafwm.html#Elanc
[16] http://www.iucnredlist.org/apps/redlist/details/7647/0
[17] http://www.ncwildlife.org/Wildlife_Species_Con/WSC_Mussel_16.htm
[18] http://www.dfo-mpo.gc.ca/species-especes/species-especes/rainbow-villeuseirisee-eng.htm

It would be expected that the least sensitive of all mussel species would be conglutinate spawners (e.g. *Ptychobranchus fasciolaris*, Gillis 2011, as listed in Appendix A). Conglutinates are made up of gelatinous material within which is encased large numbers of glochidia (ASTM, 2006). This gelatinous material acts as somewhat of a protective barrier between the glochidia and the surrounding water. These conglutinates resemble a fish prey item, and when a fish attempts to ingest it, the glochidia are released from the conglutinate and this is when the glochidia infest the host fish (ASTM, 2006). *P. fasciolaris* glochidia have been found to be significantly more sensitivie to copper when exposed as free glochidia (i.e. released from conglutinates) compared to glochidia that were encased in the conglutinate for the exposure (Gillis *et al*., 2008). No conglutinate spawners are represented in the short-term dataset.

Chloride concentrations measured in surface waters tend to be highest during the winter months (November to March), during the period of road salt application. A report by Kilgour *et al*., (2009) provided data from the City of Toronto's continuous water quality monitoring program. Chloride levels in seven streams located in four watersheds (Humber river, Don river, Highland Creek, Morningside tributary of the Rouge River) within the city limits were provided in the report. Monitoring for chloride levels in these streams has been occurring every hour, 24 hours a day, over a period from 2001 to the present. All 7 monitoring streams showed considerably higher levels of chloride measured during the winter period (November to March) when compared with the rest of the year. For example at Highland Creek, approximately 50% of the time during the winter months, stream chloride concentrations are likely to exceed the short-term benchmark concentration of 640 mg Cl^-/L. In the spring season (April to June), which most likely coincides with the release of glochidia from *Epioblasma torulosa rangiana, Lampsilis siliquoidea, Lampsilis fasciola*, stream chloride concentrations are likely to exceed 640 mg/L approximately 5% of the time, with concentrations never exceeding the short-term benchmark over the summer and early fall months (July to October) (Kilgour 2009). Therefore, the glochidia of these 3 freshwater mussel species will most likely be protected by the proposed short-term benchmark concentration of 640 mg Cl^-/L. This guideline will not likely be exceeded when glochidia are released during the spring period. However, what is not known is if brooding glochidia (held within their mothers during gestation) are affected by the salt-laden water their mothers are exposed to in early spring (Gillis, 2011).

Another factor to consider with respect to the life history of the 3 most sensitive mussel species, is that all 3 belong to the group of mantle lure spawners (P.Gillis, 2009, pers.comm.). These mussels use what is called a lure to attract a fish host. The lure is an extension of the mussel's mantle tissue which it allows to wave freely in the water current, essentially mimicking a minnow swimming. When a predator (host fish) moves in close to bite this lure it is sprayed with glochidia. For glochidia that do not attach to a host fish, one of two things can occur. The first is that glochidia can remain within the ruptured mantle, and therefore continue to be exposed to chloride in surface water. Subsequent fish attacks on the injured mantle can and do occur, therefore exposure of glochidia remaining in the mantle is a relevant exposure pathway (Bill Dimond, Michigan Department of Environmental Quality, pers.comm.). Secondly, the glochidia of luring mussel species are also found to drift in rivers (Morris *et al*., 2008). It appears that for mantle luring species, the amount of time that their glochidia would be exposed to waterborne contaminants would typically range from minutes (if they successfully attach to the lured fish) to one day (if they end up in the drift) (P.Gillis, 2009, pers.comm.). In the case of the endangered wavy-rayed lampmussel (*Lampsilis fasciola*), the proportion of glochidia that naturally survive to the juvenile stage is estimated to be as low as 0.000001% (Morris *et al*., 2008) – this is most likely the case for the other two species as well. Mussels overcome the extremely high mortality associated with this life cycle by producing large numbers of glochidia – often more than a million per female (Morris *et al*., 2008). Once glochidia attach to the host fish gill, they are more susceptible to toxicants in the fish gill than to toxicants in the water column (Cope *et al*., 2008).

Data for the juvenile life stage of 3 species of mussels was also obtained from the scientific literature (but not used in the short-term SSD, since the glochidia life stage was more sensitive). Free-living juveniles (transformed from glochidia encysted on a host fish) free themselves of the fish, and remain buried in sediment through the first 2 to 4 years of life (Cope *et al*., 2008). Chloride in sediment pore water would be the most significant exposure route at this life stage. 96 hour EC50s for the juvenile life stage of *Lampsilis fasciola*, *Lampsilis siliquoidea*, and *Villosa delumbis* were observed to be 2,414, 2,766 and 3,173 mg chloride/L (Bringolf *et al.*, 2007). The short-term benchmark concentration of 640 mg chloride/L indicates that there would be no potential for severe effects to the juvenile life stage of these three species of freshwater mussel (two of which are endangered), and as long as the sediment pore water concentrations remained below the respective LC50 concentrations.

Gillis (2011), who collected the endangered and special concern freshwater mussels (*Epioblasma torulosa rangiana* and *Lampsilis fasciola*, respectively) for use in laboratory-based toxicity studies, did not measure chloride concentrations at the site at time of collection. However, Gillis (2009, pers.comm.) did provide a summary of chloride concentrations measured in significant (stream and river) mussel habitats in southern Ontario and the number of mussel species found in each habitat (Table 9.9). Chloride measurements provided by the Provincial Water Quality Monitoring Network (PWQMN) of the Ontario Ministry of the Environment were used. Mean water quality values and ranges (minimum-maximum) are given for data collected from 1998 to 2008. Values reported as ‘Mean’ are the average of all site averages (repeated sampling at one site over time) for each conservation authority (CA). The number of individual site averages used to determine each ‘CA Mean’ or ‘CA Range’ is reported as *n*, standard deviation is given in parenthesis. NA indicates that data were not available.

Table 9.9 Summary of PWQMN measured chloride concentrations in significant (stream and river) mussel habitats in southern Ontario and the number of mussel species found in each habitat.

Conservation Authority	CA Mean Chloride (mg/L)	CA Range Chloride (mg/L)	#Mussel Species (Endangered)
Ausable Bayfield	34.9(12.6)[1], n=7	12.6-192	23 (6)
Grand River	53.1(1.3), n=45	2.4-507	25 (9)
St. Clair Region	41.5(14.1), n=9	8.0-149	34 (12)
Long Point Region	57.3(NA), n=1	25.0-114	10 (5)
Maitland Valley	38.3(28.8), n=13	6.8-212	9 (2)
Quinte	10.2(6.2), n=17	0.6-106	10 (2)
Saugeen Valley	11.5(4.5), n=14	1.4-39.9	8 (2)
Upper Thames River & Lower Thames Valley	57.7(38.1), n=38	6.2-1,300	26 (11)
Niagara Peninsula	29.2(9.1), n=3	14.8-111	10 (8)

[1]The number in parenthesis indicates the standard deviation around the mean.

Mean chloride concentrations measured at all of the CAs are well below the proposed short-term benchmark concentration for chloride of 640 mg/L. Only one chloride concentration exceeded the proposed short-term benchmark, and this was a maximum chloride concentration measured at the Upper Thames River and Lower Thames Valley CA, where the reported value was 1,300 mg/L. What is not known is the sampling period (date) and whether or not glochidia would be adversely impacted by increased chloride concentrations (e.g. glochidia are present from spring through to fall, depending on mussel species). Since endangered mussel species have been found in these CAs, protection should be provided to ensure they do not decrease in number.

Additional data from Ontario's PWQMN (see section related to *Lakes and Rivers of the Central Region, Ontario and Quebec,* earlier in this document) indicates that the short-term benchmark concentration is not expected to be exceeded in surface waters that are within undeveloped areas, areas with sparse road networks or areas that are predominantly rural residential (e.g. the Skootamotta River or the Sydenham River, where maximum measured chloride concentrations were 36 and 330 mg/L, respectively). The PWQMN data does show that the short-term benchmark concentration will be exceeded in developed watersheds (e.g. Fletcher's Creek at Brampton and Sheridan Creek, where maximum measured chloride concentrations were 4,150 and 5,320 mg/L). **Sensitive species located in surface waters within rapidly urbanizing watersheds or fully developed watersheds may be at risk of being adversely impacted by chloride (specifically from the application of road salt).**

9.1.5 Derivation of the Long-term Canadian Water Quality Guideline

Long-term exposure guidelines identify benchmarks in the aquatic ecosystem that are intended to protect all forms of aquatic life for indefinite exposure periods (≥7d exposures for fish and invertebrates, ≥24h exposures for aquatic plants and algae).

The minimum data requirements for the development of a long-term Type A (SSD-derived) guideline were met, and these are listed in Table 9.1.

A total of 28 data points (LC/EC/IC10, MATC, NOEC, ,LOEC and LC/EC/IC25 data) were used in the derivation of the long-term guideline (Table 9.11). The majority (22) of the data points plotted in the SSD represent no effects data (LC/EC/IC10, MATC, NOEC) and the remainder (6) represent low effects data (LOEC, LC/EC/IC25). Toxicity studies meeting the requirements for primary and secondary data, according to the Protocol for the Derivation of Water Quality Guidelines for the Protection of Aquatic Life (CCME 2007) protocol, were included. Intra-species variability was accounted for by taking the geometric mean of the studies considered to represent the most sensitive lifestage and endpoint. Each data point was ranked according to sensitivity, and its centralized distribution on the species sensitivity distribution (SSD) was determined using the Hazen plotting position (estimate of the cumulative probability of a data point). The plotting positions are treated as observed proportions of species affected. These positional rankings, along with their corresponding no-effects and low-effects values, were used to derive the SSDs.

The values reported in Table 9.11 range from a 24h EC10 (survival of glochidia) of 24 mg Cl^-/L for the COSEWIC special concern wavy-rayed lampmussel (*Lampsilis fasciola*) (Gillis 2009), to a 8-14 day MATC (growth inhibition) of 6,824 mg Cl^-/L for the alga, *Chlorella emersonii* (Setter *et al.*, 1982). When possible, data presented in studies was used to calculate the most preferable no-effect endpoint, being LC10 (see data order of preference in Table 9.1 as well as CCME 2007). Table 9.10 includes all studies for which sufficient data were available for calculation of an LC10. The full long-term, freshwater SSD dataset can be found in Table 9.11.

Two things to note regarding the brown trout study listed in Table 9.11. A 196h (8d) NOEC for *Salmo trutta fario* was used in the long term curve. However, the protocol calls for exposure periods ≥21d for testing on juvenile and adult fish. This study tested fingerlings. Still, CCME is in agreement for the inclusion of this data point in the long term curve, which allows this species to be represented. As well, with respect to the use of NOEC data, the CCME (2007) protocol states that "the use of toxicity data from a test where an insufficient concentration range on the higher end has been tested (i.e., where the results are expressed as "toxic concentration is greater than x"), are generally acceptable, as they will not result in an under-protective guideline. These types of data are best used as supporting evidence for other studies and <u>to help to fill minimum data requirements for guideline derivation"</u>.

Five cumulative distribution functions (normal, logistic, Gompertz, Weibull, and Fisher-Tippett) were fit to the data, both in arithmetic space (no transformation of no- and low-effect data) and log space (log transformed no- and low-effect data) using regression methods. Model fit was assessed using statistical and graphical techniques. The best model was selected based on consideration of goodness-of-fit test and model feasibility. Model assumptions were verified graphically and with the use of statistical tests.

Of the ten models tested, the log-Logistic model fit the data best (Figure 9.2). The Anderson-Darling Goodness of Fit test statistic (A^2) was 0.292 (P-value >0.10). The equation of the fitted log-Logistic model is of the form:

$$y = 1/[1+e^{-((x-\mu)/\sigma)}]$$

Where x is the log (concentration) and y is the proportion of species affected. For the fitted model, μ = 2.933 and σ = 0.292. Summary statistics for the long-term SSD are presented in Table 9.12. The 5^{th} percentile on the long-term SSD is 120 mg chloride/L. The lower fiducial limit (5%) on the 5^{th} percentile is 90 mg chloride/L, and the upper fiducial limit (95%) on the 5^{th} percentile is 155 mg chloride/L. The CWQG for protection of aquatic life is defined as the 5^{th} percentile on the SSD. **Therefore, the long-term exposure CWQG for the protection of freshwater life is 120 mg/L for the chloride ion.**

Table 9.10 Studies for which LC10s were calculated from published data and the statistical method used to calculate the LC10.

Organism	Test Duration	Calculated LC10 concentration (mg $NO_3^-\cdot L^{-1}$)	LC10 Statistical Method	Reference
Lampsilis fasciola	24-h	24	Probit	Bringolf 2010 (based on data from Bringolf *et al.,* 2007)
Epioblasma torulosa rangiana	24-h	42	Probit	Gillis 2009 (based on data published in Gillis *et al.,* 2011)
Daphnia ambigua	10-d	259	Linear Interpolation	Harmon *et al.,* 2003
Daphnia pulex	21-d	368	Point estimates calculated by Elphick *et al.,* 2010 using linear interpolation based on original data from Birge *et al.,* 1985	Elphick *et al.*, 2011 (based on data from Birge *et al.*, 1985)
Elliptio complanata	24-h	406	Probit	Bringolf 2010 (based on data from Bringolf *et al.,* 2007)
Pimephales promelas	33-d	598	Point estimates calculated by Elphick *et al.,* (2010) by using Multiple Linear Estimation (Probit) based on original data provided in Birge *et al.,* (1985)	Elphick *et al.*, 2011 (based on data from Birge *et al.*, 1985)
Villosa delumbis	24-h	716	Probit	Bringolf 2010 (based on data from Bringolf *et al.,* 2007)
Villosa constricta	24-h	789	Probit	Bringolf 2010 (based on data from Bringolf *et al.,* 2007)
Xenopus laevis	7-d	1,307	Linear Interpolation	Beak 1999
Lampsilis siliquoidea	96-h	1,474	Probit	Bringolf 2010 (based on data from Bringolf *et al.,* 2007)

Table 9.11 Long-term no effect and low effect concentrations for species exposed to chloride in freshwater.

Rank	Scientific Name	Common Name	Endpoint	Effective Concentration (mg Cl⁻/L)	Data Quality	Hazen Plotting Position	Reference
1	*Lampsilis fasciola*[a]	Wavy-rayed Lampmussel	24h EC10 (glochidia survival)	24	S	0.02	Bringolf *et al*., 2007
2	*Epioblasma torulosa rangiana*[b]	Northern Riffle Shell	24h EC10 (glochidia survival)	42	S	0.05	Gillis 2010
3	*Musculium securis*	Fingernail clam	60-80d LOEC (reduced natality)	121	S	0.09	Mackie 1978
4	*Daphnia ambigua*	Water flea	10-d EC10 (mortality and reproduction)	259	S	0.13	Harmon *et al*., 2003
5	*Daphnia pulex*	Water flea	21-d IC10 (reproduction)	368	S	0.16	Birge *et al*., 1985 In: Elphick *et al*., 2010
6	*Elliptio complanata*	Freshwater mussel	24-h EC10 (glochidia survival)	406	S	0.20	Bringolf *et al*., 2007
7	*Daphnia magna*	Water flea	21-d EC25 (reproduction)	421	P	0.23	Elphick *et al*., 2011
8	*Hyalella azteca*	Amphipod	28-d EC25 (growth, dry weight)	421	S	0.27	Bartlett 2009 (unpublished)
9	*Ceriodaphnia dubia*	Water flea	7-d IC25 (reproduction)	454	P	0.30	Elphick *et al*., 2011
10	*Tubifex tubifex*	Oligochaete	28-d IC10 (reproduction)	519	P	0.34	Elphick *et al*., 2011
11	*Pimephales promelas*	Fathead minnow	33-d LC10 (survival)	598	S	0.38	Birge *et al*., 1985 In: Elphick *et al*., 2010
12	*Salmo trutta fario*	Brown trout	8-d NOEC (survival)	607	S	0.41	Camargo and Tarazona 1991
13	*Villosa delumbis*	Freshwater mussel	24-h EC10 (glochidia survival)	716	S	0.45	Bringolf *et al*.,l 2007
14	*Villosa constricta*	Freshwater mussel	24-h EC10 (glochidia survival)	789	S	0.48	Bringolf *et al*., 2007
15	*Lumbriculus variegates*	Oligochaete	28-d EC25 (reproduction)	825	P	0.52	Elphick *et al*., 2011
16	*Oncorhynchus mykiss*	Rainbow trout	7-d EC25 (embryo viability)	989	P	0.55	Beak 1999
17	*Lemna minor*	Duckweed	96h MATC (frond production)	1,171	S	0.59	Taraldson and Norberg-King 1990
18	*Brachionus calyciflorus*	Rotifer	48-h IC10 (reproduction)	1,241	P	0.63	Elphick *et al*., 2011
19	*Xenopus laevis*	African clawed frog	7-d LC10 (survival)	1,307	P	0.66	Beak 1999
20	*Lampsilis*	Freshwater	96-h EC10	1,474	S	0.70	Bringolf *et al*.,

Rank	Scientific Name	Common Name	Endpoint	Effective Concentration (mg Cl^-/L)	Data Quality	Hazen Plotting Position	Reference
	siliquoidea	mussel (juveniles)					2007
21	*Gammarus pseudopinmaeus*	Amphipod	60-d NOEC (survival)	2,000	S	0.73	Williams *et al*., 1999
22	*Physa* sp.	Snail	60-d NOEC (survival)	2,000	S	0.77	Williams *et al*., 1999
23	*Stenonema modestum*	Mayfly	14-d MATC (development)	2,047	S	0.80	Diamond *et al*., 1992
24	*Chironomus tentans*	Midge	20-d IC10 (growth, biomass)	2,316	P	0.84	Elphick *et al*., 2011
25	*Rana pipiens*	Northern leopard frog	108-d MATC (survival)	3,431	S	0.88	Doe 2010
26	*Chlorella minutissimo*	Alga	28d MATC (Growth)	6,066	S	0.91	Kessler 1974
27	*Chlorella zofingiensis*	Alga	28d MATC (Growth)	6,066	S	0.95	Kessler 1974
28	*Chlorella emersonii*	Alga	8-14d MATC (Growth Inhibition)	6,824	S	0.98	Setter *et al*. 1982

[a]Status - Special Concern - as designated by COSEWIC.
[b]Status – Endangered - as designated by COSEWIC.
[c]Based on testing with $CaCl_2$ salt (all others based on testing with NaCl salt).
Data Quality:
S = Secondary; P = Primary
Grouped: Indicates that the geomean of multiple values was used to calculate the effect concentration

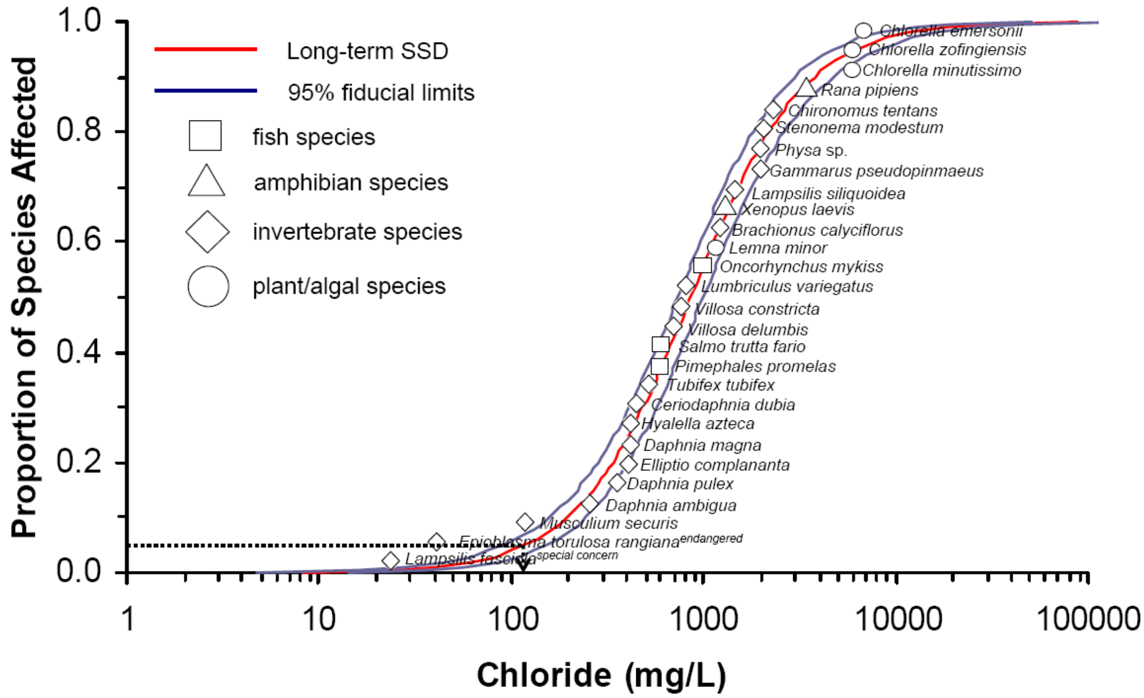

Figure 9.2 SSD of long-term no- and low-effect endpoint toxicity data for the chloride ion in freshwater (where mussels are present) derived by fitting the Logistic model to the logarithm of acceptable data for 28 aquatic species versus Hazen plotting position (proportion of species affected). The arrow at the bottom of the graph denotes the 5th percentile and the corresponding long-term Canadian Water Quality Guideline value.

Table 9.12 Long-term freshwater CWQG for the chloride ion resulting from the SSD Method – mussels present.

	Concentration
SSD 5th percentile	120 mg/L
SSD 5th percentile, 90% LFL (5%)	90 mg/L
SSD 5th percentile, 90% UFL (95%)	155 mg/L

In general, the most sensitive invertebrate species (daphnids and amphipods) are grouped towards the lower end of the SSD, with the fish species grouped midway, and the algal species grouped towards the upper end of the SSD. Two data points fall below the long-term SSD 5th percentile value of 120 mg Cl^-/L. These include the 24h EC10s of 24 mg Cl^-/L (Bringolf, 2010) and 42 mg Cl^-/L (Gillis, 2009) for two species of mantle lure spawning freshwater mussels (glochidia lifestage), including *Lampsilis fasciola*

(COSEWIC special concern) and *Epioblasma torulosa rangiana* (COSEWIC endangered) (Table 9.13). The CCME guideline derivation protocol (CCME 2007) provides the option of implementing the Protection Clause in situations where a data point for a species at risk, a species of commercial or recreational importance, or an ecologically important species falls below the HC5 (CWQG) value on the long-term SSD. In areas where the COSEWIC special concern mussel (*L. fasciola*) or the COSEWIC endangered mussel (*E. torulosa rangiana*) are present, the protection clause can be implemented, resulting in a guideline value ranging from 24 to 42 mg Cl^-/L. In all other areas where non-endangered freshwater mussels are present, the long-term SSD 5th percentile value of 120 mg Cl^-/L should be used as the guideline value. **Discussion with provincial regulators should occur if there is a need to develop more protective site specific values.**

As was discussed earlier, studies that are utilized for long-term CWQG (as well as short-term benchmark concentration) derivation commonly rely on exposures utilizing reconstituted water. Therefore by design, CWQGs will be conservative values. The proposed long-term chloride CWQG of 120 mg Cl^-/L exceeds background levels detected in unimpacted Canadian surface waters. With the exception of the naturally saline[4] lakes found in the Prairie Region (Manitoba, Saskatchewan, and Alberta) as well as the Pacific Region (British Columbia), background chloride concentrations in Canadian surface waters have been measured to be <1 to 30 mg/L. Chloride concentrations above background are commonly detected in densely populated areas (e.g. small urban watersheds) where road densities are high.

Table 9.13 24h EC10 values (survival of glochidia) for 2 species of COSEWIC assessed freshwater mussels.

COSEWIC Endangered Species	24h EC10 (mg Cl^-/L)	95% Confidence Intervals	Reference
Lampsilis fasciola Wavy-rayed lampmussel (COSEWIC special concern)	42	24, 57	Bringolf *et al*., 2007
Epioblasma torulosa rangiana Northern riffleshell mussel (COSEWIC endangered)	24	-79[5], 127	Gillis 2009

The northern riffleshell mussel is indigenous to the Ausable, Grand, Sydenham and Thames Rivers, as well as the Lake St. Clair delta. The wavy-rayed lampmussel is

[4] Prairie saline lakes can be classified as per Hammer (1986): subsaline (0.5-3 g/L total dissolved solids or TDS), hyposaline (3-20 g/L TDS), mesosaline (20-50 g/L TDS), and hypersaline (>50 g/L TDS).

[5] The negative lower fiducial limit is an artefact of the statistics. Biologically this can be interpreted as meaning that a 10% effect can be observed between a concentration of 0 and the upper 95% confidence limit. Therefore, the effect is not significantly different from the control (no-effect concentration) and could be due to natural variability.

indigenous to the lower Great Lakes and associated tributaries, specifically western Lake Erie, the Detroit River, Lake St. Clair and several southwestern Ontario streams.

Referring again to the four representative watershed types found within the province of Ontario (PWQMN data from pre-1980 to 2007), water samples collected from streams in undeveloped (Skootamotta River) or agricultural (Sydenham River) areas mostly had measured chloride concentrations at or below the proposed long-term chloride CWQG of 120 mg/L. One exception is two reported spikes in the Sydenham (164 and 330 mg/L). The median chloride concentrations for the Skootamotta and Sydenham Rivers were 2 and 10 mg/L, respectively. Water samples collected from streams in rapidly urbanizing (Fletcher's Creek) or fully developed (Sheridan Creek) areas had median measured concentrations exceeding the proposed long-term CWQG of 120 mg/L. The median chloride concentrations for Fletcher's and Sheridan Creek were 131 and 292 mg/L, respectively. Sensitive species are expected to be impacted in surface waters located in urbanized areas that receive road salt loadings.

Using the standard laboratory-cultured species, it is often reported that daphnids are the most sensitive receptors to chloride (US EPA, 1986; Iowa Cl WQG derivation, 2009). However, studies with alternative species (e.g. freshwater mussels, freshwater clams), suggest that more sensitive species are present in the environment, and the guidelines need to be conservative enough to protect these species, especially those that are endangered or at risk.

One long-term study that was not added to the SSD dataset used road salt in place of NaCl salt, but is worth discussing here. The study involved exposing egg clutches of the spotted salamander (*Ambystoma maculatum*) to three chloride concentration treatments: 1 mg/L (chloride measured in vernal pools >200m from a highway), 145 mg/L (mean chloride measured in vernal pools within 200m of a highway), and 945 mg/L (maximum chloride measured in vernal pools within 200m of a highway) (Karraker and Gibbs, 2011. Egg clutches were exposed to these chloride concentrations for a 9 day period, after which they were transferred to control water for another 9 day period and were weighed at day 3, 6, and 9 following transfer into clean water. The transfer into clean water was intended to mimic the dilution that occurs in vernal breeding pools following spring rainfall. Over the entire 18 day test period, clutches in the 1 mg/L treatment increased in mass by an average 25%, those in the 145 mg/L treatment lost an average mass of 2%, while clutches in the 945 mg/L treatment lost an average mass of 45%. Diluting rains may therefore aid in ameliorating the effects of moderate chloride concentrations in vernal breeding pools, however high chloride in breeding habitats may permanently disrupt the ability of egg clutches to osmoregulate, or take up water. This could result in increasing risk of predation, freezing, malformations and other adverse effects to embryos of the spotted salamander (Karraker and Gibbs, 2011). The CWQG of 120 mg Cl^-/L [1] is expected to be protective of the early life stage of the spotted salamander.

10.0 RESEARCH NEEDS

Chloride occurs in combination with cations forming salts such as NaCl, KCl, $CaCl_2$, and $MgCl_2$, and the interactions of different ions with chloride have been shown to affect toxicity. Although generalizations can be made as to which salt is the most toxic, there is a need for studies with the sole purpose of comparing the toxicity of chloride in combination with various cations. It has been shown that cations such as calcium can decrease chloride toxicity (Grizzle and Mauldin, 1995). In addition to this, there is a need for new methods to assess additive and synergistic effects of contaminants in surface water systems, since organisms are often exposed to more than one contaminant.

Laboratory studies have shown that aquatic species are more tolerant of salts in water with high oxygen concentrations (Fairchild, 1955; Evans and Frick, 2001). In the environment, chloride affects oxygen concentrations which can positively or negatively affect an ecosystem through various indirect effects. For example, loadings of chloride in lakes can result in the formation of meromictic lakes (do not experience complete overturn or complete vertical mixing) resulting in anoxic conditions in deeper waters which stress the ecosystem (Smol *et al.*, 1983). More studies of these complex ecosystem interactions should be conducted, as these interactions affect aquatic organisms in how they respond to chloride toxicity.

More studies (specifically long-term) assessing the impact of hardness on chloride toxicity are required in order to derive a national hardness-adjusted CWQG for the chloride ion.

11.0 HARDNESS AND SULPHATE CONCENTRATIONS IN CANADIAN SURFACE WATERS WITH A COMPARISON TO IOWA STATE WATER QUALITY

Table 11.1 provides an overview of the range of hardness and sulphate concentrations in the Canadian Regions (C. Lochner, 2009, pers.comm.). The Ontario data was provided by the Ontario Ministry's Environmental Monitoring and Reporting Branch as well as the Dorset Research Station. The 10^{th} percentile, median, 90^{th} percentile as well as max and min concentrations are presented in Table 11.1.

Table 11.1 Water quality summary for total hardness (as $CaCO_3$) and sulphate for the geographic regions of Canada.

Region	Province	Hardness (mg/L)					Sulphate (mg/L)				
			Percentiles					Percentiles			
		Min	10th	50^{th}	90^{th}	Max	Min	10^{th}	50^{th}	90^{th}	Max
Atlantic	Newfoundland and Labrador[5]	0.45	2.4	6.3	40	662	0.24	0.78	1.9	12	193
	Nova Scotia[5]	0.25	1.2	2.1	4.6	94	0.18	1.0	2.2	4.4	71
	New Brunswick[5]	0.62	2.2	9.7	66	831	0.11	1.8	2.9	9.8	2,442
	Prince Edward Island[5]	0.17	33	54	110	459	0.02	3.6	5.6	9.6	12
Central	Quebec[5]	2.9	9.0	38	112	1,078	0.25	2.8	7.5	24	210
	Ontario	5.8[1] 0.2[3]	93[1] 47[3]	226[1] 118[3]	318[1] 170[3]	1,920[1] 1,920[3]	3.5[2] 0.5[3]	3.7[2] 6.9[3]	4.9[2] 24[3]	5.5[2] 45[3]	6[2] 5,606[3]
Prairie	Manitoba[5]	41	46	287	402	590	3.1	4.0	116	226	395
	Saskatchewan[5]	25	145	300	531	702	1.7	22	124	235	402
	Alberta[5]	23	86	126	207	602	1.7	11	27	70	809
Pacific	British Columbia[4]	0.33	39	68	185	267	0.5	4.5	10	39	128
Territories	Yukon[5]	0.24	44	85	147	688	0.5	5.5	13	42	541
	Northwest Territories[5]	42	99	134	214	357	7.6	24	40	71	125
	Nunavut	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA

[1]PWQMN data collected 2003 to 2007 (P.Desai, Ontario MOE, 2009, pers.comm.).
[2]Dorset inland lake monitoring data collected in 2007 (A.Paterson, Ontario MOE, 2009, pers.comm.).
[3]Great Lakes Great Lakes Connecting Channel data from Environmental Monitoring and Reporting Branch collected 1990 to 2007 (P.Desai, Ontario MOE, 2009, pers.comm.)
[4]British Columbia Federal-Provincial river trend sites, with data collected from 1979 to 2009 (T. Dessouki, British Columbia MOE, 2009, pers.comm.)
[5]C. Lochner, Water Quality Monitoring and Surveillance, Environment Canada, 2009, pers.comm
NA = data was not available

In Section 7.4 (Hardness), it was indicated, as a footnote to Table 7.1 (which listed studies that investigated hardness as a toxicity modifying factor) that the reasonable extreme for Canadian surface water hardness levels is 5 to 240 mg/L (as $CaCO_3$). This is supported by water quality monitoring data presented by Natural Resources Canada in the Hydrological Atlas of Canada, where it is stated that a general and arbitrary classification for hardness (as $CaCO_3$) of Canadian waters is as follows: soft 0-60 mg/L, moderately hard 61-120 mg/L, hard 121-250 mg/L, and very hard >250 mg/L (NRCAN, 1978). As can be seen in Table 11.1, the 90^{th} percentile of recently measured water hardness in Canada rarely exceeds 240-250 mg/L (as $CaCO_3$), except for in Ontario, Manitoba and Saskatchewan, where the respective 90^{th} percentiles are 318, 402 and 531 mg/L (as $CaCO_3$).

The supporting document for the Canadian Drinking Water Quality Guideline (CDWQG) for hardness indicates that, based on a survey of surface waters conducted in 1975 to 1977, the median value measured at each station (41 were selected as being representative of Canadian waters) rarely exceeded 120 mg/L, except for in the Nelson-Saskatchewan and Mississippi basins (Heath Canada, 1979). The waters of these river systems are considered to be hard, as most measurements exceeded 180 mg/L. None of the median concentrations for these 41 stations exceeded 500 mg/L. Average hardness levels at the time of monitoring were found to range as follows: British Columbia, 7 to 180 mg/L; Northwest Territories, 5 to 179 mg/L; Alberta, 98 to 329 mg/L; Saskatchewan, 12 to 132 mg/L; Manitoba, 15 to 716 mg/L; upper Great Lakes, 40 to 80 mg/L; and Ontario lakes and streams, 2 to 1803 mg/L (but most measurements were between 40 and 200 mg/L). No data was available for the maritime provinces. Figure 11.1 provides mapping of hardness measured in Canadian surface waters.

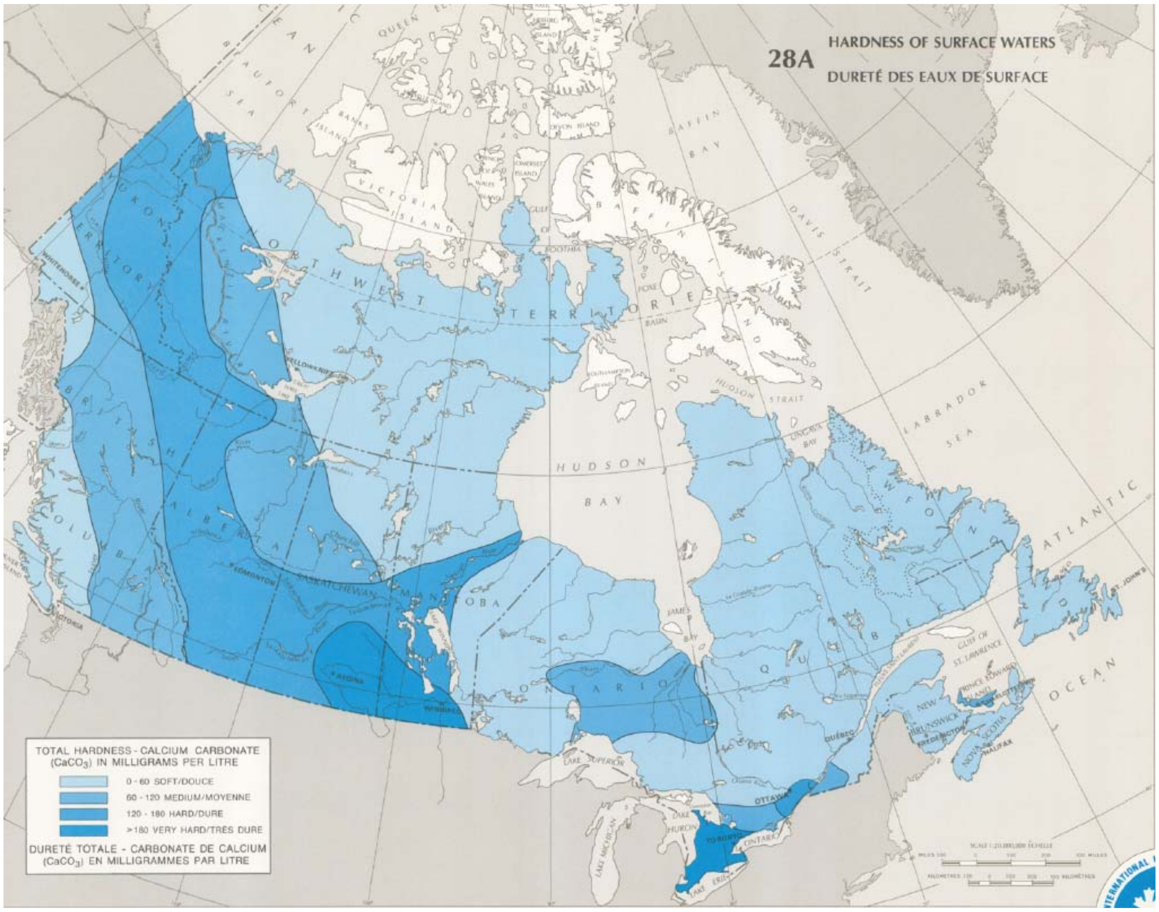

Figure 11.1 Total hardness of surface waters as calcium carbonate ($CaCO_3$) in mg/L (NRCAN, 1978).

As can be noted by the measurements presented in Table 11.1 and Figure 11.1, some areas of Saskatchewan, Manitoba and Ontario do have waters classified as being very hard (>180 mg/L as $CaCO_3$). For example the distribution of hardness concentrations for several streams arising within Saskatchewan are presented in Figure 11.2 below (J.M. Davies, Saskatchewan Watershed Authority, pers.comm.). All but one of these streams has a 90th percentile chloride concentration <100 mg/L and the majority of these streams have a median TDS that is freshwater (<1000 mg/L see Last & Ginn 2005 for definition). The horizontal line in the figure below = 240 mg/L. Therefore, this does provide evidence that there are some areas within Canada that fall outside of the designated reasonable extreme for Canadian surface water hardness levels of 5 to 240 mg/L (as $CaCO_3$).

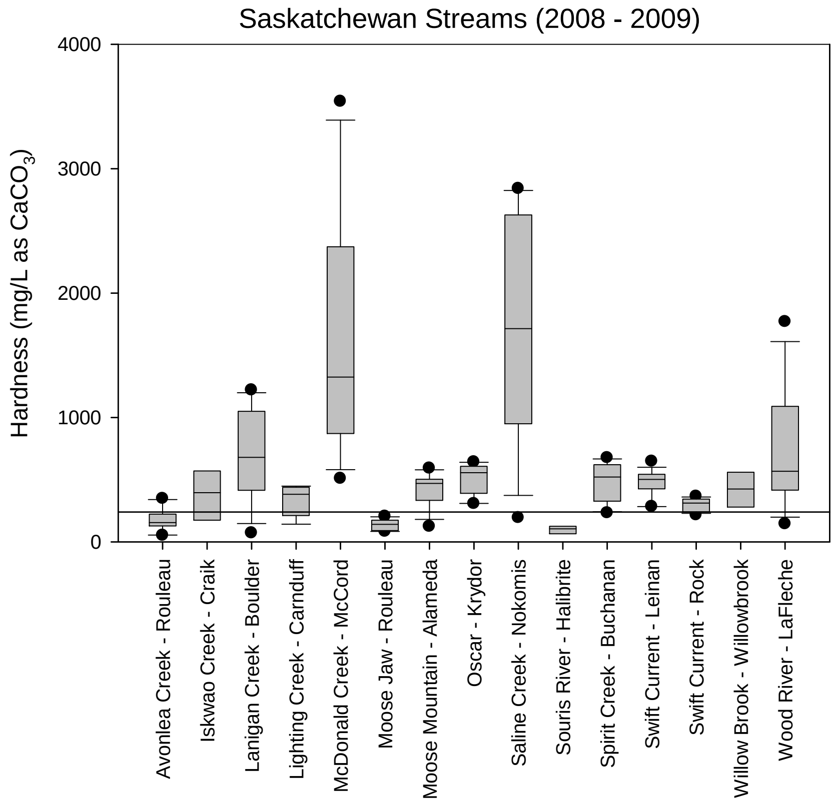

Figure 11.2 The distribution of hardness concentrations for several streams arising within Saskatchewan (J.M. Davies, Saskatchewan Watershed Authority, pers.comm.).

For comparative purposes, Iowa water quality (hardness and suplate) is presented in Table 11.2. This data is provided since the state of Iowa has developed a draft hardness- and sulphate-adjusted water quality guideline for the chloride ion. Iowa statewide ambient water quality monitoring data from 2000 to 2008 provided a 10th percentile for hardness of 200 mg/L as $CaCO_3$ (much harder than most Canadian surface waters). The corresponding sulphate concentration was selected by regression analysis of sulphate versus hardness, resulting in a statewide sulphate default value of 63 mg/L (C.Dou, Iowa DNR, pdf presentation 2009). These were the standard default values used in the proposed chloride criteria generated in March 2009. The most recent update in May 2009 uses standard or default hardness (median) and sulphate values of 300 and 65 mg/L, respectively. The US EPA has calibrated the hardness adjustment equation in the range of 25 to 800 mg $CaCO_3$/L and the sulphate adjustment equation in the range of 22.9 to 729 mg sulphate/L, so the relationships should be good over these ranges (C.Stephan, US EPA, 2009, pers.comm.). Hardness and sulphate data for the state of Iowa surface waters measured from 2000 to 2008 is presented in Table 11.2.

Table 11.2 Iowa water quality summary for total hardness (as $CaCO_3$) and sulphate 2000-2008 (Iowa DNR 2009).

Parameter	Unit	Number of Samples	Min Value	Percentiles					Max Value
				10th	25th	50th	75th	90th	
Hardness	mg/L	8,319	55	200	240	300	360	410	820
Sulphate	mg/L	7,368	<1	20	26	37	60	96	400

The 90th percentile of total hardness (as $CaCO_3$) measured in Iowa surface waters was 410 mg/L. The 90th percentile total hardness measurements for Canadian surface waters were all less than 410 mg/L (ranging from 4.6 to 402 mg/L), with the exception of Saskatchewan surface waters, where the 90th percentile was 531 mg/L. Saskatchewan is an anomaly as this province has naturally elevated salinity in surface waters, and thus chloride levels tend to be higher.

Based on information related to hardness-toxicity relationships for chloride presented in Section 7.4.1, it was decided that insufficient data was available in order to develop a hardness relationship for chronic toxicity. Therefore, a hardness based national CWQG was not developed. CCME will re-visit the chloride guidelines when sufficient studies are available. Jurisdictions have the option of deriving site-specific hardness adjusted water quality criteria if they so choose.

In the case of adjusting the chloride guideline for sulphate, it has also been decided to not pursue this for the development of the chloride CWQG. Over a range of sulphate concentrations (25 to 600 mg/L) and constant hardness (300 mg/L), only a 12% reduction in chloride LC50 concentrations was observed for *Ceriodaphnia dubia* when comparing exposures in low sulphate (25 mg/L) to exposures in high sulphate (600 mg/L) (see Table 7.1). This 12% reduction in LC50 is not considered to be a significant increase in chloride toxicity, especially when consideration is given to sulphate concentrations in Canadian surface waters. Canadian surface water 90th percentile sulphate concentrations are fairly low, ranging from 4.4 to 71 mg/L. Higher 90th percentile sulphate measurements were reported for provinces in the prairie region (Manitoba and Saskatchewan), where lakes are found to be naturally high in salinity due to underlying geology.

12.0 COMPARISON OF GUIDELINE VALUES TO FIELD VALUES

Comparison between chloride guideline values and observations of effects in the field can be skewed by the fact that there may be a combination of stressors present in the field (e.g. salinity, temperature, sediment, nutrient and habitat change), besides just increased chloride levels, which can affect species absence or presence. When using a guideline value (e.g. chloride) to compare to measurements made in the field, one must also look to other stressors and make comparisons to the respective guideline values. A problem in interpretation of cause of effect may arise when two or more stressors approach their respective guideline values, so that additive, synergistic or antagonistic effects are possible (Rutherford and Kefford, 2005).

In Kilgour *et al.,* (2009), water quality monitoring data collected hourly at seven locations by the City of Toronto in four major watersheds (Rouge, Highland, Humber, Don River) was used to assess fish (collected once every 2-3 years) and benthos (collected annually) monitoring data collected by the Toronto Region Conservation Authority (close to the sites of water sampling). It was recognized that in the four Toronto watersheds, chloride would not be the single factor affecting fish and benthos distribution. Constrained ordination indicated that chloride explained at least 13% but not more than 30% of the variation in benthic taxa distribution, and that the effects of aluminum, total phosphorus, bankfill width, stream depth and substrate size all covaried with chloride. A distinct change in benthic community structure was evident at measured chloride concentrations of 250 mg/L (abundances of stoneflies, beetles and water mites was reduced), however, at this chloride concentration, total phosphorus and aluminum commonly exceeded Ontario's Provincial Water Quality Objective of 0.03 mg/L and 75 ug/L, respectively. In the case of fish distribution, after the potential influences of aluminum, phosphorus, stream width and depth were removed, chloride accounted for 17% of the variation in community composition.

Maximum field distributions (MFD), or maximum chloride concentrations at which species are observed in the field, were constructed by Kilgour *et al.,* (2009) using data collected for 251 benthic taxa (to create a benthos MFD) and 22 fish species (to create a fish MFD). Similar to an SSD, percent of taxa impacted was plotted against the chloride concentration. The benthic MFD 5^{th} percentile (or concentration that would protect 95% of organisms) was 38 mg Cl^-/L. This fish MFD 5^{th} percentile (or concentration that would protect 95% of organisms) was 189 mg Cl^-/L.

A study by Watson-Leung (2002) investigated road salt induced invertebrate community structure changes in lentic systems. It was noted that historically saline lakes (e.g. having naturally elevated chloride) will likely have fauna present that are genetically and physiologically adapted to these conditions. However, aquatic biota (e.g. pond invertebrates in storm-water or naturally-occuring ponds) exposed to high chloride levels too rapidly (e.g. during spring thaw) likely do not have the time to evolve physiological tolerance. Therefore these invertebrate communities may be altered by either direct physiological effects, or indirectly by impacting on the food chain. Average chloride concentrations measured in the stormwater ponds ranged from 165 to 3977 mg Cl^-/L and for naturally-occuring ponds, chloride levels ranged from 95 to 220 mg/L. Multivariate stastical techniques provided indication that many environmental variables were correlated making it difficult to determine which environmental variable, or combination of variables, was found to be most important for determing invertebrate community structure. Overall, it was concluded that land use was identified as the most important variable, with chloride concentration being secondary.

12.1 Zooplankton Communities in Naturally Saline Lakes in Canada

In freshwater systems, as salinity increases, the diversity of aquatic species decreases due to the exceedance of organism-specific osmotic tolerances (Derry *et al.,* 2003). Hammer

(1993) sampled 17 saline lakes in Saskatchewan and 3 in Alberta, where salinity ranged from 2.8 to 269 mg/L. Hammer (1993) observed greatest species richness (15-16 species) at salinities <7g/L, 6-8 zooplankton species in lakes with salinity ranging from 7-100 g/L, whereas the most saline lakes (>100 g/L) had the fewest species (2-5). What is not well understood is whether it is salt ion composition or salt concentration (e.g. overall salinity) that is the dominant factor in determining aquatic community assemblages (Derry *et al.*, 2003). Derry *et al.*, (2003) conducted a study with an attempt to identify the relationship between types of salts and resulting zooplankton communities. Zooplankton communities were collected and compared between Canadian inland saline lakes dominated by Cl (a rarity in Canada) and inland saline lakes dominated by SO_4/CO_3 (common in Canada). All lakes had small surface areas (≤150 ha) and were shallow (<3.4m mean depth). Twelve lakes were selected for the study, with Cl-dominated lakes located in the northeastern part of Alberta and SO_4/CO_3-dominated lakes located in central Alberta. The categories of lake-water salinity encountered were:

1) sub-saline (0.5-1 g/L TDS) with either $-SO_4$ or -Cl ion dominating,
2) hypo-saline (7-14 g/L TDS) with $-SO_4$ ion dominating,
3) hypo-saline (7-14 g/L TDS) with -Cl ion dominating,
4) meso-saline (37-40 g/L TDS) with -Cl ion dominating, and
5) hyper-saline (~100 g/L TDS) with $-CO_3$ ion dominating.

With respect to the subsaline lakes, one is dominated by SO_4^{2-} with a chloride concentration of 29 mg/L. The 4 other subsaline lakes were all chloride dominated, with chloride concentrations ranging from 133 to 465 mg/L. Two of the 4 hyposaline lakes were SO_4^{2-} dominated, with chloride levels of 171 and 280 mg/L. The other 2 hyposaline lakes that were Cl-dominated had chloride concentrations of 3832 and 5672 mg/L. The mesosaline lakes were all chloride dominated, with chloride concentrations of 13,485 and 16,765 mg/L. The 1 hypersaline lake, dominated by CO_3^{2-}, had a chloride concentration of 856 mg/L.

Ion composition varied among the study lakes (Table 12.1). One of the 5 sub-saline lakes was dominated by SO_4, whereas the other 4 were dominated by Cl. The 4 saline lakes from northern Alberta dominated by Cl also had high concentrations of SO_4. In contrast, the 3 SO_4/CO_3 dominated lakes from central Alberta had low Cl concentrations and were more alkaline (28-68 times) than the Cl-dominated lakes. Nutrient concentrations also varied between the 2 types of saline waters, with Cl-dominated lakes from northern Alberta being mesotrophic (as per TN:TP ratio), and SO_4/CO_3-dominated lakes from central Alberta being hyper-eutrophic. Differences in zooplankton communities observed among study lakes with contrasting ion composition was confounded by covariation in nutrient levels and predation pressure (e.g. nine-spiked stickleback, *Pungitius pungitius*, were detected in 6 of the 8 Cl-dominated lakes, whereas lakes dominated by SO_4/CO_3 were fishless). Other variables that affected zooplankton species composition was lake surface area, mean depth, and the presence of associated streams.

With respect to the Cl-dominated subsaline and saline lakes (which were small and shallow, and had surface connectioins to other rivers and lakes), nine-spined stickleback fish (commonly found in both brackish and freshwaters) and corixid predators (bugs which feed on other insects) were present. Many rotifer (Table 12.2) and crustacean

(Table 12.3) species were abundant in the more dilute subsaline lakes (where chloride concentrations ranged from 29-465 mg/L). Low to intermediate concentrations of chloride (133-5672 mg/L) were tolerated by several rotifer species (*Lophocharis salpina, Keratella quadrata, Notholca acuminate*). The hypo-saline (chloride range of 3832-5672 mg/L) and meso-saline (chloride range of 13,485-16,765 mg/L) Cl-dominated lakes of northern Alberta had an abundance of the euryhaline rotifer *Brachionus plicatilis*, as well as the rotifers *Hexartha fennica* and *Cletocamptus albuquerquensis*. A broad range in salinity was tolerated by the halophile rotifer *Brachionus plicatilis*, tolerating 840-26,318 mg/L TDS.

Different zooplankton communities were found to exist in hyposaline lakes that were high in Ca and dominated by Cl, when compard to zooplankton in hyposaline lakes that were alkaline, hyper-eutrophic and dominated by SO_4/CO_3. The most abundant zooplankton in hyposaline SO_4/CO_3-dominated lakes (where chloride concentrations ranged from 171-856 mg/L) were the copepods *Leptodiaptomus sicilis* and *Diaptomus nevadensis*, and the water flea *Daphnia similis.* These 3 species have all been observed in other North American SO_4-dominated saline waters, but have never been reported to exist in North American Cl-dominated waters (Derry *et al*., 2003).

Overall, it was argued that zooplankton distribution is associated with physiological ion tolerance. Large crustacean zooplankton such as the copepods *Leptodiaptomus sicilis* and *Diaptomus nevadensis*, and the water flea *Daphnia similis*, were only found in SO_4-dominated lakes, while the anostracan *Artermia franciscana* was found in hypersaline CO_3-dominated lakes (which also lacked fish predators) (Tables 12.2 and 12.3). Cl-dominated waters had high densities of corixids and nine-spiked stickleback fish, but large calanoid copepods and cladocerans were absent. It has been documented by Koel and Peterka (1995, In: Derry *et al.,* 2003) that SO_4-dominated waters are more stressful for osmoregulation in fish larvae than Cl-dominated waters, and these differences in ion strength may also select zooplankton that inhabit salt lakes. Further testing on specific ion tolerance in different species would be helpful in determining biogeographic patterns of zooplankton habitat utilization (Derry *et al.,* 2003).

Historically saline lakes (e.g. having naturally elevated chloride) will likely have fauna present that are genetically and physiologically adapted to these conditions.

Table 12.1 Average measurements of TDS (mg/L), conductivity (uS/cm), major nutrients (ug/L), and ion (mg/L) concentrations for the study lakes over the summer of 1999. pH, DOC (mg/L), turbidity (NTU), colour (mg/L Pt), chl *a* (ug/L) and Secchi depth (m) are also presented. SO_4 dominated saline waters in central Alberta were measured only in June. ND indicates where no data was available and B represents "bottom" for Secchi depths (Derry *et al.*, 2003). [At the end of the abbreviations for the study lakes, $-SO_4$ is a sulphate dominated saline lake, $-CO_3$ is a carbonate dominated saline lake, -Cl is a chloride dominated saline lake, -D is a dilute subsaline lake. Salinity classifications are as follows: subsaline (0.5-3 g/L TDS), hyposaline (3-20 g/L TDS), mesosaline (20-50 g/L TDS) and hypersaline (>50 g/L TDS)].

Saline Lakes Salinity Category	$OL-CO_3$ Hyper-Saline	$PN-SO_4$ Hypo-Saline	$FL-SO_4$ Hypo-Saline	GB-Cl Meso-Saline	HC-Cl Meso-Saline	SP-Cl Hypo-Saline	SL-Cl Hypo-Saline
TDS	96 228	14 277	8028	26 318	25 605	12 676	7282
Conductivity	73 336	16 441	10 230	40 217	36 805	18 926	12 144
TP	25 530	2915	1322	18	64	29	85
TN	862	47	62	741	848	1673	1473
NO_2+NO_3	N.D.	N.D.	N.D.	2	39	2	10
Na^+	37 048	4898	2857	9520	8205	3726	2491
Ca^{2+}	0.6	4.0	2.8	457	679	806	146
K^+	607	128	79	7	25	4	7
Mg^{2+}	46	95	58	36	236	48	64
Fe^{2+}	2.0	0.2	1.2	0.13	0.15	0.06	0.15
Mn^{3+}	0.10	0.02	0.03	0.04	0.06	0.04	0.10
Cl^-	856	280	171	16 765	13 485	5672	3832
$SO4^{2-}$	3 387	7 544	3 590	1209	1412	2219	340
$CaCO_3$	45 232	3264	3778	59	106	54	137
HCO_3^-	9377	2666	3024	72	129	66	167
CO_3^{2-}	22 508	646	778	0	0	0	0
pH	10.2	9.6	9.3	8.4	8.8	8.2	8.3
DOC	290	72	75	14	17	29	20
Turbidity	3.0	10.0	27.0	1.4	3.4	3.3	2.0
Colour	67	35	97	11	68	17	34
Chl *a*	2.0	4.9	2.7	3.2	12.8	6.0	2.6
Secchi Depth	0.6	0.55	0.30	B	B	B	B

Subsaline Lakes	GL-D	GW-D	BP-D	FP-D	WR-D
TDS	1090	982	608	840	556
Conductivity	1267	1805	862	1222	824
TP	15	65	39	124	32
TN	620	1150	1047	1520	908
NO_2+NO_3	47	38	26	33	19
Na^+	31	225	68	127	74
Ca^{2+}	193	77	50	71	39
K^+	3.0	8.0	4.9	3.6	6.0
Mg^{2+}	47	43	42	39	32
Fe^{2+}	0.003	0.3	0.02	0.4	0.1
Mn^{3+}	0.007	0.2	0.007	0.05	0.005
Cl^-	29	465	133	247	178
$SO4^{2-}$	649	16	90	45	0.6
CaCO3	83	130	170	203	134
$HCO3^-$	101	157	1047	239	161
$CO3^{2-}$	0	0.7	3.3	4.0	0.8
pH	8	7.8	8.6	7.9	7.8
DOC	10	27	24	39	22
Turbidity	0.7	2.2	0.9	6.6	3.2
Colour	10	94	78	206	92
Chl *a*	4.0	4.1	6.9	25.3	3.6
Secchi Depth	3.5	1.7	1.4	0.7	1.2

Table 12.2 Peak density observed for rotifer species (#individuals/L lake water) in each category of lakewater salinity over the summer of 1999 (Derry *et al.,* 2003). [At the end of the abbreviations, -SO_4 is a sulphate dominated lake, -CO_3 is a carbonate dominated lake, -Cl is a chloride dominated lake. Salinity classifications are as follows: subsaline (0.5-3 g/L TDS), hyposaline (3-20 g/L TDS), mesosaline (20-50 g/L TDS) and hypersaline (>50 g/L TDS)].

Species	Hyper-Saline CO_3	Hypo-Saline SO_4	Meso-Saline Cl	Hypo-Saline Cl	Sub-Saline SO_4 or Cl
Ascomorpha ecaudis	0	0	0	0	89
Ascomorpha ovalis	0	0	0	0	113
Anuraeopsis fissa	0	0	0	0	26
Asplanchna brightwelli	0	0	0	0	15
Asplanchna priodonta	0	0	0	0	428
Brachionus plicatilis	0	0	869	202	2712*
Brachionus quadridentatus	0	0	0	1	18
Brachionus rubens	0	0	0	0.4	0
Brachionus urceolaris	0	26	0	0	0
Collotheca mutabilis	0	0	0	0	171
Collotheca pelagica	0	0	0	0	2
Colurella obtusa	0	0	0	0	6
Colurella uncinata	0	0	0	0	7
Encentrum sp.	0	0	0	0	1
Filinia longiseta	0	0	0	0	2383
Gastropus stylifer	0	0	0	0	113
Hexarthra sp.	2	0	0.2	752	3
Keratella cochlearis	0	0.3	0	0	1441
Keratella hiemalis	0	0	0	1	122
Keratella quadrata	0	0.1	3	19	6733
Keratella serrulata	0	0	0	0	29
Keratella testudo	0	0.1	0	0	3372
Keratella ticinensis	0	0	0	0	1
Keratella valga	0	0	0	0	1
Lecane luna	0	0	0	0	13
Lecane ohioensis	0	0	0	0	6
Lepadella acuminata	0	0	0	0	9
Lepadella patella	0	0	0	0	13
Lophocharis salpina	0	0	0	2	3
Monostyla bulla	0	0	0	0	239
Monostyla closterocerca	0	0	0	0	13
Monostyla lunaris	0	0	0	0	13
Monostyla quadridentus	0	0	0	0	6
Mytilina ventralis var *brevispina*	0	0	0	0	53
Notholca acuminata	0	0	0	5	26
Notommata sp.	0	0	0	0	51
Platyias patulus	0	0	0	0	29
Polyarthra dolichoptera	0	0	0	0	13
Polyarthra vulgaris	0	0.4	0	0	3130
Pompholyx sp.	0	0	0	0	733
Synchaeta sp.	0	0	0	0	3975
Testudinella patina	0	0	0	0	12
Trichocerca longiseta	0	0	0	0	8
Trichocerca lophoessa	0	0	0	0	3
Trichocerca multicrinis	0	0	0	0	167
Trichocerca rattus	0	0	0	0	5
Trichotria pocillum	0	0	0	0	9
Trichotria tetractis	0	0	0	0	6
Vanoyella globosa	0	0	0	0	3

Table 12.3 Peak density observed for crustacean species (#individuals/L lake water) in each category of lakewater salinity (summer 1999) (Derry *et al.,* 2003). [At the end of the abbreviations, $-SO_4$ is a sulphate dominated lake, $-CO_3$ is a carbonate dominated lake, -Cl is a chloride dominated lake. Salinity classifications are as follows: subsaline (0.5-3 g/L TDS), hyposaline (3-20 g/L TDS), mesosaline (20-50 g/L TDS) and hypersaline (>50 g/L TDS)].

Species	Hyper-Saline CO_3	Hypo-Saline SO_4	Meso-Saline Cl	Hypo-Saline Cl	Sub-Saline Cl or SO_4
Anostracans					
Artemia franciscana	30	0	0	0	0
Calanoid Copepods					
Agalodiaptomus leptopus	0	0	0	0	68
Diaptomus arcticus	0	0	0	0	13
Diaptomus nevadensis	0	0.4	0	0	0
Leptodiaptomus nudus	0	0	0	18	0
Leptodiaptomus sicilis	0	6	0	0	0.2
Cyclopoid Copepods					
Acanthocyclops carolinianus	0	0	0	0.8	0
Acanthocyclops robustus	0	0	0	0	15
Acanthocyclops venustoide	0	0	0	0	97
Acanthocyclops vernalis	0	0	0	0	5
Diacyclops navus	0	0	0	0.5	3
Harpacticoid Copepods					
Cletocamptus sp.	0	0	61	0.8	0
Cladocerans					
Alona circumfimbriata	0	0	0	0	6
Alona costata	0	0	0	0	3
Alona guttata	0	0	0	0	5
Alona rectangula	0	0	0	0.2	2
Bosmina sp.	0	0	0	0	28
Ceriodaphnia laticaudata	0	0	0	0	76
Ceriodaphnia pulchella	0	0	0	4	0
Chydorus brevilabris	0	0	0	0	96
Chydorus piger	0	0	0	0	19
Chydorus sphaericus	0	0	0	0	19
Daphnia parvula	0	0	0	0	38
Daphnia pulicaria/pulex	0	0	0	6	262
Daphnia rosea	0	0	0	0	29
Daphnia schoedleri	0	0	0	0	32
Polyphemus pediculus	0	0	0	0	6

13.0 GUIDANCE ON APPLICATION OF THE GUIDELINES

13.1 General Guidance on the Use of Guidelines

The short-term benchmark concentration and long-term CWQG for chloride are set to provide protection for short- and long-term exposure periods, respectively. They are based on generic environmental fate and behaviour and toxicity data. The guideline is a conservative value below which all forms of aquatic life, during all life stages and in all Canadian aquatic systems, should be protected. Because the guideline is not corrected for any toxicity modifying factors (e.g. hardness), it is a generic value that does not take into account any site-specific factors. Moreover, since it is mostly based on toxicity tests using naïve (i.e., non-tolerant) laboratory organisms, the guideline may not be relevant for areas with a naturally elevated concentration of chloride and associated adapted ecological community (CCME 2007). Thus, if an exceedence of the guideline is observed (due to anthropogenically enriched water or because of elevated natural background concentrations), it does not necessarily suggest that toxic effects will be observed, but rather indicates the need to determine whether or not there is a potential for adverse environmental effects. In some situations, such as where an exceedence is observed, it may be necessary or advantageous to derive a site-specific guideline that takes into account local conditions (water chemistry, natural background concentration, genetically adapted organisms, community structure) (CCME 2007).

The guideline should be used as a screening and management tool to ensure that chloride does not lead to the degradation of the aquatic environment. The CWQG for chloride could, for example, be the basis for the derivation of site-specific guidelines and objectives (derived with site-specific data as well as consideration of technological, site-specific, socioeconomic or management factors) (CCME 2007).

Fiducial limits are reported along with the HC5 or guideline value. Fiducial limits (FLs) are essentially the inverse of confidence intervals (CIs), where FLs are horizontal around the X (concentration) for a specified Y (HC5) whereas CIs are verticle around the Y, for a specified X. For example, in the case of FLs, there is 95% certainty that at the HC5 (assume this to be 1 mg/L), the concentration is between 0.8 and 1.2 mg/L, with a mean of 1 mg/L. In the case of CIs, there is 95% certainty that at a concentration of 1 mg/L, that HC% is between 2.1 and 6.7 with a mean of 5. For guideline development, an inverse prediction is being used, specifying a Y (HC5) to estimate an X value (concentration), so FLs are more appropriate than CIs. FLs are essentially reported because they help to assess the fit of the selected curve or model to the dataset. As the number of data points plotted on an SSD increases, the fit of FLs should be tighter. FLs can also be used to help interpret monitoring data, particularly if the guideline and method detection limit are close. Only the HC5 is used as the guideline value.

CWQG values are calculated such that they protect the most sensitive life stage of the most sensitive aquatic life species over the long term. Hence, concentrations of a parameter that are less than the applicable CWQG are not expected to cause any adverse effect on aquatic life. Concentrations that exceed the CWQGs, however, do not necessarily imply that aquatic biota will be adversely affected, or that the water body is impaired; the concentration at which such effects occur may differ depending on site-specific conditions. Where the CWQGs are exceeded, professional advice should be sought in interpreting such results. As with other CWQGs, the guidelines for nitrate are intended to be applied towards concentrations in ambient surface waters, rather than immediately adjacent to point sources such as municipal or industrial effluent outfalls. Various jurisdictions provide guidance on determining the limits of mixing zones when sampling downstream from a point source (see, for example, BC MELP 1986 and MEQ 1991), though Environment Canada and the CCME do not necessarily endorse these methods.

13.2 Monitoring and Analysis of Chloride Levels

In comparing surface water measurements of chloride to the Canadian water quality guidelines, it is important to be aware of potential seasonal and meteorological impacts at the time of sampling. Chloride concentrations in surface waters can peak for short periods of time during storm events and spring melt. As these pulses often occur in the spring when the most sensitive life stages (e.g., larvae) for many organisms are present, their relationship to the guideline should be considered. A stream may normally have a low baseline concentration of chloride, but during and immediately following (1-2 days) one of these events, the chloride concentrations could exceed the guideline value. The exceedance could result from one of two scenarios. First, the increase in chloride could occur as a result of a natural increase in background levels. Second, the source of the chloride in storm- or meltwater may not be natural; for example, it could be due to runoff from urban areas where road salt has been applied. In the former case the guidelines do not strictly apply (because a guideline cannot be set lower than natural background levels for a naturally occurring substance). Nonetheless, we recommend that if chloride levels are found to exceed the recommended guideline values, that data on the frequency and severity of the exceedances should be evaluated on a site-specific basis to determine whether they warrant any preventative or remedial actions.

For monitoring long-term temporal trends in chloride levels, an undue weighting should not be given to samples that were collected during, or immediately following a storm event, or during the spring thaw. Due to seasonal variability in chloride levels, comparison of long-term trend data should occur between standardized collection intervals over similar time periods (i.e, spring, summer, fall, winter).

13.3 Developing Site-Specific Guidelines and Objectives

National guidelines, such as the one for chloride, can be the basis for the derivation of site-specific guidelines (e.g. derived with site-specific scientific data) as well as objectives (e.g. derived with site-specific scientific data as well as consideration of technological, site-specific socioeconomic, or management factors) (CCME 2007). There are some cases in which the development of site-specific objectives for chloride should be considered. The guidelines were derived to be protective of all forms of aquatic life and all aspects of the aquatic life cycles, including the most sensitive life stage of the most sensitive species over the long term. However, in locations where highly sensitive or endangered species occur, or in areas where species of commercial / recreational importance occur, water managers may wish to consider the use of a more conservative site-specific objective. Conversely, where certain sensitive species are historically absent, the use of less conservative site-specific objectives for those particular areas could be justified. For example, in the derivation of the freshwater long-term CWQG, two data points fall below the long-term SSD 5th percentile value of 120 mg Cl^-/L. These include the 24h EC10s of 24 (Bringolf, 2010) and 42 (Gillis, 2009) mg Cl^-/L for two species of mantle lure spawning freshwater mussels (glochidia lifestage), including *Lampsilis fasciola* (COSEWIC special concern) and *Epioblasma torulosa rangiana* (COSEWIC endangered). In such cases, jurisdictions have the option of adopting the lower data point as the water quality guideline value in watersheds where, as in this example, endangered or special concern species occur and are considered an important component of the ecosystem.

With respect to deriving a site-specific hardness-adjusted water quality guideline value, it was decided by the CCME Water Quality Task Group that insufficient data was available in order to develop a hardness relationship for chronic toxicity. Therefore, a hardness based national CWQG was not developed. CCME will re-visit the chloride guidelines when sufficient studies are available. However, jurisdictions have the option of deriving site-specific hardness adjusted water quality criteria if they so choose. CCME has outlined several procedures to modify the national water quality guidelines to site-specific water quality guidelines or objectives to account for unique conditions and/or requirements at the site under investigation (CCME 1991; CCME 2003; Intrinsik 2010).

13.4 Naturally Saline Lakes

With respect to the saline lakes located within the northern Great Plains of Canada (stretching from Winnipeg, Maniboba westward to the Rocky mountain foothills), they are mostly dominated by sulphate or bicarbonate/carbonate anions, with variation in the predominant cations. Chloride dominated saline lakes are more rare and are located in northern Alberta (Derry *et al.,* 2003), with a few also located in the Saskatchewan River Delta (Hammer 1993) and on the interior plateau of British Columbia (Bos *et al.,* 1996). In the case of these naturally occuring saline lakes, the source of the ions present in these lakes is the underlying geology which impacts the ionic composition of groundwater. The shallow bedrock aquifers of southern Alberta are dominated by Na^+ and HCO_3^-, in Saskatchewan are dominated by Ca^{2+}, Mg^{2+}, Na^+ and SO_4^{2-}, and in western Manitoba are dominated by Ca^{2+}, Mg^{2+}, Na^+ and HCO_3^- (Last 1992). The deeper bedrock contains higher salinity water usually dominated by Na^+ and Cl^- (Last 1992). Therefore, prairie saline lakes 1) may not always be dominated by chloride inputs, and 2) may vary considerably in ion composition. As a result, prairie saline lakes can be classified as per Hammer (1986): subsaline (0.5-3 g/L TDS), hyposaline (3-20 g/L TDS), mesosaline (20-50 g/L TDS), and hypersaline (>50 g/L TDS). It may be best to apply the interm CWQG for salinity in cases such as these, which states that “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 level expected at that time and depth” (CCME, 1999b). This can account for changes in precipitation / evaporation patterns due to climate change over a temporal scale.

14.0 GUIDELINE SUMMARY

The short-term data met the toxicological and statistical requirements for the Type A guideline derivation method (Table 9.1). The log-Logistic model was used for short-term benchmark concentration derivation. As seen in Table 9.5, the data requirements for the SSD were surpassed, and a total of 52 data points from 52 species were used in the derivation of the benchmark concentration. Both LC50 and EC50 values were used in derivation.

The long-term data met the toxicological and statistical requirements for the Type A guideline derivation method (Table 9.1). The log-Logistic model was used for long-term guideline derivation. As seen in Table 9.11, the data requirements for the SSD were surpassed, and a total of 29 data points from 29 species were used in the derivation of the guideline.

Neither a short-term benchmark concentration nor a long-term guideline were developed for marine waters. Sea water salt concentrations are approximately 35,000 mg/L of which approximately 55% is chloride, which equates to 19,250 mg chloride/L. For this reason, brine discharges to marine environments were not evaluated.

Canadian Water Quality Guideline for the Chloride Ion[a] for the Protection of Aquatic Life

	Long-Term Canadian Water Quality Guideline[b] (mg Cl^-/L)	Short-Term Benchmark Concentration[c] (mg Cl^-/L)
Freshwater	120[d]	640
Marine	NRG	NRG

[a]Derived from toxicity tests utilizing both $CaCl_2$ and NaCl salts
[b]Derived with mostly no- and some low-effect data and are intended to protect against negative effects to aquatic ecosystem structure and function during indefinite exposures (e.g. abide by the guiding principle as per CCME 2007).
[c]Derived with severe-effects data (such as lethality) and are not intended to protect all components of aquatic ecosystem structure and function but rather to protect most species against lethality during severe but transient events (e.g. inappropriate application or disposal of the substance of concern).
[d]The long-term CWQG may not be protective of certain species of endangered and special concern freshwater mussels (as designated by the Committee on the Status of Endangered Wildlife in Canada, or COSEWIC). This specifically applies to two species; the wavy-rayed lampmussel (*Lampsilis fasciola*) (COSEWIC, 2010a) and the northern riffleshell mussel (*Epioblasma torulosa rangiana*) (COSEWIC, 2010b) (table below). The wavy-rayed lampmussel is indigenous to the lower Great Lakes and associated tributaries, specifically western Lake Erie, the Detroit River, Lake St. Clair and several southwestern Ontario streams. The northern riffleshell mussel is indigenous to the Ausable, Grand, Sydenham and Thames Rivers in Ontario, as well as the Lake St. Clair delta. Discussion with provincial regulators should occur if there is a need to develop more protective site specific values.
NRG = no recommended guideline

24h EC10 values (survival of glochidia) for 2 species of COSEWIC assessed freshwater mussels.

COSEWIC Assessed Species	24h EC10 (mg Cl^-/L)	95% Confidence Intervals	Reference
Lampsilis fasciola Wavy-rayed lampmussel (COSEWIC special concern)	24	-79[1], 127	Bringolf, 2010
Epioblasma torulosa rangiana Northern riffleshell mussel (COSEWIC endangered)	42	24, 57	Gillis, 2009

[1] The negative lower fiducial limit is an artefact of the statistics. Biologically this can be interpreted as meaning that a 10% effect can be observed between a concentration of 0 and the upper 95% confidence limit. Therefore, the effect is not significantly different from the control (no-effect concentration) and could be due to natural variability.

The short-term benchmark concentration and long-term CWQG for chloride are set to provide protection for short- and long-term exposure periods, respectively. They are based on generic environmental fate and behaviour and toxicity data. The guideline is a conservative value below which all forms of aquatic life, during all life stages and in all Canadian aquatic systems, should be protected. Because the guideline is not corrected for any toxicity modifying factors (e.g. hardness), it is a generic value that does not take into account any site-specific factors. Moreover, since it is mostly based on toxicity tests using naïve (i.e., non-tolerant) laboratory organisms, the guideline may not be relevant for areas with a naturally elevated concentration of chloride and associated adapted ecological community. Thus, if an exceedence of the guideline is observed (due to anthropogenically enriched water or because of elevated natural background concentrations), it does not necessarily suggest that toxic effects will be observed, but rather indicates the need to determine whether or not there is a potential for adverse environmental effects. In some situations, such as where an exceedence is observed, it may be necessary or advantageous to derive a site-specific guideline that takes into account local conditions (water chemistry such as hardness, natural background concentration, genetically adapted organisms, community structure).

The guideline should be used as a screening and management tool to ensure that chloride does not lead to the degradation of the aquatic environment. The CWQG for chloride could, for example, be the basis for the derivation of site-specific guidelines and objectives (derived with site-specific data as well as consideration of technological, site-specific, socioeconomic or management factors).

15.0 REFERENCES

Abegg, R. 1949. Some implications from the changes in the mucous covering of fish subjected to salt solutions. Proceedings of the Louisiana Academy of Science. 12: 7-10.

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*). Journal of Fisheries Research Board of Canada. 33: 203-208.

Al-Daham, N.K., and M.N. Bhatti. 1977. Salinity Tolerance of *Gambusia affinis* (Baird & Girard) and *Heteropneustes fossilis* (Bloch). Journal of Fisheries Biology. 11:309-313. Aldenberg, T., Jaworska, J.S., and Traas, T.P. 2002. Normal species sensitivity distributions and probabilistic ecological risk assessment. In: L. Posthuma, G.W. Suter II and T.P. Traas (Eds.), Species Sensitivity Distributions in Ecotoxicology. CRC Press LLC, Boca Raton, Florida. pp. 49-102.

Allen, E.W. 2008. Process water treatment in Canada's oil sands industry: I. Target pollutants and treatment objectives. Journal of Environmental Science and Engineering. 7:P123-138.

Anderson, B.G. 1946. The Toxicity Thresholds of Various Sodium Salts Determined by the Use of *Daphnia magna.* Sewage Works Journal. 18(1):82-87. (A: 2130).

Anderson, B.G. 1948a. The apparent thresholds of toxicity to *Daphnia magna* for chlorides of various metals when added to Lake Erie water. Transactions of the American Fisheries Society. 78: 96-113.

Anderson, B.G., Chandler, B.G., Andrews, T.F., and Jahoda, W.J. 1948b. The evaluation of aquatic invertebrates as assay organisms for the determination of the toxicity of industrial wastes. American Petroleum Institute Project Final Report 51.

Aragao, M.A., and Pereira, E.V. 2003. Sensitivity of *Ceriodaphnia dubia* of different ages to sodium chloride. Bulletin of Environmental Contamination and Toxicology. 70: 1247-1250.

Arambasic, M.B., S. Bjelic and G. Subakov. 1995. Acute toxicity of heavy metals (copper, lead, zinc), phenol and sodium on *Allium cepa* L., and *Daphnia magna* St.: comparative investigations and the practical applications. Water Research 29: 497-503.

Augspurger, T., A.E. Keller, M.C. Black, W.G. Cope and F.J. Dwyer. 2003. Water quality guidance for protection of freshwater mussels (*Unionidae*) from ammonia exposure. Environmental Toxicology and Chemistry. 22:2569-2575.

Bailer, J.A., and Oris, J.T. 1997. Estimating inhibition concentrations for different response scales using generalized linear models. Environmental Toxicology and Chemistry 16: 1554-1559.

Barnhart, M. C., W. R. Haag, and W. N. Roston. 2008. Adaptations to host infection and larval parasitism in Unionoida. Journal of the North American Benthological Society 27:370-394.

Bartlett, A. 2009. An assessment of the chronic toxicity of sodium chloride to *Hyalella azteca*. Unpublished data presented at the 1st International Conference on Urban Drainage and Road Salt Management in Cold Climates: Advances In Best Practices. University of Waterloo, May 26, 2009.

Baudoin, M.F. and P. Scoppa. 1974. Acute Toxicity of Various Metals to Freshwater Zooplankton. Bulletin of Environmental Contamination and Toxicology 12(6):745-751.

Beak International Inc. 1999. Ecotoxicology test results. Unpublished report for M.S. Evans, Environment Canada, National Water Research Institute.

Beamish, R.J., and Van Loon, J.C. 1977. Precipitation loading of acid and heavy metals to a small acid lake near Sudbury, Ontario. Journal of the Fisheries Research Board of Canada. 34: 649. Cited In: Health Canada, 1987.

Bell, R.T. 1996. Evaporites; in Geology of Canadian Mineral Deposit Types, (ed.) O.R. Eckstrand, W.D. Sinclair and R.I. Thorpe; Geological Survey of Canada, Geology of Canada, no. 8, p. 121-127.

Berube, M. 2009(personal communication). Surface water chloride monitoring data. Ministere du Developpement durable, de l'Environnment et des Parcs.

Biesinger, K.E., and G.M. Christensen. 1972. Effects of Various Metals on Survival, Growth, Reproduction, and Metabolism of *Daphnia magna*. Journal of the Fisheries Research Board of Canada. 29(12):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 warm water species of aquatic life in the Commonwealth of Kentucky. Memorandum of Agreement No. 5429. Kentucky Natural Resources and Environmental Protection Cabinet. Lexington, KY.

Black, H.H. 1950. Federal industrial pollution studies. Sewage and Industrial Wastes 22: 1049-1059.
Brauner, C.J., Wilson, J., Kamunde, C., and Wood, C.M. 2003. Water Chloride Porvides Partial Protection during Chronic Exposure to Waterbourne Silver in Rainbow Trout (*Oncorhynchus mykiss).* Physiological and Biochemical Zoology. 76(6): 803-815.
Bright, D.A. and J. Addison. 2002. Derivation of matrix soil standards for salt under the British Columbia Contaminated Sites Regulation. Royal Roads University. Prepared for BC Ministry of Water, Land and Air Protection, BC Ministry of Transportation and Highways, BC Buildings Corp., and the Canadian Association of Petroleum Producers. Victoria, BC.
Bringolf, R.B., W.G. Cope, C.B. Eads, P.R. Lazaro, M.C. Barnhart, and D. Shea. 2007. Acute and Chronic Toxicity of Technical grade Pesticides to Glochidia and Juveniles of Freshwater Mussels (Unionidae). Environmental Toxicology and Chemistry. 26:2086-2093.
Bringolf, R.B. 2010. Email to M. Nowierski June 14. Calculation of <EC50 effect concentrations from data presented in Bringolf *et al.*, 2007.
Bubeck, R.C., W.H. Diment, B.L. Deck, A.L. Baldwin and S.D. Lipton. 1971. Runoff of deicing salts: effects on Irondequoit Bay, Rochester, New York. Science. 172: 1128-1132.
Bubeck, R.C., W.W. Staubitz, A.D. Weidermann, and L.P. Spittal. 1995. Physical and chemical limnology of Ides Cove near Rochester, New York, 1970-1982. US Geological Survey, Water-Resources Investigations Report 92-4052. 45 p.
Buchmann, K., A. Felsing, and H.C. Slotved. 1992. Effects of Metrifonate, Sodium Chloride and Bithionol on an European Population of the Gill Parasitic *Monogeneans Pseudodactylogyrus* spp. Bulletin of the European Association of Fish Pathology. 12(2):57-60. Cited In: Bright and Addison, 2002.
Buckley, J.A., K.P. Rustagi and J.D. Laughlin. 1996. Response of *Lemna minor* to sodium chloride and a statistical analysis of continuous measurements for EC50 and 95% confidence limits calculation. Bulletin of Environmental Contamination and Toxicoogy. 57(6): 1003-1008.
Calleja, M.C., G. Persoone and Geladi, P. 1994. Comparative acute toxicity of the first 50 multicentre evaluation of in vitro cytotoxicity chemicals to aquatic non-vertebrates. Archives of Environmental Contamination and Toxicology. 26: 69-78.
Camargo, J.A., and Tarazona, J.V. 1991. Short-term toxicity of fluoride ion in soft water to rainbow trout and brown trout. Chemosphere 22: 605-611.
Canadian Centre for Occupational Health and Safety (CCOHS). 1991. Cheminfo: Chemical Safety Sheets. Kluwer Academic Publishers, p.794.
Canadian Council of Ministers of the Environment (CCME). 1987. Canadian Water Quality Guidelines. Accessed on January 21, 2010 at the following website: http://www.ccme.ca/assets/pdf/cwqg_pn_1040.pdf
CANMET (Canada Centre for Mineral and Energy Technology). 1991. Summary Report No. 14: Salt and potash. Mineral Sciences Laboratories. November 1991.
CCME. 1991. Appendix IX— A protocol for the derivation of water quality guidelines for the protection of aquatic life (April 1991). In: Canadian water quality guidelines, Canadian Council of Resource and Environment Ministers, 1987. Prepared by the Task Force on Water Quality Guidelines. [Updated and reprinted with minor revisions and editorial changes in Canadian environmental quality guidelines, Chapter 4, Canadian Council of Ministers of the Environment, 1999, Winnipeg.]
CCME. 1999. Canadian Environmental Quality Guidelines. Winnipeg, Manitoba. Canadian Council of Ministers of the Environment.
CCME. 1999b. Canadian water quality guidelines for the protection for the protection of Aquatic life: Salinity (marine). In: Canadian environmental quality guidelines, 1999, Canadian Council of Ministers of the Environment, Winnipeg.
CCME. 2003. Canadian water quality guidelines for the protection of aquatic life: Guidance on the Site-Specific Application of Water Quality Guidelines in Canada: Procedures for Deriving Numerical Water Quality Objectives. In: Canadian environmental quality guidelines, 1999, Canadian Council of Ministers of the Environment, Winnipeg.
CCME. 2007. A Protocol for the Derivation of Water Quality Guidelines for the Protection of Aquatic Life. Canadian Council of Ministers of the Environment. Draft for Public Review. April 27, 2007.
CCME. 2010a. Canadian Water Quality Guidelines : Cadmium. Scientific

Criteria Document. Canadian Council of Ministers of the Environment, Winnipeg. Draft – not yet published.
CCME. 2010b. Canadian Water Quality Guidelines : Zinc. Scientific Criteria Document. Canadian Council of Ministers of the Environment, Winnipeg. Draft – not yet published.
CEMA (Cumulative Environmental Management Association). 2003. Development of Reach Specific Water Quality Objectives for Variables of Concern in the Lower Athabasca River: Identification of Variables of Concern and Assessment of the Adequacy of Available Guidelines. Prepared for the Water Quality Task Group of the Surface Water Working Group of the Cumulative Environmental Management Association.
Canadian Environmental Protection Act (CEPA). 1999. Domestic Substances List Categorization and Screening Program. Categorization criteria can be accessed at: http://www.ec.gc.ca/substances/ese/eng/dsl/cat_criteria_process.cfm
Cannon, C. 2009 (personal communication). Supervisor, Major Ions and Nutrients National Laboratory for Environmental Testing, Environment Canada.
Carrol JJ, SJ Ellis, WS Oliver. 1979. Influence of hardness constituents on the acute toxicity of cadmium to brook trout (*Salvelinus fontinalis*). Bulletin of Environmental Contamination and Toxicology. 22:575-581.
Celis, M.P., B. Keller and N.D. Yan. 2009. A Negative Impact Turned Positive for Ecological Recovery in Sudbury Lakes. Presentation at the 1st International Conference on Urban Drainage and Road Salt Manageement in Cold Climates: Advances in Best Practices. University of Waterloo, Waterloo, Ontario, Canada. May 25-27, 2009.
Chan, C.H., and Clignett, A.H. 1978. Short-term variation of the chemical composition of the Niagara River. Environment Canada Inland Waters Directorate, Ontario Region, 30 pp. Cited In: Evans and Frick, 2001.
Chapra, S.C., A. Dove and D.C. Rockwell. 2009. Great lakes chloride trends: long-term mass-balance and loading analysis. Journal of Great Lakes Research. 35:272:284.
Charles, A. M., Markich, S. J., Stauber, J. L., and De Filippis, L. F. 2002. The effect of water hardness on the toxicity of uranium to a tropical freshwater alga (*Chlorella* sp.). Aquatic Toxicology 60: 61-73.
Clair, T.A., I.F. Dennis, D.A. Scruton and M. Gilliss. Freshwater acidifcation research in Atlantic Canada : a review of results and predictions for the future. Environmental Review. 15:153-167.
Clemens, H.P., and W.H. Jones. 1954. Toxicity of Brine Water from Oil Wells. Transactions of the American Fisheries Society. 84:97-109.
Collins, S.J., and R.W. Russell. 2009. Toxicity of road salt to Nova Scotia amphibians. Environmental Pollution. 157:320-324.
Cooney, J.D., *et al*., 1992. Effects of Environmental and Experimental Design Factors on Culturing and Toxicity Testing on *Ceriodaphnia dubia*. Environmental Toxicology and Chemistry. 11:839-850.
Cope, W.G., R.B. Bringolf, D.B. Buchwalter, T.J. Newton, C.G. Ingersoll, N. Wang, T. Augspurger, F.J. Dwyer, M.C. Barnhart, R.J. Neves, and E. Hammer. 2008. Differential exposure, duration, and sensitivity of unionoidean bivalve life stages to environmental contaminants. Journal of the North American Benthological Society. 27:451-462.
COSEWIC (Committee on the Status of Endangered Wildlife in Canada). 2010. COSEWIC wildlife species assessments (detailed version April 2010). Accessed at: http://www.cosepac.gc.ca/eng/sct0/index_e.cfm
Cowgill, U.M. 1987. Critical Analysis of Factors Affecting the Sensitivity of Zooplankton and the Reproducibility of Toxicity Test Results. Water Research. 21:1453-1462.
Cowgill, U.M., and D.P. Milazzo. 1990. The sensitivity of two cladocerans to water quality variables: salinity and hardness. Archives of Hydrobiology. 120: 185-196.
Cressman, C.P. III and Williams, P.L. 1994. Reference Toxicants for Toxicity Testing Using *Caenorhabditis elegans* in Aquatic Media In: F J Dwyer, T R Doane, and M L Hinman (Eds), Environmental Toxicology and Risk Assessment: Modeling and Risk Assessment, 6th Volume, ASTM STP 1317. Cited In: Bright and Addison, 2002.

Crisinel, A., L. Delaunay, D. Rossel, J. Tarradellas, H. Meyer, H. Saiah, P. Vogel, C. Delisle and C. Blaise, 1994. Cyst-Based Ecotoxicological Tests Using Anostracans: Comparison of Two Species of *Streptocephalus*. Environmental Toxicology and Water Quality. 9(4):317- 326. Cited In: Bright and Addison, 2002.

Crowther, R.A., and H.B.N. Hynes. 1977. The effect of road deicing salt on the drift of stream benthos. Envioronmental Pollution. 14: 113-126.

Davies, T.D., and K.J. Hall. 2007. Importance of Calcium in Modifying the Acute Toxicity of Sodium Sulphate to *Hyalella azteca* and *Daphnia magna*. Environmental Toxicology and Chemistry. 26:1243-1247.

DeGraeve, G.M., J.D. Cooney, B.H. Marsh, T.L. Pollock, N.G. Reichenbach. 1992. Variability in the performance of the 7 day *Ceriodaphnia dubia* survival and reproduction test: an intra- and inter-laboratory comparison. Environmental Toxicology and Chemistry. 11: 851-866.

Denoel, M., Bichot, M., Ficetola, G.F., Delcourt, J., Ylieff, M., Kestemont, P. and Poncin, P. 2010. Cumulative effects of road deicing salt on amphibian behaviour. Aquatic Toxicology. 99:275-280.

Derry, A.M., E.E. Prepas, and P.D.N. Hebert. 2003. A comparison of zooplankton communities in saline lakewater with variable anion composition. Hydrobiologia. 505:199-215.

Dettner, K. and W. Peters. (1999). Lehrbuch der Entomologie. Gustav Fischer Verlag, Stuttgart, Germany. In: Holland *et al.*, 2010.

Diamond, J.M., E.L. Winchester, D.G. Mackler, and D. Gruber. 1992. Use of the Mayfly *Stenonema modestum* (Heptageniidae) in Subacute Toxicity Assessments. Environmental Toxicology and Chemistry 11(3):415-425.

Doe, K.G. 2010. Email to M. Nowierski November 2. Results of a 108 day exposure of the Northern leopard frog (*Rana pipiens*) to NaCl. Moncton Aquatic Toxicity Laboratory, Environment Canada.

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.

Dougherty, C.K. and G.R. Smith. 2006. Acute effects of road de-icers on the tadpoles of three anurans. Applied Herpetology. 3:87-93.

Dow, S., K. Pitts, P.N. Saucedo and K. Murray. 2010. Summary of Zequanox Biobox Trials at DeCew II Generation Facility, St. Catharine's, Ontario, Canada. Prepared for Niagara Plant Group, Ontario Power Generation. December 13, 2010.

Dowden, B.F., and H.J. Bennett. 1965. Toxicity of selected chemicals to certain animals. Journal of The Water Pollution Control Federation. 37: 1308-1316.

Droste R.L., and Johnson, A.C. 1993. Urban snow dump quality and pollution reduction in snowmelt by sedimentation. Canadian Journal of Civil Engineering. 20: 9-21. Cited In: Evans and Frick, 2001.

Duboudin, C., P. Ciffroy and H. Maugaud. 2004. Effects of data manipulation and statistical methods on species sensitivity distributions. Environmental Toxicology and Chemistry 23(2):489-499.

Dumont, M. 2008. Canadian Minerals Yearbook (CMY) 2008 - Salt. Minerals and Metals Sector, Natural Resources Canada. Available on-line at: http://www.nrcan-rncan.gc.ca/mms-smm/busi-indu/cmy-amc/2008revu/htm-com/sal-sel-eng.htm

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*). The Progressive Fish-Culturalist. 10:105-106. Cited In: Iowa Chloride Criteria Development Reference List May 2009.

Eli Lilly and Company. 2001. Material Safety Data Sheet, Sodium Chloride Tablets. Indianapolis, IN. Available on-line at: http://ehs.lilly.com/msds/msds_sodium_chloride_tablets.html

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, Bull. U.S. Bur. Fish. 18: 363-437. Cited In: Anderson, 1948 and Cited In: McKee and Wolf, 1963 and Cited In: Evans and Frick.

Elphick J.R.F., K.D. Bergh and H.C. Bailey. 2011. Chronic toxicity of chloride to freshwater species: effects of hardness and implications for water quality guidelines. Environmental Toxicology and Chemistry. 30:239-246.

ENVIRON International Corporation. 2009. Chloride toxicity test results. Prepared for: Iowa Water Pollution Control Association. Project Number: #20-22235A.

Environment Canada. 2001. Synopsis of Priority Substances List Assessment Report. Road Salts. Existing Substances Division, Environment Canada.
Québec, Canada. Available on-line at:
www.ec.gc.ca/substances/ese/eng/psap/final/roadsalts.cfm

Environment Canada / Health Canada (EC/HC). 2001b. Priority Substances List Assessment Report. Road Salts. CEPA, 1999.
Environment Canada. 2009. Pacific and Yukon Water Quality Monitoring & Surveillance Program website. http://waterquality.ec.gc.ca/EN/home.htm - Accessed on December 15, 2009.
Environment Canada. 2004. Code of Practice for the Environmental Management of Road Salts. EPS 1/CC/5.
Environment Canada, 2005**.** Profile of Chloride-based Dust Suppressants Used in Canada**.** Accessed at: http://www.ec.gc.ca/nopp/roadsalt/reports/en/profile.cfm)
Evans, M., and Frick, C. 2001. The Effects of Road Salts on Aquatic Ecosystems. NWRI Contribution No. 02-308. Environment Canada, National Water Research Institute, Saskatoon, SK.
Everall NC, NAA Macfarlane, RW Sedgwick. 1989. The interactions of water hardness and pH with the acute toxicity of zinc to the brown trout *Salmo trutta*. Journal of Fish Biology. 35:27-36.
Fairchild, E. J. 1955. Low dissolved oxygen effects upon the toxicity of certain inorganics salts to the aquatic invertebrate *Daphnia magna*. Louisiana State Unvi. Engineering Experiment Station Bull No. 51. 95 pp. Cited In: McKee and Wolf, 1963.
Forbes, V.E. and P. Calow. 1999. Is the per capita rate of increase a good measure of population-level effects in ecotoxicology? Environmental Toxicology and Chemistry. 18(7): 1544-1556.
Foster, A.C., and Maun. M.A. 1978. Concentration of highway deicing agents along roadsides near London. Canadian Journal of Botany 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. Cited In: Anderson, 1948.
Fraser, A.S. 1981. Salt in the Great Lakes. Environment Canada, National Water Research Institute. Cited In: Evans and Frick, 2001.
Free, B.M., and Mulamoottil, G.G., 1983. The Limnology of Lake Wabukayne, a storm-water impoundment. Water Research Bulletin. 19: 821-827.
Freedman, B., J. Kerekes and G. Howell. 1989. Patterns of water chemistry among twenty seven oligotrophic lakes in Kejimkujik National Park, Nova Scotia. Water Air Soil Pollut. 46: 119–130. Cited In: EC/HC 2001b.
FHWA (Federal Highway Administration). 1999. Accelerating Admixtures. U.S. Department of Transportation, Federal Highway Administration (FHWA), Materials Group, Washington, DC. Available on-line at: http://www.fhwa.dot.gov/infrastructure/materialsgrp/acclerat.htm
Garibay, R., and S. Hall. 2004. Chloride Threshold Recommendations for the Protection of Aquatic Life in the Upper Santa Clara River. The Advent Group, Brentwood, TN. Attachment 8: NaCl Testing with Three-Spined Stickleback; Attachment 9: NaCl Testing with Chorus Frog Tadpoles.
Garrey, W.E. 1916. The resistance of freshwater fish to changes of osmostic and chemical conditions. American Journal of Physiology 39: 313-329. Cited In: Hammer, 1977, and Cited In: Doudoroff and Katz, 1953, and Cited In: Evans and Frick, 2001.
Gillis PL, 2011. Assessing the Toxicity of Sodium Chloride to the Glochidia of Freshwater Mussels: Implications for Salinization of Surface Waters. Environmental Pollution. 159:1702-1708.
Gillis, P.L. 2009. Email to M. Nowierski June 26. Calculation of <EC25 effect concentrations from data presented in Gillis 2011.
Gillis, P.L., R.J. Mitchell, A.S. Schwalb, K.A. McNichols, G.L. Mackie, C.M. Wood and J.D. Ackerman. 2008. Sensitivity of the glochidia (larvae) of freshwater mussels to copper: assessing the effect of water hardness and dissolved organic carbon on the sensitivity of endangered species. Aquatic Toxicology. 88:137-145.
GLEC (Great Lakes Environmental Center) and INHS (Illinois Natural History Survey). 2008. Acute toxicity of chloride to select freshwater invertebrates. Prepared for the U.S. Environmental Protection Agency. EPA Contract Number: 68-C-04-006. October 28, 2008.
Goetsch, P.A., and C.G. Palmer. 1997. Salinity Tolerances of Selected Macroinvertebrates of the Sabie River, Kruger National Park, South Africa. Archives of Environmental Contamination and Toxicology. 32(1):32- 41.
Gosh, A.K., and R.N. Pal. 1969. Toxicity of four therapeutic compounds to fly of Indian major carps. Fish. Technol. 6: 120-123.

Grizzle, J.M., and Mauldin, J.C. (II). 1995. Effect of calcium on acute toxicity of sodium chloride to larval and newly transformed newly striped bass. Journal of Aquatic Animal Health 7: 298-303.
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 in the Canadian Environment National Research Council Environmental Secretariat. Ottawa, Canada. In: Evans and Frick, 2001.
Hammer, U.T. 1993. Zooplankton distribution and abundance in saline lakes of Alberta and Saskatchewan, Canada. International Journal of Salt Lake Research. 2:111-132.
Hammer, U.T. 1994. Life and times of five Saskatchewan saline meromictic lakes. Internationale Revue der gesamten Hydrobiologie und Hydrographie. 79(2):235-248.
Hanes, R. E., L.W. Zelazny, and R.E. Blaser. 1970. Effects of Deicing Salts on Water Quality and Biota: Literature Review an Recommended Research. Washington, D.C. Highway Research Board, National Academy of Sciences – National Academy of Engineering, 91. 70 pp.
Harmon, S.M., Specht, W.L., Chandler, G.T. 2003. A Comparison of the Daphnids *Ceriodaphnia dubia* and *Daphnia ambigua* for their utilization in routine toxicity testing in the Southeastern United States. Archives of Environmental Contamination and Toxicology 45: 79-85.
Hassell, K.L., Kefford, B.J., Nugegoda, D. 2006. Sub-lethal and chronic salinity tolerances of three freshwater insects: *Cloeon* Sp. And *Centroptilum* sp. (Ephemeroptera: Baetidae) and *Chironomus* sp. (Diptera: Chironomidae). The Journal of Experimental biology 209: 4024-4032.
Health Canada. 1979. Guidelines For Canadian Drinking Water Quality - Supporting Documents – Hardness. Accessible at: http://www.hc-sc.gc.ca/ewh-semt/pubs/water-eau/hardness-durete/index-eng.php
Health Canada. 1987. Guidelines for Canadian Drinking Water Quality: Chloride. Available on-line at: http://www.hc-sc.gc.ca/ewh-semt/alt_formats/hecs-sesc/pdf/pubs/water-eau/chloride-chlorure/chloride-chlorure-eng.pdf
Health Canada. 1987. Chloride. Available from: http://www.hc-sc.gc.ca/ewh-semt/pubs/water-eau/chloride-chlorure/index-eng.php (accessed December 2009).
Heijerick DG, KAC De Schamphelaere, CR Janssen. 2002. Predicting acute zinc toxicity for *Daphnia magna* as a function of key water chemistry characteristics: development and validation of a biotic ligand model. Environmental Toxicology and Chemistry. 21:1309-1315.
Herbst , D.B. 2001. Gradients of salinity stress, environmental stability and water chemistry as a templet for defining habitat types and physiological strategies in inland salt waters. Hydrobiologia, 466:209-219.
Hinton, M.J., and A.G. Eversole. 1978. Toxicity of ten commonly used chemicals to American eels. Agricultural Experiment Station Technical Contribution 1595. Texas A&M University. College Station, TX.
Hoke, R.A., W.R. Gala, J.B. Drake, J.P. Giesy, and S. Flegler. 1992. Bicarbonate as a Potential Confounding Factor in Cladoceran Toxicity Assessments on Pore Water from Contaminated Sediments. Canadian Journal of Fisheries and Aquatic Sciences. 49:1633-1640.
Holland, A.J., A.K. Gordon and W.J. Muller. 2010. Osmoregulation in freshwater invertebrates in response to exposure to salt pollution. Report to the Water Research Commission. Unilever Centre for Environmental Water Quality, Institute for Water Research, Rhodes University, Grahamstown, South Africa. December 2010. 60 pp.
Howard, K.W.F. and J. Haynes. 1993. Groundwater contamination due to road salt deicing chemicals – salt balance implications. Geoscience Canada. 20(1):1-8. Cited In: Meriano *et al*., 2009.
HSDB (Hazardous Substances Data Bank). 2007a. Assessment record for sodium chloride. Hazardous Substances Databank Number: 6368. Last Revision Date: 20070604. Accessed at: http://toxnet.nlm.nih.gov/cgi-bin/sis/search/f?./temp/~DxMoZb:1
HSDB. 2007b. Assessment record for potassium chloride. Hazardous Substances Databank Number: 1252. Last Revision Date: 20070604. Accessed at: http://toxnet.nlm.nih.gov/cgi-bin/sis/search/f?./temp/~AEaJZf:1
HSDB. 2003a. Assessment record for calcium chloride. Hazardous Substances Databank Number: 923. Last Revision Date: 20030305. Accessed at: http://toxnet.nlm.nih.gov/cgi-bin/sis/search/f?./temp/~lSTuP6:1
HSDB. 2003b. Assessment record for magnesium chloride. Hazardous

Substances Databank Number: 657. Last Revision Date: 20030912. Accessed at: http://toxnet.nlm.nih.gov/cgi-bin/sis/search/f?./temp/~nfDs8S:1

Hughes, J.S. 1973. Acute Toxicity of Thirty Chemicals to Striped Bass (*Morone saxatilis*). La Dep. Wildl. Fish. 318: -343-2417:15 p.

Intrinsik Science. 2010. Synthesis of research and methods for developing water quality guidelines for the Canadian environmental sustainability indicators (CESI) freshwater quality indicator. Prepared for Environment Canada, National Guidelines and Standards Office. February 2010.

Iowa Chloride Criteria Development Reference List. May 2009. C. Stephan, US EPA, personal communication.

Iowa DNR (Department of Natural Resources). 2009. Iowa's Water Ambient Monitoring Program: Water Quality Summary 2000-2008. Water Fact Sheet 2009-3. Accessed from: http://www.igsb.uiowa.edu/wqm/publications/fact%20sheets/default.htm

Iwao, T. 1936. Comprehensive investigation of the toxicity of various metals. Jiken Yakubutsugaku Zassi (Jour. Exp. Pharnacol.) (Japanese) 10: 357-380. Cited In: Doudoroff and Katz, 1953 and In Evans and Frick, 2001.

Jackman, P. 2010. Email to M. Nowierski December 9. Reference toxicant data for NaCl tested on various aquatic organisms. Moncton Aquatic Toxicity Laboratory, Environment Canada.

Jackson BP, PJ Lasier, WP Miller, PW Winger. 2000. Effects of calcium, magnesium and sodium on alleviating cadmium toxicity to *Hyalella azteca*. Bulletin of Environmental Contamination and Toxicology. 64:279-286.

Jacobson, P.J., R.J. Neves, D.S. Cherry, J.L Farris. 1997. Sensitivity of glochidial life stages of freshwater mussels (Bivalvia: Unionidae) to copper. Environmental Toxicology and Chemistry. 16:2384-2392.

Jeffries, D.S. 1997. 1997 Canadian acid rain assessment: The effects on Canada's lakes, rivers and wetlands. Vol. 3. Aquatic effects. Environment Canada, Ottawa, Ontario. Cited In: EC/HC 2001b.

Jones, J.R.E. 1940. A further study of the relation between toxicity and solution pressure, with *Polycelis nigra* as test animals. Journal of Experimental Biology17: 408-415.

Jones, J.R.E. 1941. A study of the relative toxicity of anions, with *Polycelis nigra* as test animals. Journal of Experimental Biology 18: 170-181.

Judd, J.H. 1970. Lake stratification caused by runoff from street deicing. Water Res. 4: 521-532.

Kanygina, A.V., and Lebedeva, M.P. 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 (December 1957). Cited In: McKee and Wolf, 1963.

Kayama M., A.M. Quoreshi, S. Kitaoka, Y. Kitahashi, Y. Sakamoto, Y. Maruyama, M. Kitao, and T. Koike. 2003. Effects of deicing salt on the vitality and health of two spruce species, *Picea abies* Karst.,and *Picea glehnii* Masters planted along roadsides in northern Japan. Environ. Pollut. 124: 127-137.

Karraker, N.E. and J.P. Gibbs. 2011. Road deicing salt irreversibly disrupts osmoregulation of salamander egg clutches. Environmental Pollution. 159:833-835.

Karraker, N.E. and J.P. Gibbs. 2011. Road deicing salt irreversibly disrupts osmoregulation of salamander egg clutches. Environmental Pollution. 159:833-835.

Kaushal, S.S., Groffman, P.M., Likens, G.E., Belt, K.T., Stack,, W.P., Kelly, V.R., Band, L.E., and Fisher, G.T. 2005. Increased salinization of fresh water in the northeastern United States. Proceedings of the National Academy of Sciences of the United States of America. 102: 13517-13520.

Keizer, P.D., D.C. Gordon, J.W. Rowell, R. McCurdy, D. Borgal, T.A. Clair, D. Taylor, J.D. Ogden and G.E.M. Hall. 1993. Synoptic water quality survey of Halifax/Dartmouth Metro Area lakes on April 16, 1991. Biological Services Branch, Scotia-Fundy Region, Department of Fisheries and Oceans. Cited In: EC/HC 2001b.

Keller, A.E. and D.S. Ruessler. 1997. The toxicity of malathion to unionid mussels: Relationship to expected environmental concentrations. Environmental Toxicology and Chemistry. 16:1028-1033.

Kelly, D., J. Underwood and J. Hansen. 1976. Influence of highway deicing salts upon water quality in Chocolate Lake, Halifax, Nova Scotia. Nova Scotia Department of the Environment, Halifax, Nova Scotia. 118 pp. Cited In: EC/HC 2001b.

Kerekes, J. and B. Freedman. 1989. Seasonal variation of water chemistry in oligotrophic streams and rivers in Kejimkujik National Park, NS. Water Air Soil Pollut. 46(1–4): 131–144. Cited In: EC/HC 2001b.

Kerekes, J., B. Freedman, S. Beauchamp and R. Tordon. 1989. Physical and chemical characteristics of three acidic oligotrophic lakes and their watersheds in Kejimkujik Park, Nova Scotia. Water Air Soil Pollut. 46: 99–117. Cited In: EC/HC 2001b.

Kerr, D.R., and Meador, J.P. 1996. Modeling dose response using generalized linear models. Environmental Toxicology and Chemistry 15: 395-401.

Kersey, K. 1981. Laboratory and field studies on the effects of road deicing salt on steam invertebrates SIC-9. R.J. Mackay, ed. University of Toronto, Toronto. Cited In: Evans and Frick, 2001.

Kessler, E. 1974. Physiological and biochemical contributions to the taxonomy of the genus Chlorella. IX: Salt tolerance as a taxonomic character. Archives of Microbiology. 100:51-56.

Khangarot, B.S. 1991. Toxicity of metals to a freshwater tubificid worm, *Tubifex tubifex* (Muller). Bull. Environ. Contam. Toxicol. 46: 906-912.

Khangarot, B.S., and P.K. Ray. 1989. Investigation of Correlation Between Physicochemical Properties of Metals and Their Toxicity to the Water Flea *Daphnia magn*a Straus. Ecotoxicol. Environ. Saf. 18(2):109-120. (

Khanna, N., C.P. Cressman II, C.P. Tatara, and P.L. Williams. 1997. Tolerance of the nematode *Caenorhabditid elegans* to pH, salinity, and hardness in aquatic media. Arch. Environ. Contam. Toxicol. 32: 110-114. Cited In: Evans and Frick, 2001.

Kilgour, B.W., B. Gharabaghi, L. Trudel, S. Jarvie and N. Perera. 2009. Ecological benefits of the road salt Code of Practice in the City of Toronto. Submitted to Environment Canada. March 23, 2009. Project Number 0052.

Kostecki, P.T., and J.J. Jones. 1983. The Effect of Osmotic and Ionosmotic Stresses on the Mortality of Rainbow Trout (*Salmo gairdner*i). Comp. Biochem. Physiol. 74A:773-775.

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.

Kuntz, J.E. and McBride, A. 1993. Salt in well water: is a softener in the background? Journal of Environmental Health. 55(5):13-16.

Lampert, Winfried and Sommer, Ulrich (1997). *Limnoecology: The Ecology of Lakes and Streams* (Oxford University Press, Oxford). Translation by James F. Haney. Accessed from Wikipedia website on 29 January 2010. http://en.wikipedia.org/wiki/Meromictic_lake

Lange, M., W. Gebauer, J. Markl and R. Nagel. 1995. Comparison of testing acute toxicity on embryo of zebrafish, *Brachydanio rerio* and RTG-2 cytotoxicity as possible alternatives to the acute fish test. Chemosphere 30: 2087-2102.

Lasier, P.J., and I.R. Hardin. 2010. Observed and predicted reproduction of *Ceriodaphnia dubia* exposed to chloride, sulphate and bicarbonate. Environmental Toxicology and Chemistry. 29:347-358.

Lasier, P.J., Winger, P.V., Hardin, I.R. 2006. Effects of hardness and alkalinity in culture and test waters on reproduction of *Ceriodaphnia dubia*. Environmental Toxicology and Chemistry 25: 2781-2786.

Lasier, P.J., P.V. Winger, and R.E. Reinert. 1997. Toxicity of Alkalinity to *Hyalella azteca*. Bull. Environ. Contam. Toxicol. 59:807-814.

Last, W.M., and F.M. Ginn. 2005. Saline systems of the Great Plains of western Canada: an overview of the limnogeology and paleolimnology. Saline Systems. 1:1-38.

LeClerc, E., and F. Devlaminck. 1950. Toxicological experiments with some products utilized in tanneries (estimated). Reprint from Centre Belge d'Etude et de Documentsdes Eaux 541. Cited In: McKee and Wolf, 1963, and in Evans and Frick, 2001.

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. Cited In: McKee and Wolf, 1963 and Cited In: Evans and Frick, 2001.

Legendre, P., D. Chodorowski, W. Chodorowska, P. Pichet and P. Potvin. 1980. Qualité des eaux: interprétation des données lacustres (1971-1977). Service de la qualité des eaux, Ministère de l'Environnement du Québec. 409 p.

Lewis, Jr. W. M. and D. P. Morris. 1986. Toxicity of nitrite to fish: A review. Transactions of the American Fisheries Society. 115(2): 183-195.

Lillius, H., T. Hastbacka, and B. Isomaa. 1995. A comparison of the toxicity of 30 reference chemicals to *Daphnia magna* and *Daphnia pulex*. Environ. Toxicol. Chem. 14: 2085-2088.

Little, A.D. 1971. Inorganic chemical pollution of freshwater. U.S. Environmental Protection Agency, Washington, DC (1971). Cited In: Health Canada 1987.

Lockheart, E.E., Tucker, C.L., and Merritt, M.C. 1955. The effect of water impurities on the flavor of brewed coffee. Food Res. 20: 598. Cited In: Health Canada, 1987.

Lowell, R.B., J.M. Culp, and F.J. Wrona. 1995. Toxicity testing with artificial stream: effects of differences in current velocity Environmental Toxicology and Chemistry. 14: 1209-1217.

Mackie, G.L. 1978. Effects of pollutants on natality of *Musculium securis* (Bivalvia: Pisidiidae). The Naurulus 92: 25-33.

MacLeod, C.L., Borcsik, M.P., and Jaffe, P.R. 1996. Effect of infiltrating solutions on the desorption of mercury from freshwater sediments. Environmental Technology 17: 465-475. Cited In: Evans and Frick, 2001.

Mahajan, C.L., S.D. Sharma, and Sharma, S.P. 1979. Tolerance of Aquatic Organisms to Chloride Salts. Indian J Exp Biol 17(11):1244-1245.

Markert, B. 1994. The biological system of the elements (BSE) for terrestrial plants (Glycophytes). Sci. Total Environ. 155: 221-228.

Marsalek, J. 2003. Road salts in urban storm water: An emerging issue in storm water management in cold climates. Wat. Sci. Tech. 48(9):61-70.

Mayer, T., Marsalek, J., and Delos Reyes, E. 1996. Nutrients and metal contaminants status of urban stormwater ponds. J. Lake and Resevior Management 12: 348-363.

Mayer, T., Rochfort, Q., Marsalek, J., Servos, M., Jurkovic, A., and McInnis, R. 1998. (draft) Effect of deicing salts on the chemistry and toxicity of highway runoff. Report submitted to the Road Salt Environmental Working Group, Chemical Evaluation Branch, Environmental Protection, Hull. 88 pp.

Mayer, T., Snodgrass, W.J., and Morin, D. 1999. Spatial Characterization of the Occurrence of Road Salts and Their Environmental Concentrations as Chlorides in Canadian Surface Waters and Benthic Sediments. Water Quality Research Journal of Canada 34: 545-574.

Mayer, T., Q. Rochfort, U. Borgmann, W. Snodgrass. 2008. Geochemistry and toxicity of sediment porewater in a salt-impacted urban stormwater detention pond. Env Poll 156:143-151.

Meador, J.P. 2000. An analysis in support of tissue and sediment based threshold concentrations of polychlorinated bipenyls (PCBs) to protect juvenile salmonids listed by the endangered species act. NOAA White Paper, Northwest Fisheries Science Center, Environmental Conservation Division, Seattle, Washington.

Meriano, M., N. Eyles and K.W.F. Howard. 2009. Hydrogeological impacts of road salt from Canada's busiest highway on a Lake Ontario watershed (Frenchmans' Bay) and lagoon, City of Pickering. Journal of Contaminant Hydrology. 107:66-81.

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, physicochemistry and biotics of sodium chloride dominated saline waters on the western shore of Lake Winnipegosis, Man. Can. Field-Nat. 106(4): 454–473. Cited In: EC/HC 2001b.

Meriano, M., N. Eyles, and K.W.F. Howard. 2009. Hydrogeologica impacts of road salt from Canada's busiest highway on a Lake Ontario watershed (Frenchman's Bay) and lagoon, city of Pickering. Journal of Contaminant Hydrology. 107:66-81.

Michigan Natural Features Inventory. 2004. Status of Northern Riffleshell (*Epioblasma torulosa rangiana* http://web4.msue.msu.edu/mnfi/abstracts/zoology/Epioblasma_torulosa_rangiana.pdf

Ministère des Transports du Québec. 1980. *Prévention contre la pollution du Lac à la Truite en hiver*. Document prepared by François Berard, *Office des autoroutes du Québec*. Document obtained from the Quebec Ministry of Transportation's document centre. Cited In: EC/HC 2001b.

Ministère des Transports du Québec. 1999. Personal communication from G. D'Astous (*Direction des Laurentides-Lanaudière*) and A. Décréon (*Service de l'aménagement des infrastructures et de l'environnement*) to D. Morin (Environment Canada) regarding the impact of road salt loadings on lac à la

Truite. Cited In: EC/HC 2001b.
MOE. 1999. Surface water monitoring and assessment: 1997 Lake Ontario report featuring a summary of tributary and nearshore conditions and trends for the Lake Ontario Basin. Environmental Monitoring and Reporting Branch. 49pp.
MOE. 2002. Drinking Water Surveillance Program Summary Report for 2000, 2001, and 2002. Ontario Ministry of the Environment. Available on-line at: http://www.ene.gov.on.ca/envision/water/dwsp/0002/index.htm
MOE. 2005. Provincial Water Quality Monitoring Data for Chloride - 1964 to 2005. Environmental Monitoring and Reporting Branch. Environmental Sciences and Standards Division.
MOE. 2009. Provincial Water Quality Monitoring Data for Chloride - 1964 to 2008. Environmental Monitoring and Reporting Branch. Environmental Sciences and Standards Division.
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. Cited In: Evans and Frick, 2001.
Moll. R.A., Rossman, R., Barres, J.A., and Horvath, F.J. 1992. Historical trends of chlorides in the Great Lakes. Pages 303-322 in F.M. D'Itri, ed. Chemical Deciers and the Environment. Lewis Publishers, Florida. Cited In: Evans and Frick, 2001.
Morin, D., and M. Perchanok. 2000. Road Salt Loadings in Canada. Document Submitted to the Environmental Resource Group for Road Salts, Commercial Chemicals and Evaluation Branch, Environment Canada. 85 pp.
Morris, T.J., D.J. McGoldrick, J.L. Metcalfe-Smith, D. Zanatta, and P.L. Gillis. Pre-COSEWIC assessment of the Wavyrayed Lampmussel (*Lampsilis fasciola*). Canadian Science Advisory Secretariat. Research Document 2008/083.
Mount, D.R., D.D. Gulley, J.R. Hockett, T.D. Garrison and J.M. Evans. 1997. Statistical models to predict the toxicity of major ions to *Ceriodaphia dubia, Daphnia magna,* and *Pimephales promelas* (Fathead minnows). Environmental Toxicology and Chemistry. 16: 2009-2019.
Mulcrone, R. 2006a. "Elliptio complanata" (On-line), Animal Diversity Web. Accessed January 13, 2010 at http://animaldiversity.ummz.umich.edu/site/accounts/information/Elliptio_complanata.html
Mulcrone, R. 2006b. "Lampsilis siliquoidea" (On-line), Animal Diversity Web. Accessed January 13, 2010 at http://animaldiversity.ummz.umich.edu/site/accounts/information/Lampsilis_siliquoidea.html
Mulcrone, R. 2006c. "Lampsilis fasciola" (On-line), Animal Diversity Web. Accessed January 13, 2010 at http://animaldiversity.ummz.umich.edu/site/accounts/information/Lampsilis_fasciola.html
Myron L. Company. 2006. Material Safety Data Sheet Potassium Chloride. Carlsbad, California, USA. Available on-line at: http://www.myronl.com/PDF/msds/KCl_02_08_06-G.pdf
Nagpal. N.K., Levy, D.A, and Macdonald, D.D. 2003. Ambient Water Quality Guidelines for Chloride- Overview Report. British Columbia. Water, Air and Climate Change Branch.
NAQUADAT. 1985. Water Quality Branch, Inland Waters Directorate, Environment Canada, Ottawa (1985). National Water Quality Data Bank. Cited In: Health Canada, 1987.
Naumann, E. 1934. Uber die Beeinflussung con *Daphnia magna* durch einege Salze und Salzmischungen. K. fysiogr. Sallsk. Lund Forh., Bd. 4: 11-12. Cited In: Anderson, 1948.
Newman, M.C.and M.S. Aplin. 1992. Enhancing Toxicity Data Interpretation and Prediction of Ecological Risk with Survival Time Modeling: An Illustration Using Sodium Chloride Toxicity. Aquatic Toxicology. 23(2):85-96. Cited In: Bright and Addison, 2002.
Newman, M.C., Ownby, D.R., Mézin, L.C.A., Powell, D.C., Christensen, T.R.L., Lerberg, S.B., Anderson, B-A., and Padma, T.V. 2002. Species sensitivity distributions in ecological risk assessment: Distributional assumptions, alternate bootstrap techniques, and estimation of adequate number of species. In: Posthuma, L., G.W. Sutter II and T.P. Traas (Eds.). Species Sensitivity Distributions in Ecotoxicology. Lewis Publishers, New York, NY. Pp 119-132.
Northcote, T.G. and T.G. Halsey. 1969. Seasonal changes in the limnology of some meromictic lakes in southern British Columbia. J. Fish. Res. Board Can. 26: 1763–1787. Cited In: EC/HC 2001b.

NRC (National Research Council of Canada). 1977. The effects of alkali halides in the Canadian environment. NRC No. 15019, Associate Committee on Scientific Criteria for Environmental Quality, Ottawa. National Research Council of Canada. Cited In: Health Canada, 1987.

NRCAN (Natural Resources Canada). 1978. Hydrological Atlas of Canada. Water Quality of Surface Waters – hardness. Accessible at: http://atlas.nrcan.gc.ca/auth/english/maps/archives/hydrological_atlas_1978/water_quality/28_Water_Quality_Surface_Waters_1978/#download

Oberg, G. 2006. The biogeochemical cycle of chlorine. *Risk Assessment and cycling of natural organochlorines - Focus on Chlorine Science* special edition for the 16th SETAC meeting, 2006. Accessed at: http://www.eurochlor.org/upload/documents/document236.pdf

Oberg, G. and P. Sanden. 2005. Retention of chloride in soil and cycling of organic-matter bound chlorine. Hydrological Processes. 19(11):2123-2136.

OECD. 2001. SIDS Initial Assessment Report for Potassium Chloride CAS No: 7447-40-7. UNEP Publication. Available on-line at: http://www.inchem.org/documents/sids/sids/kchloride.pdf. [March 22, 2007].

Oliver, B.J., Milne, J.B., LaBarre, N. 1974. Chloride and lead in urban snow. J. Wat. Poll. Cont. Fed. 46: 766-791. Cited In: Evans and Frick, 2001.

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. Cited In: Doudoroff and Katz, 1953, and Cited In: Evans and Frick, 2001.

Pahdye, A.D., and H.V. Ghate. 1992. Sodium chloride and potassium chloride tolerance of difference stages of the frog, *Microhyla ornate.* Herpetological Journal 2: 18-23.

Palmer, C.G., *et al*., 2004. The Development of a Toxicity Database Using Freshwater Macroinvertebrates, and Its Application to the Protection of South African Water Resources. South African Journal of Science. 100:643-650 (with attached Table 1).

Paquin PR, JW Gorsuch, S Apte, GE Batley, KC Bowles, PGC Campbell, CG Delos, DM DiToro, RL Dwyer, F Galvez, RW Gensemer, GG Goss, C Hogstrand, CR Janssen, JC McGeer, RB Naddy, RC Playle, RC Santore, U Schneider, WA Stubblefield, CM Wood, and KB Wu. 2002. The biotic ligand model: a historical overview. Comparative Biochemistry and Physiology C. 133:3-35.

Part P, O Svanberg, A Kiessling. 1985. The availability of cadmium to perfused rainbow trout gills in different water qualities. Water Research. 19:427-434.

Patrick, R., J.J. Cairns., and A. Scheier. 1968. The relative sensitivity of diatoms, snails and fish to twenty common constitute of industrial wastes. The Progressive Fish-Culturist 30: 137-140.

Peredo-Alvarez, V.M., Sarma, S.S.S., and Nandini, S. 2003. Combined effect concentrations of algal food (*Chlorella vulgaris*) and salt (sodium chloride) on the population growth of *Brachionus calyciflorus* and *Brachionus patulus* (Rotifera). Revista de Biología Tropical 51: 399-408.

Perera N, Gharabaghi B, Noehammer P, Kilgour B. 2010. Road Salt Application in Highland Creek Watershed, Toronto, Ontario - Chloride Mass Balance Water Quality Research Journal of Canada. Vol. 45, no. 4, 2010.

Pettyjohn, W.A. 1971. Water pollution by oil-field brines and related industrial wastes in Ohio. Ohio J. Sci., 71: 257 (1971). Cited In: Health Canada, 1987.

Pettyjohn, W.A. 1972. Water quality in a stressed environment. Burgess Publishing Co., MN (1972). Cited In: Health Canada, 1987.

Phillips, A.M.J. 1944. The physiological effects of sodium chloride upon brook trout. Trans. Amer. Fish. Soc. 74: 297-309.

Phippen, B.W., D.C. Parks, L.G. Swain, R. Nordin, C.J.P. McKean, J.B. Holms, P.D. Warrington, R. Nijman, J. Deniseger and L. Erickson. 1996. A ten year Assessment of water quality in six acid-rain sensitive British Columbia lakes (1984–1994). British Columbia Ministry of Environment, Lands and Parks. 250 pp. Cited In: EC/HC 2001b.

Pickering, Q.H., J.M. Lazorchak and K.L. Winks. 1996. Subchronic sensitivity of one-, four-, and seven-day-old fathead minnow Environ. Toxicol. Chem. 15: 353-359.

Pinel-Alloul, B., D. Planas, R. Carignan and P. Magnan. 2002. Review of ecological impacts of forest fires and harvesting on lakes of the boreal ecozone in Québec.

Rev. Sci. Eau 15(1): 371-395.
PMRA (Pest Management Regulatory Agency). 2006. Proposed regulatory decision document – sodium chloride. PRDD2006-01. February 3, 2006.
Powers, E.B. 1917. The goldfish (*Carassius carassius)* as a test animals in the study of toxicity. ILL Biol. Monogr. 4(2): 1-73. Cited In: Evans and Frick, 2001.
Prud'Homme, M. 1986. Salt. Cited In: Canadian minerals yearbook 1985- review and outlook. Mineral Report 34, Mineral Resources Branch, Department of Energy, Mines and Resources, Ottawa (1986). Cited In: Health Canada, 1987.
Ralson, J.G., and Hamilton, F. 1987. Chloride inputs into Lake Ontario and the significance of deicing salts. Unpublished report. Water Resources Branch, Ontario Ministry of the Environment. 25 pp. Cited In: Evans and Frick, 2001.
Reimann, C. and P. de Caritat. 1998. Chemical elements in the environment. Berlin (Germany): Springer-Verlag. 398 p.
Rescan Environmental Services Ltd. 2007. Ekati Diamond Mine Proposed Discharge Criterion for the Sable Kimberlite Pipe Development. Toxicity Testing for Chloride: Appendix A- Data Report. Yellowknife, Northwest Territories and Vancouver, B.C.
Rescan Environmental Services Ltd. 2008. Ekati Diamond Mine: site specific water quality objective for chloride. October 2008. Project Number: 785-1 (0648-035). Yellowknife, Northwest Territories and Vancouver, B.C.
Reynoso, G.T., B.A. de Gamboa., and S.R. Mendoza. 1982. Induction of resistance to salinity in the freshwater alga *Chlamydomonad reinhardtii.* Environ. Sci. Res. 23: 531-534.
Roback, S.S. 1965. Environmental Requirements of Trichoptera. Cited In: C M Tarzwell (Ed), Biological Problems in Water Pollution, Trans 3rd Seminar, Aug 13-17, 1962, Tech Rep 999-WP25, U S Public Health Service, R A Taft Sanitary Engineering Center, Cincinnati, OH:118-126.
Rodney, S.I., D. Moore, and R.S. Teed. 2008. SSD Master – Determination of hazardous concentrations with species sensitivity distributions. Version 2.0 January 2008.
Rosicky, P., J. Kouril and I. Prikryl. 1987. The Effect of the Length of Short-Term Exposure to NaCl on the Tolerance of Early Common Carp and Silver Carp Fry to NaCl Baths. Bul Vyzk Ustav Ryb Hydrobiol Vodnany 23(1):8-13 (CZE) (ENG ABS).
Russell, R.W., and S.J. Collins. 2009. Amphibians as indicators of disturbance in forests: a progress report. A report prepared for: Nova Scotia Habitat Conservation Fund. Submitted January 30, 2009.
Russo, R. C., R. V. Thurston, and K. Emerson. 1981. Acute toxicity of nitrite to rainbow trout (*Salmo gairdneri*): Effects of pH, nitrite species and anion species. Canadian Journal of Fisheries and Aquatic Sciences. 38: 387-393.
Ruth, O. 2003. The effects of deicing in Helsinki urban streams, Southern Finland. Water Science and Technology. 48(9):33-43. Cited In: Meriano *et al*., 2009.
Rutherford, J.C. and B.J. Kefford. 2005. Effects of salinity on stream ecosystems: improving models for stream macroinvertebrates. CSIRO Land and Water Technical Report 22/05. CSIRO Land and Water, Canberra.
Salt Institute. 2011. Vaccum pan refining. Accessed at: http://www.saltinstitute.org/Production-industry/Production-technologies/Evaporated-salt-refined-salt/Vacuum-pan-refining
Sanzo, D., and Hecnar, S.J. 2006. Effect of road deicing salt (NaCl) on larval wood frogs (*Rana sylvatica)*. Environmental Pollution 140: 247-256.
Scanton, D. 1999. Personal communication (letter dated May 19, 1999). Ontario Ministry of Transporation,Toronto, Ontario. Cited In: Evans and Frick, 2001.
Schiewer, U. 1984. The ecophysiology of salt adaptation in freshwater blue-green algae. Limnologica. 15:555-558.
Schlekat, C.E., J.C. McGeer, R. Blust, U. Borgmann, K.V. Brix, N. Bury, Y. Couillard, R.L. Dwyer, S.N. Luoma, S. Robertson, K.G. Sappington, I. Schoeters, and D.T.H.M. Sijm. 2007. Bioaccumulation; hazard identification of metals and inorganic metal substances. In: Adams, W.J., Chapman, P.M. (eds.). Assessing the hazard of metals and inorganic metal substances in aquatic and terrestrial systems. Pensacola (FL): SETAC Publications, CRC Press. Chap 4: 55-87.
Schneider, W.J. 1970. Hydrologic implications of solid-waste disposal. U.S. Geol. Surv. Circ., 601-F: F1. Cited In: Health Canada, 1987.
Scott, W.S. 1980a. Road salt movement into two Toronto streams. Journal of the Environmental Engineering Division EE3: 547-560. Cited In: Evans and Frick, 2001.

Scott, W.S. 1980b. Deicing Salt Levels in Toronto Stream Banks and Roadside Soils. Bull. Environm. Contam. Toxicol. 25, 208-214. Cited In: Evans and Frick, 2001.

Setter, T.L., Greenway, H., and Kuo, J. 1982. Inhibition of cell division by high external NaCl concentrations in synchronized cultures of *Chlorella emersonii*. Aust. J. Plant. Physiol. 9: 170-196.

Seymour, D.T., A.G. Verbeek, S.E. Hrudey, and P.M. Fedorak. 1997. Acute Toxicity and Aqueous Solubility of Some Condensed Thiophenes and Their Microbial Metabolites. Environ. Toxicol. Chem. 16:658-665.

Smol, J.P., Brown. S.R., and McNeely, R.M. 1983. Cultural disturbances and trophic history of a small meromictic lake from Central Canada. Hydrobiol. 103: 125-130.

Snodgrass, J.W., R.E. Casey, D. Joseph, and J.A. Simon. 2008. Microcosm investigations of stormwater pond sediment toxicity to embryonic and larval amphibians: variation in sensitivity among species. Environmental Pollution. 154:291-297.

Soucek, D.J., and Kennedy, J. 2005. Effects of hardness, chloride, and acclimation on the acute toxicity of sulphate to freshwater invertebrates. Environmental Toxicology and Chemistry 24: 1204-1210.

Spehar, R.L. 1986. Criteria Document Data. Memorandum to D.J. Call. September 16. Cited In: Iowa Chloride Criteria Development Reference List May 2009.

Spehar, R.L. 1987. Criteria Document Data. Memorandum to C. Stephan. June 24. Cited In: Iowa Chloride Criteria Development Reference List May 2009.

St-Cyr, L. 2000. Rapport de présentation d'un plan de contrôle et d'un suivi de l'eutrophisation au lac Carré. Report presented to the Municipalité de Saint-Faustin-Lac-Carré. 48 p.

Stangenberg, M. 1975. The Influence of the Chemical Composition of Water on the Pike Perch (*Lucioperca lucioperca* L.) Fry From the Lake Gopio. Limnologica 9(3):421-426. Cited In: Bright and Addison, 2002.

Stanley, R.A. 1974. Toxicity of heavy metals and salts to Eurasian milfoil (*Myriophyllum spicatum* L.). Arch. Environ. Contam. Toxicol. 2(4): 331-341.

Stephan, C. 2009. Environmental Scientist, United States Environmental Protection Agency. Personal Communication.

Stom, D.I. and L.D. Zubareva. 1994. Comparative Resistance of *Daphnia* and *Epischura* to Toxic Substances in Acute Exposure. Hydrobiol J 30(3):35-38; Gidrobiol J 29(2):31-34 (1993).

Sutcliffe, D.W. 1961. 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. Cited In: Evans and Frick, 2001.

Suter, G.W., S.B. Norton and A. Fairbrother. 2005. Individuals versus organisms versus populations in the definition of ecological assessment endpoints. Integrated Environmental Assessment and Management. 1: 397-400.

Taraldsen, J.E. and T.J. Norberg-King. 1990. New method for determining effluent toxicity using Duckweed (*Lemna minor*). Environmental Toxicology and Chemistry. 9:761-767.

Teeter, J.W. 1965. Effects of sodium chloride on the sago pondweed. J. Wildl. Manage. 129: 838-845. Cited In: US EPA, 1988.

Thornton, K.W., and J.R. Sauer. 1972. Physiological effects of NaCl on *Chironomus attenuatus* (Diptera: Chironomidae). Annals of the Entomological Social 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(1):89-92.

Topping, M.S. and G.G.E. Scudder. 1977. Some physical and chemical features of saline lakes in central British Columbia. Syesis 10: 145–166. Cited In: EC/HC 2001b.

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.

US EPA (United States Environmental Protection Agency). 1976. Quality Criteria for Water. U.S EPA, Washington, D.C. Cited in: Evans and Frick, 2001. US EPA. 1988. Ambient water quality criteria for chloride. US EPA, Washington, D.C. Cited In: Evans and Frick, 2001. US EPA. 2002. *Ceriodaphnia dubia* chronic static renewal 6-day survival and reproduction test, method 1002.0, EPA-821-R-02-013.

US EPA. 1988. Ambient aquatic life water quality criteria for chloride. NTIS Number: PB88-175 047.

US EPA. 1991. Methods for Aquatic Toxicity Identification Evaluations. Phase I Toxicity Characterization Procedures. Second Edition. EPA/600/6-91/003.

US EPA. 2001. 2001 Update of ambient water quality criteria for cadmium. Washington, D.C., Office of Water.
US EPA. 2006. National Recommended Water Quality Criteria. U.S. Environmental Protection Agency, Office of Water, Office of Science and Technology (4304T).
US EPA. 2009. Draft 2009 Update Aquatic Life Ambient Water Quality Criteria for Ammonia – Freshwater. Supersedes 1999 update. Office of Water, EPA-822-D-09-001. December 2009.
US EPA. 2010. Final report on acute and chronic toxicity of nitrate, nitrite, boron, manganese, fluoride, chloride and sulfate to several aquatic animal species. U.S. Environmental Protection Agency, Office of Science and Technology, Health and Ecological Criteria Division, Region 5 Water Division. EPA-905-R-10-002. November 2010.
Valenti, T.W., D.S. Cherry, R.J. Neves, B.A. Locke, and J.J. Schmerfeld. 2007. Case Study: Sensitivity of Mussel Glochidia and Regulatory Test Organisms to Mercury and a Reference Toxicant. IN: Freshwater Bivalve Ecotoxicology. Farris, J.L., and J.H. Van Hassel (eds). CRC Press. pp. 351-367.
van Horn, W.M., J.B. Anderson., and M. Katz. 1949. The effect of kraft pulp mill wastes on some aquatic organisms. Tran. Amer. Fish. Soc. 79: 55-63.
Varsamos, S., C. Nebel and G. Charmantier. 2005. Ontogeny of osmoregulation in postembryonic fish: a review. Comparative Biochemistry and Physiology. Part A. 141:401-429.
Viertel, B. 1999. Salt tolerance of *Rana temporaria*: spawning site selection and survival during embryonic development (Amphibia, Anura). Amphibia-Reptilia. 20:161-171.
Vosyliene, M.Z., P. Baltrenas and A. Kazlauskiene. 2006. Toxicity of Road Maintenance Salts to Rainbow Trout *Oncorhynchus mykiss*. Ekologija 2:15-20.
Wallen, I.E., W.C. Greer, and R. Lasater. 1957. Toxicity to *Gambusia affinis* of certain pure chemicals in turbid waters. The Environmental Future 29: 695.
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, N. 2007. Email to C. Stephan. September 11. Cited In: Iowa Chloride Criteria Development Reference List May 2009.
Wang, N. and C.J. Ingersoll. 2010. Email to M. Nowierski December 20. Reference toxicity test data for NaCl and various aquatic invertebrates. United States Geological Survey.
Wang, J.S., Hunag, P.M., Liaw, W.K., and Hammer, U.T. 1991. Kinetics of desorption of mercury from selected freshwater sediments as influenced by chloride. Water, Air and Soil Pollution 56: 533. Cited In: Evans and Frick, 2001.
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) to D. Morin (Environment Canada, Existing Substances Branch, Hull). Communication between October 2000, and January, 2001. Cited In: Evans and Frick, 2001.
Watson-Lueng, T.L. 2002. Investigation of chloride concentrations and littoral invertebrate community structure in stormwater and non-stormwater ponds (Ontario, Canada). A thesis submitted for the degree Master of Science, Graduate Department of Zoology, University of Toronto.
Wedemeyer, G. A. and W. T. Yasutake. 1978. Prevention and treatment of nitrite toxicity in juvenile steelhead trout (*Salmo gairdneri*). J. Fish. Res. Board Can. 35: 822-827.
Welsh PG. 1996. Influence of dissolved organic carbon on the speciation, bioavailability and toxicity of metals to aquatic biota in soft water lakes. PhD Thesis. University of Waterloo. Waterloo, Ontario.
Westin, D. T. 1974. Nitrate and nitrite toxicity to salmonid fishes. Prog. Fish Cult. 36(2): 86-89.
Wetzel. 1983. Limnology. Saunders College Publishing, Toronto.
Wetzel, R.G. 2001. Limnology – Lake and river ecosystems, 3rd ed. London, San Diego: Academic Press. Chap. 13: 239-288.
Whitfield, P.H. and N.L. Wade. 1992. Monitoring transient water quality events electronically. Water Resource Bulletin. 28(4): 703–711. Cited In: EC/HC 2001b.
WHO (World Health Organization). 2003. Background document for development of WHO Guidelines for Drinking-water Quality: Chloride in Drinking-water. World Health Organization.
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. Transactions of the American Fisheries Society. 64: 81-86. Cited In: Evans and Frick, 2001.
Wikipedia. 2011. List of diamond mines. Accessed at: http://en.wikipedia.org/wiki/List_of_diamond_mines

Williams, D.D., Williams, N.E., and Cao, Y. 1999. Road salt contamination of groundwater in a major metropolitan area and development of a biological index to monitor its impact. Water Research. 34: 127- 138.

Wilson, P. 2009 (personal communication). Supervisor, Water Chemistry Unit General Chemistry and Microbiology Section, Laboratory Services Branch, Ontario Ministry of the Environment.

Wilson, P.W. and D.T. Long. 1993. Geochemistry and isotope chemistry of Michigan Basin brines: Devonian formations. Applied Geochemistry. 8:81-100.

Wisconsin State Laboratory of Hygiene (WISLOH). 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. Cited In: Nagpal *et al*., 2003.

WISLOH. 2007. Tables of Data (provided to C. Stephan, US EPA). Cited In: Iowa Chloride Criteria Development Reference List May 2009.

Wright DA. 1980. Cadmium and calcium interactions in the freshwater amphipod *Gammarus pulex*. Freshwater Biology. 10:123-133.

Wurtz, C.B. and C.H. Bridges. 1961. Preliminary results from macroinvertebrate bioassays. Proc. Pennsylvania Academy of Science, 35:51-56.

Yanbo, W., Wenju, Z., Zirong, X. 2006. Acute toxicity of nitrite on tilapia (*Oreochromis niloticus*) at different external chloride concentrations. Fish Physiology and Biochemistry 32: 49-54.

Yarzhombek, A.A., A.E. Mikulin and A.N. Zhdanova. 1991. Toxicity of Some Substances to Fish in Relation to Form of Exposure. (Toksichnost Vestichestv diya ryb v Zavisimosti ot Sposoba Vozdejstviya). J Ichthyol 31(7):99-106; Vopr Ikhtiol 31(3):496-502 (RUS). Cited In: Bright and Addison, 2002.

Young, R.T. 1923. Resistance of fish and alkalinity. Amer. Jour. Physiol. 63: 373. Cited In: Hanes *et al*., 1970, and Cited In: Evans and Frick, 2001.

APPENDIX I
Chloride short-term and long-term aquatic toxicity data.

Short-Term Aquatic Toxicity Data Table

Compound: Sodium Chloride and Calcium Chloride

Species (Life Stage)	Response	pH	Temperature (°C)	Dissolved Oxygen (mg/L)	Alkalinity	Hardness (mg $CaCO_3$/L)	Effect Concentration (mg Cl/L)	Data Codes	Data Quality	Reference
ACUTE - VERTEBRATE										
Americal eel (*Anguilla japonica*) (young)	Survival (50-h)		20-22				7,091	A	?	Oshima 1931 (In Doudoroff and Katz 1953; in Evans and Frick 2001)
American eel (*Anguilla rostrata*) (black eel stage)	96h LC50	7.2-7.6	22±1	≥40% saturation	30-35	40-48	13,012	A,S	S	Hinton and Eversol 1979
American eel (*Anguilla rostrata*) (glass eel stage)	96h LC50						10,846	A,S	?	Hinton and Eversol 1978 (In Nagpal et al 2003 and In Evans and Frick 2001)
American toad (*Bufo americanus*) (Gosner stage 25)	96h LC50					33 (taken from a paper found at http://www.ajcn.org/cgi/reprint/37/1/37)	3,926	A,S,M	S	Collins and Russell 2009
Bannerfin Shiner (*Cyprinella leedsi*) (12d old, total length 0.9 to 1.3 cm)	96h LC50	7.98	21.5	9.5	85	296	6,070	A, R, M	P	Environ, 2009
Black bullhead (*Ameiurus melas*)	96h LC50		23±1			22	4,849	A,S,M	U (no mention of control survival)	Clemens and Jones 1954
Bluegill sunfish (*Lepomis macrochirus*)	96h LC50	7.58±0.15	21.7±0.1	7.1±0.3	60.3±3.4	101.7±7.6 (ASTM recon water)	3,543	A,F,M	S	Birge et al. 1985
Bluegill sunfish (*Lepomis macrochirus*)	96h LC50		16-20	5-9		39.2 (listed as SRW in EPA 1991 and actual hardness provided in EPA Cl update data)	7,853	A	U (control survival not reported)	Patrick et al. 1968

Bluegill sunfish (*Lepomis macrochirus*) (Length: 5 to 9 cm; Average wt: 1 to 9 g).	96h LC50		16-20				7,846	A	S	Trama 1954
Bluegill sunfish (*Lepomis macrochirus*) (20-35 grams)	24h LC50	7.3±0.4	22±0.2				8,553	A,S	?	Abegg 1949, 1950 (In Doudoroff and Katz 1953; in Evans and Frick 2001)
Bluegill sunfish (*Lepomis macrochirus*)	24h LC50					Standard Reference Water	8,568	A	U (no mention of control survival)	Dowden and Bennett 1965
Bluegill sunfish (*Lepomis macrochirus*) (avg lt=3.5cm, avg wt=0.6g)	0% Mortality (24-h)	8.2±0.5	12±1	≥60% saturation	100±10	130-150	6,066	A,S,U	S	Waller et al. 1996
Bluegill sunfish (*Lepomis macrochirus*) (avg lt=3.5cm, avg wt=0.6g)	0% Mortality (24-h)	8.2±0.5	17±1	≥60% saturation	100±10	130-150	6,066	A,S,U	S	Waller et al. 1996
Bluegill sunfish (*Lepomis macrochirus*) (avg wetwt=1.03±0.5g, avg lt=4.37±0.59cm)	6h LC100	7.37-7.87	18.8-20.1	≥40% saturation	54-59	74-116	8,978	A,F,M	S	Kszos et al. 1990
Bluegill sunfish (*Lepomis macrochirus*) (avg lt=3.5cm, avg wt=0.6g)	6h LC47	8.2±0.5	17±1	≥60% saturation	100±10	130-150	12,132	A,S,U	S	Waller et al. 1996
Brook trout (*Salvelinus fontinalis)*	Survival and Recovery (0.5-1h)						18,198	A	U (exposure via ingestion)	Phillips 1944
Brook trout (*Salvelinus fontinalis)*	0.25h LC50						30,330	A	U (exposure via ingestion)	Phillips 1944

Brown trout (*Salmo trutta*) (avg lt=14.0cm, avg wt=30.0g)	24h LC0	8.2±0.5	12±1	≥60% saturation	100±10	130-150	6,066	A,S,U	S	Waller et al. 1996
Channel catfish (*Ictalurus ounctatus)* (avg lt=4.7cm, avg wt=1.2g)	24h LC0	8.2±0.5	12±1	≥60% saturation	100±10	130-150	6,066	A,S,U	S	Waller et al. 1996
Channel catfish (*Ictalurus ounctatus*) (avg lt=4.7cm, avg wt=1.2g)	24h LC0	8.2±0.5	17±1	≥60% saturation	100±10	130-150	6,066	A,S,U	S	Waller et al. 1996
Channel catfish (*Ictalurus ounctatus*) (avg lt=4.7cm, avg wt=1.2g)	6h LC100	8.2±0.5	17±1	≥60% saturation	100±10	130-150	12,132	A,S,U	S	Waller et al. 1996
Chorus frog (*Pseudacris triseriata feriarum*) (<24h post hatch)	48h LC50	7.4-7.9	24.5-25.7	>4.0		84.8	3,550	A,R,M	S	Garibay and Hall 2004
Chorus frog (*Pseudacris triseriata feriarum*) (<24h post hatch)	96h LC50	7.4-7.9	24.5-25.7	>4.0		84.8	3,506	A,R,M	S	Garibay and Hall 2004
Chorus frog (*Pseudacris triseriata feriarum*) (72h post hatch)	48h LC50	7.4-7.9	24.5-25.7	>4.0		84.8	3,550	A,R,M	S	Garibay and Hall 2004
Chorus frog (*Pseudacris triseriata feriarum*) (72h post hatch)	96h LC50	7.4-7.9	24.5-25.7	>4.0		84.8	2,320	A,R,M	S	Garibay and Hall 2004

Common eel *(Anguilla anguilla)*	24h LC0						12,132	A	?	Buchmann et al. 1992 (In Bright and Addison 2002)
Common frog (*Rana temporaria*) (Gosner stage 8/9, early/mid cleavage, embryonic stage egg capsules)	96h LC47.6 (mortality of Gosner stage 20/21 related to initial number of st. 8/9 embryos)	7.2-7.5	19-22	at saturation		Association of Analytical Chemists exposure water (10 mOsmol, 10 °dH equivalents to 3.57 mval, 650 µs)	3,140	A,S,M	S	Viertel 1999
Common, mirror, colored, carp (*Cyprinus carpio*)	Mortality (LC50, 0.167d)	6.1-6.6			0.43-1.00 mmol/L??		7,461	A	U (invasive species)	Rosicky et al. 1987
Crucian carp (*Carassius carassius*)	24h LC50					Standard Reference Water	8,341	A	U (no mention of control survival)	Dowden and Bennett 1965
Eastern mosquitofish (*Gambusia holbrooki*)	96h LC50	6.07-6.43	20.4±0.8	6.1±0.3	11±0.9	5.7-11.7	7,000	A,F,M	U (pH <6.5)	Newman and Alpin 1992
Fathead minnows (*Pimephales promelas*)	6h LC100	8.2±0.5	17±1	≥60% saturation	100±10	130-150	12,132	A,S,U	S	Waller et al. 1996
Fathead minnow (*Pimephales promelas*)	24h LC0	8.2±0.5	12±1	≥60% saturation	100±10	130-150	6,066	A,S,U	S	Waller et al. 1996
Fathead minnow (*Pimephales promelas*)	24h LC0	8.2±0.5	17±1	≥60% saturation	100±10	130-150	6,066	A,S,U	S	Waller et al. 1996
Fathead minnow (*Pimephales promelas*) (1-7d old)	24h LC50	7.5-9	25	>40% saturation		control/dilution water for tests was MHRW (80-100 mg CaCO3/L)	>4255 (>6660 as CaCl2)	A,S,U	S	Mount et al 1997

Fathead minnow (*Pimephales promelas*) (1-7d old)	48h LC50	7.5-9	25	>40% saturation		control/dilution water for tests was MHRW (80-100 mg CaCO3/L)	>4191 (>6560 as CaCl2)	A,S,U	S	Mount et al 1997
Fathead minnow (*Pimephales promelas*) (1-7d old)	96h LC50	7.5-9	25	>40% saturation		control/dilution water for tests was MHRW (80-100 mg CaCO3/L)	2958 (4630 as CaCl2)	A,S,U	S	Mount et al 1997
Fathead minnow (*Pimephales promelas)* (≤24h old)	96h LC50					39.2 (presented as SRW in EPA ref)	2,790	A,S,U	?	USEPA 1991 (Data from ERL-Duluth)
Fathead minnow (*Pimephales promelas)* (≤24h old)	96h LC50					39.2 (presented as SRW in EPA ref)	2,123	A,S,U	?	USEPA 1991 (Data from ERL-Duluth)
Fathead minnow (*Pimephales promelas)* (≤24h old)	96h LC50					339 (presented as VHRW in EPA ref)	2,244	A,S,U	?	USEPA 1991 (Data from ERL-Duluth)
Fathead minnow (*Pimephales promelas)* (1-7d old)	96h LC50	7.5-9	25	>40% saturation		84.8 (MHRW)	3,876	A,S,U	S	Mount et al 1997
Fathead minnow (*Pimephales promelas)* (11 wks old, mean wt 0.12-0.38 g)	96h LC50		25				4,640	A	S	Adelman et al. 1976
Fathead minnow (*Pimephales promelas)* (larvae)	96h LC50	7.81±0.12	21.7±0.4	7.9±0.3	69.6±5.3	96.3±6.7 (ASTM recon water)	6570	A,F,M	S	Birge et al. 1985
Fathead minnow (*Pimephales promelas*) (juvenile)	96h LC50	7.47-8.03	24-26	6.9-8.7	60	76	4,079	A,M	P	Elphick et al 2011
Fathead minnow (*Pimephales promelas)*	96h LC50					84.8	4,167	A,S,U	?	WISLOH 2007 (In EPA 2008)

Fathead minnow (*Pimephales promelas)*	96h LC50					169.5	4,127	A,S,U	?	WISLOH 2007 (In EPA 2008)
Fathead minnow (*Pimephales promelas*)	96h LC50		22-24				5,288	A,S,M	U (no mention of control survival)	Clemens and Jones 1954
Fathead minnow (*Pimephales promelas*)	96h LC50		22-24				5,431	A,S,M	U (no mention of control survival)	Clemens and Jones 1954
Fathead minnow (*Pimephales promelas*) (juvenile)	96h NOEC	7.47-8.03	24-26	6.9-8.7	60	76	2,173	A,M	P	Rescan Environmental Services Ltd., 2007
Fathead minnow (*Pimephales promelas*) (juvenile)	96h LOEC	7.47-8.03	24-26	6.9-8.7	60	76	4,293	A,M	P	Rescan Environmental Services Ltd., 2007
Frog *(Microhyla ornata)* (late grastula stage, Gosner stage 11/12)	24h LOC50	7.5-7.8	23-27		<60	<75	3,932	A,S,U	U (not representative of a temperate species)	Padhye and Ghate 1992
Frog *(Microhyla ornata)* (late grastula stage, Gosner stage 11/12)	48h LC50	7.5-7.8	23-27		<60	<75	3,399	A,S,U	U (not representative of a temperate species)	Padhye and Ghate 1992
Frog *(Microhyla ornata)* (late grastula stage, Gosner stage 11/12)	72h LC50	7.5-7.8	23-27		<60	<75	2,561	A,S,U	U (not representative of a temperate species)	Padhye and Ghate 1992
Frog *(Microhyla ornata)* (late grastula stage, Gosner stage 11/12)	96h LC50	7.5-7.8	23-27		<60	<75	1,644	A,S,U	U (not representative of a temperate species)	Padhye and Ghate 1992

Frog *(Microhyla ornata)* (8d old tadpoles, Gosner stage 24)	96h LC50	7.5-7.8	23-27		<60	<75	3,049	A,S,U	U (not representative of a temperate species)	Padhye and Ghate 1992
Frog *(Microhyla ornata)* (hind-limb stage tadpoles, Gosner stage 39)	96h LC50	7.5-7.8	23-27		<60	<75	4,203	A,S,U	U (not representative of a temperate species)	Padhye and Ghate 1992
Frog (*Rana breviceps*)	76h NOEC Mortality	5.6	not reported	not reported	8	20	1,820	A	U (pH too low, temp not reported, not resident of Canada)	Mahajan et al. 1979
Frog (*Rana breviceps*)	76h LOEC Mortality	5.6	not reported	not reported	8	24	3,033	A	U (pH too low, temp not reported, not resident of Canada)	Mahajan et al. 1979
Bullfrog (*Rana catesbeiana*) (tadpoles, avg wet wt 1.2g, total length 4.5 to 5.5 cm)	96h LC50	8.02	22.5	8.8	56	300	5,846	A, S, M	P	Environ, 2009
Golden shiners (*Notemigonus crysoleucas*) (9.5-11.0 cm)	Average Survival Time (97-h)	7.8-7.9	22-22.5	7-8			6,066	A	?	Wiebe et al. 1934 (In Evans and Frick, 2001)
Golden shiners (*Notemigonus crysoleucas)* (10.0-11.0 cm)	Average Survival Time (4.73-h)	7.8-7.9	22-22.5	7-8			9,099	A	?	Wiebe et al. 1934 (In Evans and Frick 2001)
Golden shiners (*Notemigonus crysoleucas*) (9.5-11.5 cm)	Average Survival Time (1.33-h)	7.8-7.9	22-22.5	7-8			12,132	A	?	Wiebe et al. 1934 (In Evans and Frick 2001)

Goldfish (*Carassius auratus*)	96h LC50		25				4,453	A	U (treated with potassium permanganate and tetracylcline to kill parasite)	Adelman et al. 1976
Goldfish *(Carassius auratus)*	Mortality or Immobilization (17-h)						7,137	A	?	Ellis 1937 (In McKee and Wolf 1963, In Evans and Frick 2001)
Goldfish (*Carassius auratus*)	Mortality (0.46-0.63h)		21				21,292	A	?	Powers 1917 (In Hammer 1977; Doudoroff and Katz; in Evans and Frick 2001)
Green frog *Rana clamitans* (Gosner stage 25)	96h LC50					33 (taken from a paper found at http://www.ajcn.org/cgi/reprint/37/1/37)	3,109	A,S,M	S	Collins and Russell 2009
Green sunfish (*Lepomis cyanellus*)	96h LC50		22-24			22	6,499	A,S,M	U (no mention of control survival)	Clemens and Jones 1954
Guppy (*Poecilia reticulata*)	24h LC50						12,132	A	U (not a temperate species)	Yarzhombek et al. 1991 (In Bright and Addison 2002)
Guppy (*Poecilia reticulata*) (juveniles, mean wet wt 0.14g, total length 1.3 to 2cm)	96h LC50	8.03	22.5	8.6	60	290	>11,700	A, R, M	U (not a temperate species)	Environ, 2009
Indian carp fry (C*atla catla, Labeo rohoto,Cirrhinius trifascia)*	48h LC50	7.8-8.2	28-32	4.5-5.5	193-322		3,640	A	U (not temperate species, test temp too high)	Gosh and Pal, 1969

Indian carp fry (C*atla catla, Labeo rohoto,Cirrhinius trifascia)*	24h LC50	7.8-8.2	28-32	4.5-5.5	193-322		4,550	A	U (not temperate species, test temp too high)	Gosh and Pal, 1969
Lake trout (*Salvelinus namaycush*)	24h LC0	8.2±0.5	12±1	≥60% saturation	100±10	130-150	6,066	A,S,U	S	Waller et al. 1996
Lake Whitefish (*Coregonus clupeaformis*) (fry)	Immobilization (Lake Erie water)						10,009	A	?	Edmister and Gray 1948 (In Anderson 1948; also listed in EPA reference list)
Leopard frog (*Lithibates pipiens* previously *Rana pipiens*) (tadpoles, Gosner stage 25)	96h LC50						3,385	A,S,M	S	Jackman 2010
Minnows (length of 5-8 cm)	Mortality or Immobilization (6-h)		18	approx 6.42	12.5	distilled water	6,066	A	U (Genus / species unknown)	LeClerc 1960 and LeClerc and Devlaminck 1950 (In McKee and Wolf 1963; in Evans and Frick 2001)
Minnows (length of 5-8 cm)	Mortality or Immobilization (6-h)		19	approx 6.42	150	hard water	6,976 - 7,279	A	U (Genus / species unknown)	LeClerc 1960 and LeClerc and Devlaminck 1950 (In McKee and Wolf 1963; in Evans and Frick 2001)
Mosquito fish (*Gambusia affinis*)	96h LC50						10,646	A	U (fish treated with terramycin during holding)	Wallen et al. 1957
Mosquito fish (*Gambusia affinis*)	96h LC50		22-24			22	6,472	A,S,M	U (no mention of control survival)	Clemens and Jones 1954
Mosquito fish (*Gambusia affinis*)	96h LC50		16.7-20.0				9,099	A,S,U	S	Al-Daham and Bhatti 1977
Pikeperch *(Stizostedion lucioperca)*	Mortality (0.38-d)						3,034	A	?	Stom and Zubareva 1994 (In Bright and Addison 2002)

Pikeperch *(Stizostedion lucioperca)* (11.5 mm length)	Mortality (0.17h)					130	24,268	A	S	Stangenberg 1975
Plains killfish (*Fundulus kansae*)	96h LC50		22-24			22	9,706	A,S,M	U (no mention of control survival)	Clemens and Jones 1954
Rainbow trout (*Oncorhynchus mykiss)*	24h LC0	8.2±0.5	12±1	≥60% saturation	100±10	130-150	6,066	A,S,U	S	Waller et al. 1996
Rainbow trout (*Oncorhynchus mykiss)*	24h LC0	8.2±0.5	17±1	≥60% saturation	100±10	130-150	6,066	A,S,U	S	Waller et al. 1996
Rainbow trout (Oncorhynchus mykiss)	6h LC40	8.2±0.5	17±1	≥60% saturation	100±10	130-150	12,132	A,S,U	S	Waller et al. 1996
Rainbow trout (*Oncorhynchus mykiss*) (juvenile)	96h NOEC (Mortality)	7.01-7.44	13-15	8.7-9.9	36	40	4,265	A,M	P	Rescan Environmental Services Ltd., 2007
Rainbow trout (*Oncorhynchus mykiss*) (juvenile)	96h LOEC (Mortality)	7.01-7.44	13-15	8.7-9.9	36	40	8,400	A,M	P	Rescan Environmental Services Ltd., 2007
Rainbow trout (*Oncorhynchus mykiss*) (juvenile)	96h LC50	7.01-7.44	13-15	8.7-9.9	36	40	6,030	A,M	P	Elphick et al 2011
Rainbow trout (*Oncorhynchus mykiss*) (fingerlings) (mean wt 0.31±0.06g)	96h LC50	8.06-8.46	14-16	9.9-10.1		119	9,886	A,S,M	S	Dow et al. 2010
Rainbow trout (*Oncorhynchus mykiss*) (juvenile, 12.9-14.4g)	96h LC50	8	12-13.5	8-10	244	284	12,363	A,U,R	S	Vosyliene et al. 2006
Rainbow trout (*Oncorhynchus mykiss*) (juvenile)	96h LC50					46	6,743	A,F,M	?	Spehar 1986, 1987 (Acute test results used in Iowa chloride criteria development)
Rainbow trout (*Salmo gairdneri*) (total length 15-20 cm)	24h LC50		14-16				3,336	A,R,	S	Kostecki and Jones 1983
Red shiner (*Notropis lutrensis*)	96h LC50		22-24			22	5,771	A,S,M	U (no mention of control survival)	Clemens and Jones 1954

Red shiner (*Notropis lutrensis*)	96h LC50		22-24			---	5,920	A,S,M	U (no mention of control survival)	Clemens and Jones 1954
Sailfin molly (*Poecilia latipinna*)	48h LC50					Standard Reference Water	10,066	A	U (not resident of Canada, no control survival)	Dowden and Bennett 1965
Silver carp (*Hypophthalmichthys molitrix*)	Mortality (LC50, 0.167-d)						6,855	A	U (not resident of Canada)	Rosicky et al. 1987 (In Bright and Addison 2002)
Small freshwater cyprinodont *(Orizias latipes)*	Mortality (24-h)						8,864-17,727	A	?	Iwao 1936 (In Doudoroff and Katz 1953; in Evans and Frick 2001)
Smallmouth bass (*Micropterus dolomieu)*	3.3% Mortality (24-h)	8.2±0.5	12±1	≥60% saturation	100±10	130-150	6,066	A,S,U	S	Waller et al. 1996
Spotted salamandar (*Ambystoma maculatum)* (1.74 ± 0.08g)	96h LC50					33 (taken from a paper found at http://www.ajcn.org/cgi/reprint/37/1/37)	1,178	A,S,M	S	Collins and Russell 2009
Spring Peeper *Pseudacris crucifer* (Gosner stage 25)	96h LC50					33 (taken from a paper found at http://www.ajcn.org/cgi/reprint/37/1/37)	2,830	A,S,M	S	Collins and Russell 2009
Striped bass *(Morone saxatilis)*	96h LC50						607	A	U (fish not acclimated properly)	Hughes 1973
Striped bass *(Morone saxatilis)*	96h LC50						3,033	A	U (fish not acclimated properly)	Hughes 1973

Threespine stickelback (*Gasterosteus aculeatus*)	96h EC50	maintained between 6 and 9	19-21	>4.0		84.8	10,200	A,R,M	S	Garibay and Hall 2004
Walleye (*Stizostedion vitreum*)	24h LC0	8.2±0.5	12±1	≥60% saturation	100±10	130-150	6,066	A,S,U	S	Waller et al. 1996
Walleye (*Stizostedion vitreum*)	24h LC0	8.2±0.5	17±1	≥60% saturation	100±10	130-150	6,066	A,S,U	S	Waller et al. 1996
Walleye (*Stizostedion vitreum*) (fry)	Immobilization (Lake Erie water)						2,341	A	?	Edmister and Gray 1948 (In Anderson 1948; also listed in EPA reference list)
Wood frog *Lithibates sylvatica* (previously *Rana sylvatica*) (tadpoles, Gosner stage 25)	96h LC50		18.7-19.3				1599 (S-K)	A, R, U	U (innacurate effect concentration calculated using Spearman-Karber)	Sanzo and Hecnar, 2006
Wood frog *Lithibates sylvatica* (previously *Rana sylvatica*) (tadpoles, Gosner stage 25)	96h LC50		18.7-19.3				3099 (probit)	A, R, U	S	Sanzo and Hecnar, 2006
Wood frog *Rana sylvatica* (Gosner stage 25 - first active feeding stage)	96h LC50					33 (taken from a paper found at http://www.ajcn.org/cgi/reprint/37/1/37)	1,721	A,S,M	S	Collins and Russell 2009
Wood frog *Lithibates sylvatica* (previously *Rana sylvatica*) (tadpoles, Gosner stage 25)	96h LC50						3,755	A,S,M	S	Jackman 2010

Yellow perch (*Perca flavescens*)	24h LC0	8.2±0.5	12±1	≥60% saturation	100±10	130-150	6,066	A,S,U	S	Waller et al. 1996
Yellow perch (*Perca flavescens*)	24h LC0	8.2±0.5	17±1	≥60% saturation	100±10	130-150	6,066	A,S,U	S	Waller et al. 1996
Zebrafish (*Brachydanio rerio*) (embryo)	Terat. (EC50, 48-h)						7,290	A	U (tropical freshwater fish)	Lange et al. 1995
ACUTE - INVERTEBRATE										
Amphipod *(Gammarus pseudolimnaeus)*	20% Mortality (24-h)		11				2,500	A,S,M	S	Crowther and Hynes 1977
Amphipod *(Gammarus pseudolimnaeus)*	96h LC0		7			dilution water was spring water collected from Greater Toronto Area (<10 mg/L Cl)	3,000	A,S,U	S	Williams et al. 1999
Amphipod *(Crangonyx sp.)*	96h LC0		7			dilution water was spring water collected from Greater Toronto Area (<10 mg/L Cl)	3,000	A,S,U	S	Williams et al. 1999
Amphipod (*Hyalella azteca*) (7-8 d)	Mortality (96 hr, NOEC)	7.7-7.9	22-24	7.5-8.4	60	76 (MHSW)	1,123	A,S,M	P	Rescan Environmental Services Ltd., 2007
Amphipod (*Hyalella azteca*) (7-8 d)	Mortality (96 hr, LOEC)	7.7-7.9	22-24	7.5-8.4	60	76 (MHSW)	2,190	A,S,M	P	Rescan Environmental Services Ltd., 2007
Amphipod (*Hyalella azteca*) (7-8 d)	Mortality (96 hr, IC25)	7.7-7.9	22-24	7.5-8.4	60	76 (MHSW)	1,186	A,S,M	P	Rescan Environmental Services Ltd., 2007
Amphipod (*Hyalella azteca*) (7-14 d old)	96h LC50	8.3-9.3	23		70	102.5 (mod hard recon water)	3,947	A,S,U	S	Lasier et al 1997
Amphipod (*Hyalella azteca*) (7-8 d)	Mortality (96 hr, LC50)	7.7-7.9	22-24	7.5-8.4	60	76 (MHSW)	1,382 (trimmed S-K)	A,S,M	P	Elphick et al 2011
Amphipod (*Hyalella azteca*) (7-8 d)	Mortality (96 hr, LC50)	7.7-7.9	22-24	7.5-8.4	60	76 (MHSW)	1,521 (linear interpolation)	A,S,M	P	Rescan Environmental Services Ltd., 2007

Amphipod (*Hyalella azteca*) (7 d)	Mortality (48 hr, LC50)					ASTM hard	3,700	A,S,U	S	Wang and Ingersoll 2010
Amphipod (*Hyalella azteca*) (7 d)	Mortality (48 hr, LC50)					ASTM hard	3,215	A,S,U	S	Wang and Ingersoll 2010
Amphipod (*Hyalella azteca*) (7 d)	Mortality (48 hr, LC50)					ASTM hard	3,094	A,S,U	S	Wang and Ingersoll 2010
Amphipod (*Hyalella azteca*) (7 d)	Mortality (48 hr, LC50)					ASTM hard	3,094	A,S,U	S	Wang and Ingersoll 2010
Amphipod (*Hyalella azteca*) (7 d)	Mortality (48 hr, LC50)					ASTM hard	3,458	A,S,U	S	Wang and Ingersoll 2010
Damselfly *(Agria* sp.)	96h LC50	7.85			60	100	14,558	A,S	U (no mention of control survival)	Wurtz and Bridges 1961
Damselfly *(Agria* sp.*)*	96h LC50	7.3			20	20	13,952	A,S	U (no mention of control survival)	Wurtz and Bridges 1961
Caddisfly *(Anaobolia nervosa) (larvae)*	72h LC75		14-17				6,027	A	U (exposures used diluted sea water)	Sutcliffe 1961b (In Evans and Frick 2001)
Caddisfly *(Anaobolia nervosa)* (caddisfly larvae)	72h LC50		14-17				4,255	A	U (exposures usedh diluted sea water)	Sutcliffe 1961b (In Evans and Frick 2001)
Caddisfly *(Chimarra marginata)*	Mortality (0%, 4-d)						155-190	A	?	Camargo and Tarazona 1990
Caddisfly *(Chimarra)*	Mortality (0%, 0.5-d)						315	A	?	Goetsch and Palmer 1997
Caddisfly *(Chimarra sp)*	Mortality (4-d)						3,428	A	?	Goetsch and Palmer 1997

Caddisfly *(Hydropsyche bulbifera)*	Mortality (0%, 4-d)						155-190	A	?	Camargo and Tarazona 1990
Caddisfly *(Hydropsyche exocellata)*	Mortality (0%, 4-d)						155-190	A	?	Camargo and Tarazona 1990
Caddisfly *(Hydropsyche lobata)*	Mortality (0%, 4-d)						155-190	A	?	Camargo and Tarazona 1990
Caddisfly *(Hydropsyche pellucidulla)*	Mortality (0%, 4-d)						155-190	A	?	Camargo and Tarazona 1990
Caddisfly (*Hydropsyche*)	48h LC50						5,459	A	U (field data relating chloride and caddisflies)	Roback 1965
Caddisfly *(Hydroptila angusta)* (3rd and 4th larval instar)	48h LC100	7.9-8.7	12			118-130	6,148	A	U (field collected specimens tested within 24h of collection)	Hamilton et al., 1975
Caddisfly (*Hydroptila angusta*) (3rd and 4th larval instar)	48h LC50	7.9-8.7	12			118-130	4,016	A	U (field collected specimens tested within 24h of collection)	Hamilton et al., 1975
Caddisfly (*Lepidostoma* sp.)	96h LC0		7			dilution water was spring water collected from Greater Toronto Area (<10 mg/L Cl)	3,000	A,S,U	S	Williams et al. 1999
Caddisfly (*Lepidostoma* sp.)	96h LC50		7			dilution water was spring water collected from Greater Toronto Area (<10 mg/L Cl)	6,000	A,S,U	S	Williams et al. 1999

Caddisfly (*Parapsyche* sp.)	96h LC0		7			dilution water was spring water collected from Greater Toronto Area (<10 mg/L Cl)	3,000	A,S,U	S	Williams et al. 1999
Chironomid *(Chironomus attenatus)* (4th instar)	12h LC50		25				6,062	A,S	S	Thornton and Sauer, 1972
Chironomid *(Chironomus attenatus)* (4th instar)	24h LC50		25				5,956	A,S	S	Thornton and Sauer, 1972
Chironomid *(Chironomus attenatus)* (4th instar)	36h LC50		25				5,814	A,S	S	Thornton and Sauer, 1972
Chironomid *(Chironomus attenatus)* (4th instar)	48h LC50		25				4,850	A,S	S	Thornton and Sauer, 1972
Chironomid (*Chironomus attenatus*) (4th instar)	12,24,36,48-h LC100		25				7,275	A,S	S	Thornton and Sauer, 1972
Chironomid (*Chironomus dilutus / tentans*) (third instar larvae)	96h NOEC	7.2-7.8	22-24	5.8-8.0	60	76 (MHSW)	2,150	A,M	P	Rescan Environmental Services Ltd., 2007
Chironomid (*Chironomus dilutus / tentanss*) (third instar larvae)	96h LOEC	7.2-7.8	22-24	5.8-8.0	60	76 (MHSW)	4,805	A,M	P	Rescan Environmental Services Ltd., 2007
Chironomid (*Chironomus dilutus / tentans*) (third instar larvae - approx. 10d old)	96h LC50	7.2-7.8	22-24	5.8-8.0	60	76 (MHSW)	5,867	A,M	P	Elphick et al 2011
Chironomid (*Chironomus dilutus / tentans*) (7d old)	96h LC50					ASTM hard	3,761	A,S,U	S	Wang and Ingersoll 2010
Chironomid (*Chironomus dilutus / tentans)* 2nd to 3rd instar (9d old at test initiation)	48h LC50	7.98	21.5	9.5	85	296	6,032	A, S, M	P	Environ, 2009

Chironomid *(Cricotopus trifascia)*	48h LC100	7.9-8.7	12			118-130	5,378	A	U (field collected specimens tested within 24h of collection)	Hamilton et al., 1975
Chironomid *(Cricotopus trifascia)*	48h LC50	7.9-8.7	12			118-130	3,774	A	U (field collected specimens tested within 24h of collection)	Hamilton et al., 1975
Chironomid *(Chironomus riparius)* (4d old)	48h LC50					ASTM hard	6,912	A	S	Wang and Ingersoll 2010
Copepod *(Epischura baikalensis)* (copepodite stages IV-V)	Mortality (0%, 24-h)		5				4	A	U (control survival not reported)	Stom and Zubareva 1994
Copepod (*Diaptomus* sp.)	96h LC50		22-24			22	2,571	A,S,M	U (no mention of control survival)	Clemens and Jones 1954
Copepod *(Cyclops abyssorum prealpinus)* (adult avg length of 0.62 mm)	48h LC50	7.2	9.5-10.5	air saturated	10.4	33	12,385 (7000 mg Ca/L (as CaCl2*2H2O))	A	S	Baudouin and Scoppa 1974
Copepod *(Eudiaptomus padanus padanus)* (adult avg length of 0.3 mm)	48h LC50	7.2	9.5-10.5	air saturated	10.4	33	7,077 (4000 mg Ca/L (as CaCl2*2H2O))	A	S	Baudouin and Scoppa 1974
Crayfish (*Cambarus* sp.)	96h LC50		22-24			22	10,557	A,S,M	U (no mention of control survival)	Clemens and Jones 1954
Dragonfly (*Libellulidae* sp.)	96h LC50		22-24			22	9,671	A,S,M	U (no mention of control survival)	Clemens and Jones 1954

Fairy shrimp *(Streptocephalus proboscideus)*	24h LC50						4,184	A	U (native to Africa)	Calleja et al. 1994
Fairy shrimp *(Streptocephalus rubricaudatus)*	24h LC50						1,862	A	U (native to Africa)	Crisinel et al. 1994
Fingernail clam (*Sphaerium simile*) juveniles, 4.5-6.5 mm	96h LC50	7.8	21-23	7.91	64	51	740	A,S	P	GLEC and INHS 2008
Fingernail clam (*Sphaerium simile*) juveniles, 4.5-6.5 mm	96h LC50	7.9	21-23	7.21	61	192	1,100	A,S	P	GLEC and INHS 2008
Fingernail clam (*Sphaerium tenue*)	96h LC50					100	667	A,S	U (no mention of control survival)	Wurtz and Bridges 1961
Fingernail clam (*Sphaerium tenue*)	96h LC50					20	698	A,S	U (no mention of control survival)	Wurtz and Bridges 1961
Fingernail clam (*Musculium transversum*), juveniles	96h LC50	7.9 - 8.1	22 ± 1	7.93 - 8.14	62	48 (EPA moderately hard recon water)	1930	A,S,M	S	US EPA 2010
Flatworm *(Polycelis nigra)*	Survival (48-h)		15-18				6,739	A	U (control survival not reported)	Jones 1940; 1941
Cumberlandian combshell (*Epioblasma brevidens*) (endangered in USA)	24h EC50 (survival of glochidia)						1,626	A,S	S	Valenti et al. 2007
Oyster mussel (*Epioblasma capsaeformis*) (endangered in USA) (2 months old)	96h EC50					ASTM hard	2,426	A,S,U	S	Wang and Ingersoll 2010

Oyster mussel (*Epioblasma capsaeformis*) (endangered in USA)	24h EC50 (survival of glochidia)						1,644	A,S	S	Valenti et al. 2007
Freshwater mussel (*Villosa delumbis*)	24h EC50 (survival of glochidia)	8.32-8.61	20.1-21.9	>80% saturation	116-130	170-192	2,008	A,S,M	S	Bringolf et al 2007
Freshwater mussel (*Villosa delumbis*)	48h EC50 (survival of glochidia)	8.32-8.61	20.1-21.9	>80% saturation	116-130	170-192	2,202	A,S,M	S	Bringolf et al 2007
Freshwater mussel (*Villosa delumbis*)	96h EC50 (survival of juveniles)	8.32-8.61	20.1-21.9	>80% saturation	116-130	170-192	3,173	A,S,M	S	Bringolf et al 2007
Freshwater mussel (*Villosa constricta*) (10d old)	24h EC50					ASTM hard (160-180)	2,366	A,S,M	S	Wang and Ingersoll 2010
Freshwater mussel (*Villosa constricta*)	24h EC50 (survival of glochidia)	8.32-8.61	20.1-21.9	>80% saturation	116-130	170-192	1,674	A,S,M	S	Bringolf et al 2007
Freshwater mussel (*Villosa constricta*)	48h EC50 (survival of glochidia)	8.32-8.61	20.1-21.9	>80% saturation	116-130	170-192	1,571	A,S,M	S	Bringolf et al 2007
Freshwater mussel (*Elliptio complananta*)	24h EC50 (survival of glochidia)	8.32-8.61	20.1-21.9	>80% saturation	116-130	170-192	1,620	A,S,M	S	Bringolf et al 2007
Freshwater mussel (*Elliptio complananta*)	48h EC50 (survival of glochidia)	8.32-8.61	20.1-21.9	>80% saturation	116-130	170-192	1,353	A,S,M	S	Bringolf et al 2007
Yellow lance FW mussel *(Elliptio lanceolata)* (10d old)	96h LC50					ASTM hard	1,274	A,S	S	Wang and Ingersoll 2010
Wavy-rayed lampmussel (*Lampsilis fasciola*) (change in status endangered to special concern, public comment period ending 7Jan11)	24h EC50 (survival of glochidia)	7.81±0.13	19-21	>5.0	62.6±3.9	82.9±5.8	1,868	A,S,U	S	Valenti et al. 2007

Wavy-rayed lampmussel (*Lampsilis fasciola*) (change in status endangered to special concern, public comment period ending 7Jan11)	24h EC50 (survival of glochidia)	8.32-8.61	20.1-21.9	>80% saturation	116-130	170-192	1,116	A,S,M	S	Bringolf et al 2007
Wavy-rayed lampmussel (*Lampsilis fasciola*) (change in status endangered to special concern, public comment period ending 7Jan11)	48h EC50 (survival of glochidia)	8.32-8.61	20.1-21.9	>80% saturation	116-130	170-192	1,055	A,S,M	S	Bringolf et al 2007
Wavy-rayed lampmussel (*Lampsilis fasciola*) (change in status endangered to special concern, public comment period ending 7Jan11)	96h EC50 (survival of juveniles)	8.32-8.61	20.1-21.9	>80% saturation	116-130	170-192	2,414	A,S,M	S	Bringolf et al 2007
Wavy-rayed lampmussel (*Lampsilis fasciola*) (change in status endangered to special concern, public comment period ending 7Jan11)	24h EC50 (2008) (survival of glochidia)		21			95-115 (ASTM moderately hard water)	**113**	A,S,M	S	Gillis 2011
Wavy-rayed lampmussel (*Lampsilis fasciola*) (change in status endangered to special concern, public comment period ending 7Jan11)	24h EC50 (2009) (survival of glochidia)		21			95-115 (ASTM moderately hard water)	**285**	A,S,M	S	Gillis 2011
Wavy-rayed lampmussel (*Lampsilis fasciola*) (COSEWIC special concern)	24h EC50 (2009) (survival of glochidia)		21			292 (Sydenham River, Ontario)	**1559**	A,S,M	U (field collected water)	Gillis 2011

Wavy-rayed lampmussel (*Lampsilis fasciola*) (COSEWIC special concern)	24h EC50 (2009) (survival of glochidia)		21			278 (Grand River, Ontario)	**1313**	A,S,M	U (field collected water)	Gillis 2011
Wavy-rayed lampmussel (*Lampsilis fasciola*) (COSEWIC special concern)	24h EC50 (2009) (survival of glochidia)		21			322 (Maitland River, Ontario)	**1391**	A,S,M	U (field collected water)	Gillis 2011
Wavy-rayed lampmussel (*Lampsilis fasciola*) (COSEWIC special concern)	24h EC50 (2009) (survival of glochidia)		21			306 (Thames River, Ontario)	**1265**	A,S,M	U (field collected water)	Gillis 2011
Freshwater mussel (*Lampsilis siliquoidea*)	24h EC50 (survival of glochidia)	8.32-8.61	20.1-21.9	>80% saturation	116-130	170-192	334	A,S,M	S	Bringolf et al 2007
Freshwater mussel (*Lampsilis siliquoidea*)	48h EC50 (survival of glochidia)	8.32-8.61	20.1-21.9	>80% saturation	116-130	170-192	340	A,S,M	S	Bringolf et al 2007
Freshwater mussel (*Lampsilis siliquoidea*)	96h EC50 (survival of juveniles)	8.32-8.61	20.1-21.9	>80% saturation	116-130	170-192	2,766	A,S,M	S	Bringolf et al 2007
Freshwater mussel (*Lampsilis siliquoidea*) (collected from Cox Creek in 2008)	24h EC50 (2008) (survival of glochidia)		21			95-115 (ASTM moderately hard water)	**168**	A,S,M	U (% viability from 0h to 24h changed by >10%)	Gillis 2011
Freshwater mussel (*Lampsilis siliquoidea*) (collected from Maitland River 2009)	24h EC50 (2009) (survival of glochidia)		21			95-115 (ASTM moderately hard water)	**1430**	A,S,M	S	Gillis 2011
Freshwater mussel (*Lampsilis siliquoidea*) (collected from Maitland River 2009)	24h EC50 (2009) (survival of glochidia)		21			40-48 (ASTM soft water)	**763**	A,S,M	S	Gillis 2011
Freshwater mussel (*Lampsilis siliquoidea*) (collected from Maitland River 2009)	24h EC50 (2009) (survival of glochidia)		21			160-180 (ASTM hard water)	**1962**	A,S,M	S	Gillis 2011

Freshwater mussel (*Lampsilis siliquoidea*) (collected from Maitland River 2009)	24h EC50 (2009) (survival of glochidia)		21			280-320 (ASTM very hard water)	**1870**	A,S,M	S	Gillis 2011
Freshwater mussel (*Lampsilis siliquoidea*) (2 weeks old)	96h EC50					160-180 (ASTM hard)	1517	A,S,U	S	Wang and Ingersoll 2010
Freshwater mussel (*Lampsilis siliquoidea*) (2 months old)	96h EC50					160-180 (ASTM hard)	2426	A,S,U	S	Wang and Ingersoll 2010
Freshwater mussel (*Lampsilis siliquoidea*) (2 months old)	96h EC50					160-180 (ASTM hard)	2669	A,S,U	S	Wang and Ingersoll 2010
Freshwater mussel (*Lampsilis siliquoidea*) (4 months old)	96h EC50					160-180 (ASTM hard)	2244	A,S,U	S	Wang and Ingersoll 2010
Mussel *Lampsilis siliquoidea* (≤ 2month old juvenile)	?					169.5	1,905	A,R,M	?	Wang 2007 (In EPA Iowa update, this was an email to S.Charles)
Northern Riffleshell Mussel (*Epioblasma torulosa rangiana)* (glochidia) (COSEWIC endangered, Canadian occurrence in Ontario)	24h EC50 (survival of glochidia)		21			95-115 (ASTM moderately hard water)	244	A,S,M	S	Gillis 2011
Plain Pocketbook (*Lampsilis cardium*) (glochidia)	24h EC50 (survival of glochidia)		21			95-115 (ASTM moderately hard water)	**817**	A,S,M	S	Gillis 2011
Rainbow mussel (*Villosa iris*) (2months old) (COSEWIC endangered, Canadian occurrence in Ontario)	96h EC50					160-180 (ASTM hard)	1517	A,S,U	S	Wang and Ingersoll 2010
Rainbow mussel (*Villosa iris*) (2months old) (COSEWIC endangered, Canadian occurrence in Ontario)	96h EC50					160-180 (ASTM hard)	1638	A,S,U	S	Wang and Ingersoll 2010

Rainbow mussel (*Villosa iris*) (2months old) (COSEWIC endangered, Canadian occurrence in Ontario)	96h EC50					160-180 (ASTM hard)	2244	A,S,U	S	Wang and Ingersoll 2010
Rainbow mussel (*Villosa iris*) (2months old) (COSEWIC endangered, Canadian occurrence in Ontario)	96h EC50					160-180 (ASTM hard)	1820	A,S,U	S	Wang and Ingersoll 2010
Rainbow mussel (*Villosa iris*) (2months old) (COSEWIC endangered, Canadian occurrence in Ontario)	96h EC50					160-180 (ASTM hard)	1941	A,S,U	S	Wang and Ingersoll 2010
Rainbow mussel (*Villosa iris*) (juvenile) (COSEWIC endangered, Canadian occurrence in Ontario)	?					169.5	2,069	A,R,M	?	Wang 2007 (In EPA Iowa update, this was an email to S.Charles)
Kidneyshell (*Ptychobranchus fasciolaris*) (glochidia) (COSEWIC endangered, Canadian occurrence in Ontario) - conglutinate spawner	24h EC50		21			278	3,416	A,S,M	U (exposure conducted in natural water, Grand River)	Gillis 2011
Isopod *(Lirceus fontinalis)*	96h LC50	7.73±0.22	21.7±0.2	8.5±0.2	58.6±4.2	100.8±8.2 (ASTM recon water)	2,950	A,F,M	S	Birge et al. 1985
Isopod (*Asellus communis*)	96h LC50					100	5,004	A,S	U (no mention of control survival)	Wurtz and Bridges 1961
Isopod (*Asellus communis*)	96h LC50			3730.59		20	3,094	A,S	U (no mention of control survival)	Wurtz and Bridges 1961
Leech (*Nephelopsis obscura*) (wet wt 0.35g, avg length 7 cm)	96h LC50	8.03	22.5	8.6	60	290	4,310	A, R,M	P	Environ, 2009

Leech (*Erpobdella punctata*)	96h TLm					100	4,550	A,S	U (no mention of control survival)	Wurtz and Bridges 1942
Mayfly *(Hexagenia* spp.*)* (2 months old)	48h LC50					ASTM hard	4,671	A	S	Wang and Ingersoll 2010
Mayfly *(Baetis tricaudatus)* (4-6 mm in length, excluding cerci)	48h EC50 (Immobility) (Current velocity 0 cm/s)	8.3	13	7.9-8.8	150	178	2,875	A,F,M	S	Lowell et al. 1995
Mayfly *(Baetis tricaudatus)* (4-6 mm in length, excluding cerci)	48h EC50 (Immobility) (Current velocity 6 cm/s)	8.3	13	8.6-9.9	150	178	3,233	A,F,M	S	Lowell et al. 1995
Mayfly *(Baetis tricaudatus)* (4-6 mm in length, excluding cerci)	48h EC50 (Immobility) (Current velocity 12 cm/s)	8.3	13	8.6-9.9	150	178	3,300	A,F,M	S	Lowell et al. 1995
Mayfly *(Baetis tricaudatus)*	Development (LOEC, 24-h)						4,853	A,F,M	S	Lowell et al. 1995
Mayfly *(Baetis tricaudatus)*	Immobilization (LOEC, 24-h)						4,853	A,F,M	S	Lowell et al. 1995
Mayfly *(Baetis tricaudatus)*	Development (LOEC, 48-h)						4,853	A,F,M	S	Lowell et al. 1995
Mayfly (*Baetis tricaudatus)*	Immobility (LOEC, 48-h)						3,640	A,F,M	S	Lowell et al. 1995
Mayfly *(Stenonema rubrum*)	48h LC50						1,517	A	U (field data)	Roback 1965
Mayfly *(Tricorythus)*	Mortality (0%, 1,5-d)						315	A	?	Goetsch and Palmer 1997
Mosquito *(Culex* sp. larvae)	48h LC50					Reference Dilution Water	6,187	A	U (no mention of control survival)	Dowden and Bennett 1965

Oligochaete *(Nais variabilis)*	48h LC100	7.9-8.7	12			118-130	2,266	A	U (field collected specimens tested within 24h of collection)	Hamilton et al., 1975
Nematode *(Caenorhabditis elegans)*	24h NOEC (mortality)		20			MHRW as per US EPA 1993	12,435	A	U (soil nematode)	Khanna et al. 1997 (In Evans and Frick 2001)
Nematode *(Caenorhabditis elegans)*	24h NOEC (mortality)		20			K-medium: 2.36g KCl + 3.0 g NaCl per L distilled water)	9,378	A	U (soil nematode)	Khanna et al. 1997 (In Evans and Frick 2001)
Nematode *(Caenorhabditis elegans)*	48h LC50						12,574	A	U (soil nematode)	Cressman and Williams 1994
Nematode *(Caenorhabditis elegans)*	96h NOEC (Mortality)		20			MHRW as per US EPA 1993	12,708	A	U (soil nematode)	Khanna et al. 1997 (In Evans and Frick 2001)
Nematode *(Caenorhabditis elegans)*	96h NOEC (Mortality)		20			K-medium: 2.36g KCl + 3.0 g NaCl per L distilled water)	9,402	A	U (soil nematode)	Khanna et al. 1997 (In Evans and Frick 2001)
Oligochaete or Aquatic Worm (*Lumbriculus variegatus*) (adult)	96h NOEC (Mortality)	7.4-8.2	22-24	5.4-8.5	60	76 (MHSW)	2,145	A,M	P	Rescan Environmental Services Ltd., 2007
Oligochaete or Aquatic Worm (*Lumbriculus variegatus*) (adult)	96h LOEC (Mortality)	7.4-8.2	22-24	5.4-8.5	60	76 (MHSW)	4,480	A,M	P	Rescan Environmental Services Ltd., 2007
Oligochaete or Aquatic Worm (*Lumbriculus variegatus*) (adult)	96h LC50	7.4-8.2	22-24	5.4-8.5	60	76 (MHSW)	3,100	A,M	P	Elphick et al 2011

Oligochaete or Aquatic Worm (*Lumbriculus variegatus*) (adult)	96h LC50	7.98	21.5	9.5	85	296	5,408	A, R, M	P	Environ, 2009
Oligochaete or Aquatic Worm (*Lumbriculus variegatus*)	96h LC50					ASTM hard	>4853	A,S,U	S	Wang and Ingersoll 2010
Oligochaete or Aquatic Worm (*Tubifex tubifex*) (adult)	96h NOEC (Mortality)	7.3-8.1	22-24	5.5-8.5	60	76 (MHSW)	4,575	A,M	P	Rescan Environmental Services Ltd., 2007
Oligochaete or Aquatic Worm (*Tubifex tubifex*) (adult)	96h LOEC (Mortality)	7.3-8.1	22-24	5.5-8.5	60	76 (MHSW)	8,260	A,M	P	Rescan Environmental Services Ltd., 2007
Oligochaete or Aquatic Worm *(Tubifex tubifex*)	96h EC50 (Immobility)	7.5-7.7	29.5-31	5.2 -6.0	390-410	230-250	1,204	A,R	U (test temp too high)	Khangarot, 1991
Oligochaete or Aquatic Worm *(Tubifex tubifex*)	48h EC50 (Immobility)	7.5-7.7	29.5-31	5.2 -6.0	390-410	230-250	1,567	A,R	U (test temp too high)	Khangarot, 1991
Oligochaete or Aquatic Worm *(Tubifex tubifex*)	24h EC50 (Immobility)	7.5-7.7	29.5-31	5.2 -6.0	390-410	230-250	1,928	A,R	U (test temp too high)	Khangarot, 1991
Oligochaete or Aquatic Worm (*Tubifex tubifex*) (adult)	96h LC50	7.3-8.1	22-24	5.5-8.5	60	76 (MHSW)	5,648	A,M	P	Elphick et al 2011
Oligochaete or Aquatic Worm (*Tubifex tubifex*) (adult)	96h LC50					ASTM hard	7,886	A	S	Wang and Ingersoll 2010
Oligochaete or Aquatic Worm *(Tubifex tubifex*) mixed ages	96h TLm					100	3,761	A,S	U (no mention of control survival)	Wurtz and Bridges 1961
Oligochaete or Aquatic Worm *(Tubifex tubifex*) mixed ages	96h LC50	7.6	22±1	7.7	60	52	4,278	A,S	P	GLEC and INHS 2008

Oligochaete or Aquatic Worm *(Tubifex tubifex*) mixed ages	96h LC50	7.7	22±1	7.83	56	220	6,008	A,S	P	GLEC and INHS 2008
Pond snail *(Lymnaea* sp. eggs)	48h LC50					University Lake filtered	2,055	A	U (no mention of control survival)	Dowden and Bennett 1965
Pond snail, pneumonate snail *(Physa heterostropha)*	96h LC50						2,123	A,S	U (no mention of control survival)	Wurtz and Bridges 1961
Pond snail, pneumonate snail *(Physa heterostropha)*	96h LC50						2,487	A,S	U (no mention of control survival)	Wurtz and Bridges 1961
Pond snail, pneumonate snail *(Physa heterostropha)*	96h LC50						3,094	A,S	U (no mention of control survival)	Wurtz and Bridges 1961
Pond snail, pneumonate snail *(Physa heterostropha)*	96h LC50						3,761	A,S	U (no mention of control survival)	Wurtz and Bridges 1961
Snail (*Physa* sp.)	96h TLm		22-24			22	3247	A,S,M	U (no mention of control survival)	Clemens and Jones 1954
Snail (*Physa* sp.)	96h LC0					dilution water was spring water collected from Greater Toronto Area (<10 mg/L Cl)	3,000	A,S,U	S	Williams et al. 1999
Snail (*Physa* sp.)	246h EC60 (stressed behaviour, no feeding or movement)					dilution water was spring water collected from Greater Toronto Area (<10 mg/L Cl)	4,500	A,S,U	S	Williams et al. 1999
Snail (*Gyraulus circumstriatus*)	96h LC50					100	1,941	A,S	U (no mention of control survival)	Wurtz and Bridges 1961

Snail (*Helisoma campanulata*)	96h LC50					100	3,731	A,S	U (no mention of control survival)	Wurtz and Bridges 1961
Stonefly (*Nemoura trispinosa*)	96h LC0					dilution water was spring water collected from Greater Toronto Area (<10 mg/L Cl)	3,000	A,S,U	S	Williams et al. 1999
Rotifer (*Brachionus calyciflorus*) (<4 hr old)	Mortality (24 hr, NOEC)	7.88-8.12	25.0-25.2	7.9-8.4	60	76	1,120	A,M	P	Rescan Environmental Services Ltd., 2007
Rotifer (*Brachionus calyciflorus*) (<4 hr old)	Mortality (24 hr, LOEC)	7.88-8.12	25.0-25.2	7.9-8.4	60	76	2,330	A,M	P	Rescan Environmental Services Ltd., 2007
Rotifer (*Brachionus calyciflorus*) (<4 hr old)	24h LC50	7.88-8.12	25.0-25.2	7.9-8.4	60	76	1,645	A,M	P	Elphick et al 2011
Rotifer (*Brachionus calyciflorus)* (neonate)	24h LC50						2,275	A, S, U	S	Peredo-Alvarez et al., 2003
Rotifer *(Brachionus calyciflorus*)	24h LC50						2,223	A	S	Calleja et al. 1994
Rotifer (*Brachionus patulus)* (neonate)	24h LC50						1,298	A, S, U	S	Peredo-Alvarez et al., 2003
Snail *(Physa gyrina)*	96h LC50	7.41±0.18	21.8±0.1	8.3±0.2	58.0±5.9	100.1±8.3 (ASTM recon water)	2,540	A,F,M	S	Birge et al. 1985
Snail (*Gyraulus parvus)* mixed ages, 3-5 mm	96h LC50	7.7	21-23	7.9	56	56	3,078	A	P	GLEC and INHS 2008
Snail (*Gyraulus parvus)* mixed ages, 3-5 mm	96h LC50	7.7	21-23	7.67	56	212	3,009	A	P	GLEC and INHS 2008

Water flea (*Ceriodaphnia dubia*) (<24h old)	24h LC50	7.5-9	25	>40% saturation			1444 (2260 as CaCl2)	A	S	Mount et al 1997
Water flea (*Ceriodaphnia dubia*) (<24h old)	48h LC50	7.81±0.13	19-21	>5.0	62.6±3.9	82.9±5.8	1,413	A,S,U	S	Valenti et al 2007
Water flea (*Ceriodaphnia dubia*) (neonates, < 24 hr old)	48h EC50 (lethality by immobilization)	8.11-8.66	23-27	7.46-9.14	56-76	54-72	964	A, S, M	S	Harmon et al., 2003
Water flea (*Ceriodaphnia dubia*) (<24h old)	48h LC50	7.5-9	25	>40% saturation		84.8	1,189	A,S,U	U (fed during 48h exposure)	Mount et al 1997
Water flea (*Ceriodaphnia dubia*) (<24h old)	48h LC50	7.5-9	25	>40% saturation		84.8	1,042	A,S,U	U (fed during 48h exposure)	Mount et al 1997
Water flea (*Ceriodaphnia dubia*) (<24h old)	48h LC50		25			39.2	507	A,S,U	S	Hoke et al 1992
Water flea (*Ceriodaphnia dubia*) (<24h old)	48h LC50		25			39.2	447	A,S,U	S	Hoke et al 1992
Water flea (*Ceriodaphnia dubia*) (<24h old)	48h LC50	7.5-9	25	>40% saturation			1169 (1830 as CaCl2)	A	S	Mount et al 1997
Water flea (*Ceriodaphnia dubia*)	48h LC50						1,595	A	?	WI SLOH, 1995 (In Nagpal et al., 2003)
Water flea (*Ceriodaphnia dubia*)	48h LC50					84.8	1,677	A,S,U	?	WISLOH 2007 (In EPA 2008 Cl update dataset)
Water flea (*Ceriodaphnia dubia*)	48h LC50					169.5	1,499	A,S,U	?	WISLOH 2007 (In EPA 2008 Cl update dataset)
Water flea (*Ceriodaphnia dubia*) (neonates, < 24 hr old)	48h LC50					39.2 (presented as SRW in EPA ref)	1,395	A,S,U	?	Data from ERL-Dudlth (In EPA Cl 2008 update In USEPA 1991)

Water flea (*Ceriodaphnia dubia*) (neonates, < 24 hr old)	48h LC50					39.2 (presented as SRW in EPA ref)	1,638	A,S,U	?	Data from ERL-Dudlth (In EPA Cl 2008 update In USEPA 1991)
Water flea (*Ceriodaphnia dubia*) (neonates, < 24 hr old)	48h LC50					39.2 (presented as SRW in EPA ref)	1,274	A,S,U	?	Data from ERL-Dudlth (In EPA Cl 2008 update In USEPA 1991)
Water flea (*Ceriodaphnia dubia*) (neonates, < 24 hr old)	48h LC50					39.2 (presented as SRW in EPA ref)	1,395	A,S,U	?	Data from ERL-Dudlth (In EPA Cl 2008 update In USEPA 1991)
Water flea (*Ceriodaphnia dubia*) (neonates, < 24 hr old)	48h LC50					339 (presented as VHRW in EPA ref)	1,698	A,S,U	?	Data from ERL-Dudlth (In EPA Cl 2008 update In USEPA 1991)
Water flea (*Ceriodaphnia dubia*) (neonates, < 24 hr old)	48h LC50	7.9	24-26	7.83	68	30	947	A,S	P	GLEC and INHS 2008
Water flea (*Ceriodaphnia dubia*) (neonates, < 24 hr old)	48h LC50	8.1	24-26	7.69	68	44	955	A,S	P	GLEC and INHS 2008
Water flea (*Ceriodaphnia dubia*) (neonates, < 24 hr old)	48h LC50	8.1	24-26	7.76	64	96	1,130	A,S	P	GLEC and INHS 2008
Water flea (*Ceriodaphnia dubia*) (neonates, < 24 hr old)	48h LC50	8.0	24-26	7.91	68	180	1,609	A,S	P	GLEC and INHS 2008

Water flea (*Ceriodaphnia dubia*) (neonates, < 24 hr old)	48h LC50	8.0	24-26	8.28	60	400	1,491	A,S	P	GLEC and INHS 2008
Water flea (*Ceriodaphnia dubia*) (neonates, < 24 hr old)	48h LC50	8.0	24-26	7.79	64	570	1,907	A,S	P	GLEC and INHS 2008
Water flea (*Ceriodaphnia dubia*) (neonates, < 24 hr old)	48h LC50	8.0	24-26	7.97	64	800	1,764	A,S	P	GLEC and INHS 2008
Water flea (*Ceriodaphnia dubia*) (neonates, < 24 hr old)	48h LC50	8.0	24-26	7.61	64	25	1,007	A,S	P	GLEC and INHS 2008
Water flea (*Ceriodaphnia dubia*) (neonates, < 24 hr old)	48h LC50	7.9	24-26	7.81	65	49	767	A,S	P	GLEC and INHS 2008
Water flea (*Ceriodaphnia dubia*) (neonates, < 24 hr old)	48h LC50	8.0	24-26	7.72	64	95	1,369	A,S	P	GLEC and INHS 2008
Water flea (*Ceriodaphnia dubia*) (neonates, < 24 hr old)	48h LC50	8.1	24-26	7.43	66	194	1,195	A,S	P	GLEC and INHS 2008
Water flea (*Ceriodaphnia dubia*) (neonates, < 24 hr old)	48h LC50	7.9	24-26	7.55	62	375	1,687	A,S	P	GLEC and INHS 2008

Water flea (*Ceriodaphnia dubia*) (neonates, < 24 hr old)	48h LC50	7.9	24-26	8.06	64	560	1,652	A,S	P	GLEC and INHS 2008
Water flea (*Ceriodaphnia dubia*) (neonates, < 24 hr old)	48h LC50	8.2	24-26	7.42	65	792	1,909	A,S	P	GLEC and INHS 2008
Water flea (*Ceriodaphnia dubia*) (neonates, < 24 hr old)	48h LC50	8.0	24-26	8.36	64	280	1,400	A,S	P	GLEC and INHS 2008
Water flea (*Ceriodaphnia dubia*) (neonates, < 24 hr old)	48h LC50	8.1	24-26	8.5	64	280	1,720	A,S	P	GLEC and INHS 2008
Water flea (*Ceriodaphnia dubia*) (neonates, < 24 hr old)	48h LC50	8.1	24-26	8.21	64	280	1,394	A,S	P	GLEC and INHS 2008
Water flea (*Ceriodaphnia dubia*) (neonates, < 24 hr old)	48h LC50	8.2	24-26	8.21	64	280	1,500	A,S	P	GLEC and INHS 2008
Water flea (*Ceriodaphnia dubia*) (neonates, < 24 hr old)	48h LC50	8.0	24-26	8.54	64	280	1,109	A,S	P	GLEC and INHS 2008
Water flea (*Ceriodaphnia dubia*) (neonates, < 24 hr old)	48h LC50	8.0	24-26	8.27	64	280	1,206	A,S	P	GLEC and INHS 2008
Water flea (*Ceriodaphnia dubia*) (neonates, < 24 hr old)	48h LC50	8.0	24-26	7.48	64	279	1,311	A,S	P	GLEC and INHS 2008

Water flea (*Ceriodaphnia dubia*) (neonates, < 24 hr old)	48h LC50	7.9	24-26	7.5	63	276	1,258	A,S	P	GLEC and INHS 2008
Water flea (*Ceriodaphnia dubia*) (neonates, < 24 hr old)	48h LC50	8.0	24-26	7.32	63	283	1,240	A,S	P	GLEC and INHS 2008
Water flea (*Ceriodaphnia dubia*) (neonates, < 24 hr old)	48h LC50	8.0	24-26	7.65	66	281	1,214	A,S	P	GLEC and INHS 2008
Water flea (*Ceriodaphnia dubia*) (neonates, < 24 hr old)	48h LC50	7.8	24-26	7.42	64	290	1,199	A,S	P	GLEC and INHS 2008
Water flea (*Ceriodaphnia dubia*) (neonates, < 24 hr old)	48h LC50	8.0	24-26	7.2	65	278	1,179	A,S	P	GLEC and INHS 2008
Water flea (*Ceriodaphnia dubia*) (neonates, < 24 hr old)	48h LC50	7.6-8.0	24-26	8.5-8.7	60	76 (MHSW)	1,068	A,M	P	Elphick et al 2011
Water flea (*Ceriodaphnia dubia*)	48h LC50		25	8±1.5		170	1,395	A	S	Cowgill and Milazzo, 1990
Water flea *(Daphnia magna)* (<24h old)	24h LC50	7.5-9	20	>40% saturation		control/dilution water for tests was MHRW (80-100 mg CaCO3/L)	2076 (3250 as CaCl2)	A	S	Mount et al 1997

Water flea *(Daphnia magna)* (<24h old)	48h LC50	7.5-9	20	>40% saturation		control/dilution water for tests was MHRW (80-100 mg CaCO3/L)	1770 (2770 as CaCl2)	A	S	Mount et al 1997
Water flea (*Daphnia magna*) (<24 hr neonate)	48h LC50	7.6-8.0	19-21	8.5-8.7	58	98	3,630	A,M	P	Elphick et al 2011
Water flea (*Daphnia magna*) (<24 hr neonate)	48h LC50						3,731	A,S,M	S	Jackman 2010
Water flea (*Daphnia magna*) (<24 hr neonate)	48h LC50					ASTM hard	3,458	A,S,M	S	Wang and Ingersoll 2010
Water flea (*Daphnia magna*) (<24 hr neonate)	48h LC50	7.69	20±2	8.7		136	3,559	A,S,M	S	Dow et al. 2010 (historical mean reference toxicity data from ASI Group Ltd - Appendix II)
Water flea *Daphnia magna* (water flea)	48h LC50		25	8±1.5		170	4,704	A	S	Cowgill and Milazzo, 1990
Water flea (*Daphnia magna*) (life stage not reported)	48h LC50					46 (filtered University lake water).	2,008	A	U (no mention of control survival)	Dowden and Bennett 1965
Water flea (*Daphnia magna*) (neonates, < 24 hr old)	48h LC50	7.5-9	20	>40% saturation		84.8 (Mod Hard Recon Water)	2,893	A,S,U	S	Mount et al 1997
Water flea (*Daphnia magna)* (neonates, 12±12 hrs old)	48h LC50	7.74	18±1	9	42.3	45.3	2,563	A,S	S	Biesinger and Christensen 1972
Water flea *(Daphnia magna)* (<24h old)	48h LC50		20				2,776	A,R	U (kept in dark)	Arambasic et al. 1995
Water flea *(Daphnia magna)* (<24h old)	24h LC50	7.5-9	20	>40% saturation		84.8 (Mod Hard Recon Water)	3,870	A,S,U	S	Mount et al. 1997

Water flea (*Daphnia magna*) (life stage not reported)	100h LC50					Standard Reference Water	1,889	A	U (no mention of control survival)	Dowden and Bennett 1965
Water flea *(Daphnia magna)*	48h EC50 Immobilization	7.2-7.8	11.5-14.5	5.2-6.5	390-415	240	621	A,S,U	S	Khangarot and Ray 1989
Water flea *(Daphnia magna)* (<24h old)	48h LC50					39.2	3,038	A,S,U	S	Hoke et al 1992
Water flea *(Daphnia magna)* (<24h old)	48h LC50					39.2	2,726	A,S,U	S	Hoke et al 1992
Water flea *(Daphnia magna)* (4th instar - adult)	48h LC50					39.2	2,053	A,S,U	S	Hoke et al 1992
Water flea *(Daphnia magna)*	48h LC50					?	1,008	A	U (data from original study proprietary)	Cowgill 1987 (In EPA Iowa update)
Water flea *(Daphnia magna)*	48h LC50					?	3,319	A	U (data from original study proprietary)	Cowgill 1987 (In EPA Iowa update)
Water flea *(Daphnia magna)*	48h LC50					108.7	<2,548	A,S,U	U (no control data)	Anderson 1946
Water flea *(Daphnia magna)* (<24h old)	64h LC50	8.2-8.4	25			108.7	2,232	A,S,U	U (no control data)	Anderson 1948a (In EPA Iowa update)
Water flea *(Daphnia magna)*	50h LC50					41.5	3,563	A,S,U	U (no mention of control survival)	Dowden and Bennett 1965
Water flea *(Daphnia magna)* (<24h old)	48h LC50					45.3	2,529	A,S,U	S	Biesinger and Christensen 1972
Water flea *(Daphnia magna)* (<24h old)	48h LC50					45.3	2,806	A,S,U	S	Biesinger and Christensen 1972
Water flea *(Daphnia magna)* (<24h old)	48h LC50					169.5	>2,669	A,S,U	S	Seymour et al 1997

Water flea *(Daphnia magna)* (<24h old)	48h LC50					169.5	<3,943	A,S,U	S	Seymour et al 1997
Water flea *(Daphnia magna)*	48h LC50					169.5	3,944	A,S,U	S()	WISLOH 2007 (In EPA 2008 Cl update dataset)
Water flea *(Daphnia magna)* (<24h old)	48h LC50	7.81±0.13	19-21	>5.0	62.6±3.9	82.9±5.8	3,009	A,S,U	S	Valenti et al 2007
Water flea *(Daphnia magna)* (<24h old)	48h LC50	7.5-8.1	20	≥80% saturation		100 (Ca:Mg = 0.7)	3,136	A,S,U	S	Davies and Hall 2007
Water flea *(Daphnia magna)* (<24h old)	48h LC50	7.5-8.1	20	≥80% saturation		100 (Ca:Mg = 1.8)	3,222	A,S,U	S	Davies and Hall 2007
Water flea *(Daphnia magna)* (<24h old)	48h LC50	7.5-8.1	20	≥80% saturation		100 (Ca:Mg = 7.0)	3,137	A,S,U	S	Davies and Hall 2007
Water flea *(Daphnia magna)* (2-3d old)	24h LC0						33	A	U (control survival not reported)	Stom and Zubareva 1994
Water flea (*Daphnia magna)* (4±4 hours old, to test effects at first molting)	64h EC50 Immobilization	8.2-8.4	25			Lake Erie water	2,232	A	S	Anderson 1948a
Water flea *(Dahpnia pulex)* (<24h old)	24h EC50 Immobility						2,000	A	S	Lilius et al. 1995
Water flea *(Daphnia pulex)*	48h LC50					84.8	1,159	A,S,U	S	Palmer et al 2004
Water flea *(Daphnia pulex)*	48h LC50					84.8	1,775	A,S,U	S	Palmer et al 2004
Water flea *(Daphnia pulex)*	48h LC50					84.8	1,805	A,S,U	S	Palmer et al 2004
Water flea *(Daphnia pulex)*	48h LC50					84.8	2,242	A,S,U	S	Palmer et al 2004

Water flea *(Daphnia pulex)*	48h LC50	7.83±0.09	20.0±0.1	8.7±0.1	60.8±2.3	92.8±2.6 (ASTM recon water)	892	A,S,M	S	Birge et al, 1985
Water flea *(Daphnia pulex)*	48h LC50	8.5±0.1	20.2±0.5	8.3±0.1	227±5	261±5 (natural water)	1,880	A,S,M	U (natural water used as exposure water)	Birge et al, 1985
Water flea (*Daphnia ambigua*) (neonates, < 24 hr old)	Immobilization (48-h, EC50)	8.11-8.66	19-23	7.46-9.14	56-76	54-72	1,213	A, S, M	S	Harmon et al., 2003
Water flea *(Daphnia hyalina)* (adult avg length of 1.27 mm)	48h LC50	7.2	9.5-10.5	air saturated	10.4	33	5308 (3000 mg Ca/L as CaCl2*2H2O)	A,U	S	Baudouin and Scoppa 1974
Water flea (*Daphnia longispina)*	Mortality (66-h)						1,772	A	?	Fowler 1931 (In Anderson 1948, in Evans and Frick 2001)
Zebra mussel *(Dreissena polymorphia)*	100% Mortality (Veligers, 6-h)	8.2±0.5	17±1	≥60% saturation	100±10	130-150	6,066	A,S,U	U (invasive species, tolerant of high salinity)	Waller et al. 1996
Zebra mussel *(Dreissena polymorphia)*	100% Mortality (Veligers6-h)	8.2±0.5	17±1	≥60% saturation	100±10	130-150	12,132	A,S,U	U (invasive species, tolerant of high salinity)	Waller et al. 1996
Zebra mussel *(Dreissena polymorphia)*	100% Mortality (Veligers, 12-h)	8.2±0.5	12±1	≥60% saturation	100±10	130-150	6,066	A,S,U	U (invasive species, tolerant of high salinity)	Waller et al. 1996
Zebra mussel *(Dreissena polymorphia)*	70% Mortality (Settlers, 6-h)	8.2±0.5	17±1	≥60% saturation	100±10	130-150	6,066	A,S,U	U (invasive species, tolerant of high salinity)	Waller et al. 1996
Zebra mussel *(Dreissena polymorphia)*	99% Mortality (Settlers, 6-h)	8.2±0.5	17±1	≥60% saturation	100±10	130-150	12,132	A,S,U	U (invasive species, tolerant of high salinity)	Waller et al. 1996

Zebra mussel *(Dreissena polymorphia)*	98% Mortality (Settlers, 12-h)	8.2±0.5	12±1	≥60% saturation	100±10	130-150	6,066	A,S,U	U (invasive species, tolerant of high salinity)	Waller et al. 1996

Assign 3 data codes, one from each of the following rows:

A-acute	C-chronic	
S-static	R-static renewal	F-flowthrough
U-unmeasured nominal conc.		M-measured conc.

Data Quality
- U- Unacceptable
- P- Primary
- S- Secondary
- ? - Unclassified (original document could not be obtained for review)

[a]- Value determined by regression

[d] Values are the results of an inter and intra-laboratory study to evaluate variability in the performance of the 7-d *Ceriodaphnia dubia* survival and reproduction test. The study involved 11 laboratories (4 of which performed the studies in replicate) an

MATC: The Maximum Acceptable Toxicant Concentration

** Endangered species in Canada as designated by the Committee on the Status of Endangered Wildlife in Canada (COSEWIC). Species at Risk Permit Number SECT 06 SCI 007.

Long-Term Aquatic Toxicity Data Table										
Compound: Sodium Chloride and Calcium Chloride										
Species (Life Stage)	**Response**	**pH**	**Temperature (°C)**	**Dissolved Oxygen (mg/L)**	**Alkalinity**	**Hardness (mg $CaCO_3$/L)**	**Effect Concentration (mg Cl/L)**	**Data Codes**	**Data Quality**	**Reference**
CHRONIC - VERTEBRATES										
African clawed frog (*Xenopus laevis*) (tadpoles, <2wks old)	7d NOEC (7d 90-97% Survival)	7.68-8.32	23±1	8.0-8.7	80-90	110-120	1,213	C,R	P	Beak International Inc. 1999
African clawed frog (*Xenopus laevis*) (tadpoles, <2wks old)	7d LOEC (7d 6.7% survival)	7.68-8.32	23±1	8.0-8.7	80-90	110-120	2,426	C,R	P	Beak International Inc. 1999
African clawed frog (*Xenopus laevis*) (tadpoles, <2wks old)	7d MATC (survival)	7.68-8.32	23±1	8.0-8.7	80-90	110-120	1,715	C,R	P	Beak International Inc. 1999
African clawed frog (*Xenopus laevis*) (tadpoles, <2wks old)	7d LC10	7.68-8.32	23±1	8.0-8.7	80-90	110-120	1,307	C,R	P	Beak International Inc. 1999
African clawed frog (*Xenopus laevis*) (frog embryo)	7d LC50	7.68-8.32	23±1	8.0-8.7	80-90	110-120	1,783	C,R	P	Beak International Inc. 1999
African clawed frog (*Xenopus laevis*) (frog embryo)	7d EC50 (impaired swimming behaviour)	7.68-8.32	23±1	8.0-8.7	80-90	110-120	1,523	C,R	P	Beak International Inc. 1999
African clawed frog (*Xenopus laevis*) (tadpoles, <2wks old)	7d LC100 (0% Survival)	7.68-8.32	23±1	8.0-8.7	80-90	110-120	4,853	C,R	P	Beak International Inc. 1999
African clawed frog (*Xenopus laevis*) (Gosner stage 47-49 tadpoles, mean wt 0.008 ± 0.001g)	7d NOEC (survival)						80	C,S,U	S	Dougherty and Smith 2006

African clawed frog (*Xenopus laevis*) (Gosner stage 47-49 tadpoles, mean wt 0.008 ± 0.001g)	7d LOEC (survival)						>80	C,S,U	S	Dougherty and Smith 2006
African clawed frog (*Xenopus laevis*) (Gosner stage 47-49 tadpoles, mean wt 0.008 ± 0.001g)	7d LC50						799	C,S,U	S	Dougherty and Smith 2006
American toad (*Bufo americanus*) (Gosner stage 25 tadpoles, mean wt 0.012 ± 0.001g)	7d LOEC (survival)						>80	C,S,U	S	Dougherty and Smith 2006
American toad (*Bufo americanus*) (Gosner stage 25 tadpoles, mean wt 0.012 ± 0.001g)	7d NOEC (survival)						80	C,S,U	S	Dougherty and Smith 2006
Green frog (*Rana clamitans)* (Gosner stage 25 tadpoles, mean wt 0.017 ± 0.001g)	7d LC50						246	C,S,U	S	Dougherty and Smith 2006
common frog (*Rana temporaria*) (Gosner stage 26 tadpoles)	56d NOEC (survival)		16			recon soft water	910	C,R,U	S	Denoel et al 2010
common frog (*Rana temporaria*) (Gosner stage 26 tadpoles)	56d NOEC (growth)		16			recon soft water	910	C,R,U	S	Denoel et al 2010
common frog (*Rana temporaria*) (Gosner stage 26 tadpoles)	42d NOEC (behavioural endpoint, mean swimming speed)		16			recon soft water	607	C,R,U	S	Denoel et al 2010
common frog (*Rana temporaria*) (Gosner stage 26 tadpoles)	42d LOEC (behavioural endpoint, mean swimming speed)		16			recon soft water	910	C,R,U	S	Denoel et al 2010
common frog (*Rana temporaria*) (Gosner stage 26 tadpoles)	42d MATC (behavioural endpoint, mean swimming speed)		16			recon soft water	743	C,R,U	S	Denoel et al 2010

common frog (*Rana temporaria*) (Gosner stage 26 tadpoles)	56d NOEC (behavioural endpoint, mean swimming speed)		16			recon soft water	303	C,R,U	S	Denoel et al 2010
common frog (*Rana temporaria*) (Gosner stage 26 tadpoles)	56d LOEC (behavioural endpoint, mean swimming speed)		16			recon soft water	607	C,R,U	S	Denoel et al 2010
common frog (*Rana temporaria*) (Gosner stage 26 tadpoles)	56d MATC (behavioural endpoint, mean swimming speed)		16			recon soft water	429	C,R,U	S	Denoel et al 2010
common frog (*Rana temporaria*) (Gosner stage 26 tadpoles)	56d LC10 (behavioural endpoint, mean swimming speed) (linear interpolation)		16			recon soft water	377	C,R,U	S	Denoel et al 2010
common frog (*Rana temporaria*) (Gosner stage 26 tadpoles)	56d NOEC (behavioural endpoint, total distance moved)		16			recon soft water	303	C,R,U	S	Denoel et al 2010
common frog (*Rana temporaria*) (Gosner stage 26 tadpoles)	56d LOEC (behavioural endpoint, total distance moved)		16			recon soft water	607	C,R,U	S	Denoel et al 2010
common frog (*Rana temporaria*) (Gosner stage 26 tadpoles)	56d MATC (behavioural endpoint, total distance moved)		16			recon soft water	429	C,R,U	S	Denoel et al 2010
common frog (*Rana temporaria*) (Gosner stage 26 tadpoles)	56d LC10 (behavioural endpoint, total distance moved)		16			recon soft water	292	C,R,U	S	Denoel et al 2010
Frog *(Rana breviceps)*	Mortality (6-d NOEC)	5.95	not reported	not reported	8	26	1,456	C	U (pH too low, test temp not reported, not representative of a temperate species)	Mahajan et al. 1979
Frog (*Rana breviceps)*	Mortality (6-d LOEC)	5.6	not reported	not reported	8	20	2,184	C	U (pH too low, test temp not reported, not representative of a temperate species)	Mahajan et al. 1979

Frog (*Rana breviceps)*	Mortality (5-d NOEC)	5.6	not reported	not reported	8	26	1,699	C	U (pH too low, test temp not reported, not representative of a temperate species)	Mahajan et al. 1979
Frog (*Rana breviceps)*	Mortality (5-d LOEC)	5.6	not reported	not reported	8	20	2,548	C	U (pH too low, test temp not reported, not representative of a temperate species)	Mahajan et al. 1979
Spotted salamander (*Ambystoma maculatum*) entire larval life stage exposure	LOEC Weight at metamorphosis (49d exposure)		8-12			local pond water	300	C,R,U	U (wide range between NOEC and LOEC, and really only 2 test concentrations - 8 (control), 300, 900 mg Cl/L)	Russell and Collins 2009
Spotted salamander (*Ambystoma maculatum*) entire larval life stage exposure	NOEC Weight at metamorphosis (49d exposure)		8-12			local pond water	8	C,R,U	U (wide range between NOEC and LOEC, and really only 2 test concentrations - 8 (control), 300, 900 mg Cl/L)	Russell and Collins 2009
Spotted salamander (*Ambystoma maculatum*) entire larval life stage exposure	MATC Weight at metamorphosis (49d exposure)		8-12			local pond water	49	C,R,U	U (wide range between NOEC and LOEC, and really only 2 test concentrations - 8 (control), 300, 900 mg Cl/L)	Russell and Collins 2009
Spotted salamander (*Ambystoma maculatum*) entire larval life stage exposure	LOEC survival (60%) of larvae (49d exposure)		8-12			local pond water	300	C,R,U	U (wide range between NOEC and LOEC, and really only 2 test concentrations - 8 (control), 300, 900 mg Cl/L)	Russell and Collins 2009

Spotted salamander (*Ambystoma maculatum*) entire larval life stage exposure	NOEC survival (100%) of larvae (49d exposure)		8-12			local pond water	8	C,R,U	U (wide range between NOEC and LOEC, and really only 2 test concentrations - 8 (control), 300, 900 mg Cl/L)	Russell and Collins 2009
Spotted salamander (*Ambystoma maculatum*) entire larval life stage exposure	49d MATC (survival of larvae)		8-12			local pond water	49	C,R,U	U (wide range between NOEC and LOEC, and really only 2 test concentrations - 8 (control), 300, 900 mg Cl/L)	Russell and Collins 2009
Spotted salamander (*Ambystoma maculatum*) entire larval life stage exposure	LOEC Larval period extended (73d exposure)		8-12			local pond water	900	C,R,U	U (wide range between NOEC and LOEC, and really only 2 test concentrations - 8 (control), 300, 900 mg Cl/L)	Russell and Collins 2009
Spotted salamander (*Ambystoma maculatum*) entire larval life stage exposure	NOEC Larval period extended (73d exposure)		8-12			local pond water	8	C,R,U	U (wide range between NOEC and LOEC, and really only 2 test concentrations - 8 (control), 300, 900 mg Cl/L)	Russell and Collins 2009
Spotted salamander (*Ambystoma maculatum*) entire larval life stage exposure	MATC Larval period extended (73d exposure)		8-12			local pond water	85	C,R,U	U (wide range between NOEC and LOEC, and really only 2 test concentrations - 8 (control), 300, 900 mg Cl/L)	Russell and Collins 2009

Spotted salamander (*Ambystoma maculatum*) egg clutches	18 d NOEC (increase in egg clutch mass by 25%)		11			120 (Syracuse, NY dechlor tap water)	1 (control)		U (road deicing salt used)	Karraker and Gibbs 2010
Spotted salamander (*Ambystoma maculatum*) egg clutches	18d LOEC (decrease in egg clutch mass by 2%)		11			120 (Syracuse, NY dechlor tap water)	145 (moderate)		U (road deicing salt used)	Karraker and Gibbs 2010
Spotted salamander (*Ambystoma maculatum*) egg clutches	18d MATC (change in egg clutch mass)		11			120 (Syracuse, NY dechlor tap water)	12		U (road deicing salt used)	Karraker and Gibbs 2010
Spotted salamander (*Ambystoma maculatum*) egg clutches	18 d effect conc (decrease in egg clutch mass by 45%)		11			120 (Syracuse, NY dechlor tap water)	945 (high)		U (road deicing salt used)	Karraker and Gibbs 2010
Northern leopard frog (*Rana pipiens*) eggs	108d NOEC (developmental delays)		21-25			80-100 (mod hard recon water)	1,941	C,S,M	S	Doe 2010
Northern leopard frog (*Rana pipiens*) eggs	108d NOEC (wet weight at Gosner Stage 42; forelimbs emerge)		21-25			80-100 (mod hard recon water)	1,941	C,S,M	S	Doe 2010
Northern leopard frog (*Rana pipiens*) eggs	108d NOEC (survival)		21-25			80-100 (mod hard recon water)	1,941	C,S,M	S	Doe 2010
Northern leopard frog (*Rana pipiens*) eggs	108d LOEC (survival)		21-25			80-100 (mod hard recon water)	6,066	C,S,M	S	Doe 2010
Northern leopard frog (*Rana pipiens*) eggs	108d MATC (survival)		21-25			80-100 (mod hard recon water)	3,431	C,S,M	S	Doe 2010
Northern leopard frog (*Rana pipiens*) eggs	4d LC50		21-25			80-100 (mod hard recon water)	3,397	C,S,M	S	Doe 2010
Northern leopard frog (*Rana pipiens*) eggs	7d LC50		21-25			80-100 (mod hard recon water)	3,397	C,S,M	S	Doe 2010

Northern leopard frog (*Rana pipiens*) eggs	180d LC50		21-25			80-100 (mod hard recon water)	2,265	C,S,M	S	Doe 2010
Northern leopard frog (*Rana pipiens*) eggs	108d LC10		21-25			80-100 (mod hard recon water)	4,233	C,S,M	U (LC10 > LC50)	Doe 2010
Bass (*Morone* sp.)	14d LC0						8,492	C	U	Black 1950
Bluegill sunfish (*Lepomis macrochirus*) (young-of-the-year, avg wet wt = 1.03±0.50g, avg lt = 4.37±0.59cm)	12d LC50	7.37-7.87	18.8-20.1	>40% saturation	54-59	74-116	7,401	C,F	U (no replication of test concentrations, control survival not listed)	Kszos et al. 1990
Brown trout (*Salmo trutta fario*) (fingerlings, approx 2 months old)	8d NOEC (Survival)	7.63	15-16	10.1	32.2	23	607	C,S,M	S (highest concentration tested produced no effect_see CCME 2007 protocol for direction)	Camargo and Tarazona 1991
Fathead minnow (*Pimephales promelas)* (eggs, 6-12 hours post-fertilization)	33d LC80 (mean survival ca. 20%)	7.5±0.22	25±0.3	7.6±0.7	61.6±4.0	96.9±8.7 (ASTM recon water)	1001	C,F,M	S	Birge et al. 1985
Fathead minnow (*Pimephales promelas)* (eggs, 6-12 hours post-fertilization)	33d LC100 (no survival)	7.5±0.22	25±0.3	7.6±0.7	61.6±4.0	96.9±8.7 (ASTM recon water)	1400	C,F,M	S	Birge et al. 1985
Fathead minnow (*Pimephales promelas)* (eggs, 6-12 hours post-fertilization)	33d NOEC (Survival, Table 5) (NB: Tables A19,20,21 shows this to be a NOEC for survival, length & weight respectively)	7.5±0.22	25±0.3	7.6±0.7	61.6±4.0	96.9±8.7 (ASTM recon water)	252	C,F,M	S (as per Table 5 in Birge et al 1985_values below provide better representation)	Birge et al. 1985

Fathead minnow (*Pimephales promelas)* (eggs, 6-12 hours post-fertilization)	33d LOEC (Survival, Table 5) (NB: Tables A19,20,21 shows this to be a NOEC for survival, length & weight respectively - not sure why listed as a LOEC in Birge et al 1985)	7.5±0.22	25±0.3	7.6±0.7	61.6±4.0	96.9±8.7 (ASTM recon water)	352	C,F,M	S (as per Table 5 in Birge et al 1985_values below provide better representation)	Birge et al. 1985
Fathead minnow (*Pimephales promelas)* (eggs, 6-12 hours post-fertilization)	33d MATC (Survival, Table 5) (NB: not a real MATC value because 252 & 352 mg Cl/L are both NOEC values)	7.5±0.22	25±0.3	7.6±0.7	61.6±4.0	96.9±8.7 (ASTM recon water)	298	C,F,M	S (as per Table 5 in Birge et al 1985_values below provide better representation)	Birge et al. 1985
Fathead minnow (*Pimephales promelas)* (eggs, 6-12 hours post-fertilization)	33d NOEC (Survival, lowest of 5 reps in Table A19, rep IV)	7.5±0.22	25±0.3	7.6±0.7	61.6±4.0	96.9±8.7 (ASTM recon water)	352	C,F,M	S	Birge et al. 1985
Fathead minnow (*Pimephales promelas)* (eggs, 6-12 hours post-fertilization)	33d LOEC (Survival, lowest of 5 reps in Table A19, rep IV)	7.5±0.22	25±0.3	7.6±0.7	61.6±4.0	96.9±8.7 (ASTM recon water)	~~533~~ 528	C,F,M	S	Birge et al. 1985
Fathead minnow (*Pimephales promelas)* (eggs, 6-12 hours post-fertilization)	33d MATC (Survival, lowest of 5 reps in Table A19)	7.5±0.22	25±0.3	7.6±0.7	61.6±4.0	96.9±8.7 (ASTM recon water)	~~433~~ 431	C,F,M	S	Birge et al. 1985
Fathead minnow (*Pimephales promelas)* (eggs, 6-12 hours post-fertilization)	33d NOEC (Survival, mean of 5 reps in Table A19)	7.5±0.22	25±0.3	7.6±0.7	61.6±4.0	96.9±8.7 (ASTM recon water)	498	C,F,M	S	Birge et al. 1985
Fathead minnow (*Pimephales promelas)* (eggs, 6-12 hours post-fertilization)	33d LOEC (Survival, mean of 5 reps in Table A19)	7.5±0.22	25±0.3	7.6±0.7	61.6±4.0	96.9±8.7 (ASTM recon water)	693	C,F,M	S	Birge et al. 1985
Fathead minnow (*Pimephales promelas)* (eggs, 6-12 hours post-fertilization)	33d MATC (Survival, mean of 5 reps in Table A19)	7.5±0.22	25±0.3	7.6±0.7	61.6±4.0	96.9±8.7 (ASTM recon water)	587	C,F,M	S	Birge et al. 1985
Fathead minnow (*Pimephales promelas)* (eggs, 6-12 hours post-fertilization)	33d LC10 (Survival, lowest of 5 reps in Table A19)	7.5±0.22	25±0.3	7.6±0.7	61.6±4.0	96.9±8.7 (ASTM recon water)	**598**	C,F,M	S	Birge et al. 1985 (Point estimates were calculated by Elphick et al (2011) by using Multiple Linear Estimation (Probit) based on original data provided in Birge et al (1985))
Fathead minnow (*Pimephales promelas)* (eggs, 6-12 hours post-fertilization)	33d NOEC (Growth)	7.5±0.22	25±0.3	7.6±0.7	61.6±4.0	96.9±8.7 (ASTM recon water)	533	C,F,M	S	Birge et al. 1985

Fathead minnow (*Pimephales promelas)* (eggs, 6-12 hours post-fertilization)	33d LOEC (Growth)	7.5±0.22	25±0.3	7.6±0.7	61.6±4.0	96.9±8.7 (ASTM recon water)	734	C,F,M	S	Birge et al. 1985
Fathead minnow (*Pimephales promelas)* (eggs, 6-12 hours post-fertilization)	33d MATC (Growth)	7.5±0.22	25±0.3	7.6±0.7	61.6±4.0	96.9±8.7 (ASTM recon water)	625	C,F,M	S	Birge et al. 1985
Fathead minnow (*Pimephales promelas)*	7d IC25						1,752	C	U (original reference not obtained)	WISLOH (2007) as cited in Iowa Chlorid Criteria Update 2009 (US EPA)
Fathead minnow (*Pimephales promelas)*	7d NOEC (survival)						1,274	C	S (US EPA ref tox test data)	Diamond et al (1992)
Fathead minnow (*Pimephales promelas)*	7d NOEC (survival)						2,002	C	S (US EPA ref tox test data)	Diamond et al (1992)
Fathead minnow (*Pimephales promelas)*	7d NOEC (survival)						1,597	C	S (US EPA ref tox test data)	Diamond et al (1992)
Fathead minnow (*Pimephales promelas)*	7d NOEC (survival)						1,577	C	S (US EPA ref tox test data)	Diamond et al (1992)
Fathead minnow (*Pimephales promelas)*	7d NOEC (survival)						2,002	C	S (US EPA ref tox test data)	Diamond et al (1992)
Fathead minnow (*Pimephales promelas)*	7d NOEC (survival)						1,777	C	S (US EPA ref tox test data)	Diamond et al (1992)
Fathead minnow (*Pimephales promelas)* (1-7d old)	7d MATC Population (biomass)	7.24-7.81	24-26	4.4-7.2	56-64	86-94	3,458	C,S,M	P	Pickering et al. 1996
Fathead minnow (*Pimephales promelas)* (1-7d old)	7d LOEC Survival	7.24-7.81	24-26	4.4-7.2	56-64	86-94	4,853	C,S,M	P	Pickering et al. 1996
Fathead minnow (*Pimephales promelas)* (1-7d old)	7d NOEC Survival	7.24-7.81	24-26	4.4-7.2	56-64	86-94	2,426	C,S,M	P	Pickering et al. 1996
Fathead minnow (*Pimephales promelas)* (1-7d old)	7d NOEC Growth	7.24-7.81	24-26	4.4-7.2	56-64	86-94	2,426	C,S,M	P	Pickering et al. 1996
Fathead minnow (*Pimephales promelas*) (larvae <24h)	7d NOEC (90% survival)	7.5-8.3	25±1	5.8-8.4	70-80	110-120	1,213	C,R	P	Beak International Inc. 1999

Fathead minnow (*Pimephales promelas*) (larvae <24h)	7d LOEC (72% survival)	7.5-8.3	25±1	5.8-8.4	70-80	110-120	2,426	C,R	P	Beak International Inc. 1999
Fathead minnow (*Pimephales promelas*) (larvae <24h)	7d MATC	7.5-8.3	25±1	5.8-8.4	70-80	110-120	1,715	C,R	P	Beak International Inc. 1999
Fathead minnow (*Pimephales promelas*) (larvae <24h)	7dIC25 (growth)	7.5-8.3	25±1	5.8-8.4	70-80	110-120	1,741	C,R	P	Beak International Inc. 1999
Fathead minnow (*Pimephales promelas*) (larvae <24h)	7d IC50 (growth)	7.5-8.3	25±1	5.8-8.4	70-80	110-120	3,027	C,R	P	Beak International Inc. 1999
Fathead minnow (*Pimephales promelas*) (larvae <24h)	7d LC50	7.5-8.3	25±1	5.8-8.4	70-80	110-120	3,330	C,R	P	Beak International Inc. 1999
Fathead minnow (*Pimephales promelas*) (larvae <24h)	7d impaired growth and swimming behaviour	7.5-8.3	25±1	5.8-8.4	70-80	110-120	2,426	C,R	P	Beak International Inc. 1999
Fathead minnow (*Pimephales promelas*) (embryo <36hrs old)	7d NOEC	8.1-8.3	25±1	7.8-8.3	80-100	120	607	C,R	P	Beak International Inc. 1999
Fathead minnow (*Pimephales promelas*) (embryo <36hrs old)	7d LEOC	8.1-8.3	25±1	7.8-8.3	80-100	120	1,213	C,R	P	Beak International Inc. 1999
Fathead minnow (*Pimephales promelas*) (embryo <36hrs old)	7d MATC	8.1-8.3	25±1	7.8-8.3	80-100	120	855	C,R	P	Beak International Inc. 1999
Fathead minnow (*Pimephales promelas*) (embryo <36hrs old)	7d EC50	8.1-8.3	25±1	7.8-8.3	80-100	120	874	C,R	P	Beak International Inc. 1999
Fathead minnow (*Pimephales promelas*) (embryos, < 3 hr post-fertilization)	Growth (34 d, NOEC, mean dry biomass)	7.32-8.22	24-26	4.9-9.6	52-60	80-100	558	C,R,M	P	Elphick et al 2011
Fathead minnow (*Pimephales promelas*) (embryos, <3 hr post-fertilization)	Growth (34 d, LOEC, mean dry biomass)	7.32-8.22	24-26	4.9-9.6	52-60	80-100	1,058	C,R,M	P	Elphick et al 2011
Fathead minnow (*Pimephales promelas*) (embryos, <3 hr post-fertilization)	Growth (34 d, MATC, mean dry biomass)	7.32-8.22	24-26	4.9-9.6	52-60	80-100	768	C,R,M	P	Elphick et al 2011
Fathead minnow (*Pimephales promelas*) (embryos, <3 hr post-fertilization)	Growth (34 d, MATC, mean dry biomass)	7.32-8.22	24-26	4.9-9.6	52-60	80-100	768	C,R,M	P	Rescan Environmental Services Ltd., 2007

Fathead minnow (*Pimephales promelas*) (embryos, < 3 hr post-fertilization)	Mortality (34 d, NOEC)	7.32-8.22	24-26	4.9-9.6	52-60	80-100	558	C,R,M	P	Rescan Environmental Services Ltd., 2007
Fathead minnow (*Pimephales promelas*) (embryos, <3 hr post-fertilization)	Mortality (34 d, LOEC)	7.32-8.22	24-26	4.9-9.6	52-60	80-100	1,058	C,R,M	P	Rescan Environmental Services Ltd., 2007
Fathead minnow (*Pimephales promelas*) (embryos, <3 hr post-fertilization)	Mortality (34 d, MATC)	7.32-8.22	24-26	4.9-9.6	52-60	80-100	768	C,R,M	P	Rescan Environmental Services Ltd., 2007
Fathead minnow (*Pimephales promelas*) (embryos, <3 hr post-fertilization)	Mortality (34 d, LC50)	7.32-8.22	24-26	4.9-9.6	52-60	80-100	792	C,R,M	P	Rescan Environmental Services Ltd., 2007
Fathead minnow (*Pimephales promelas*) (embryos, <3 hr post-fertilization)	Mortality (34 d, LC25)	7.32-8.22	24-26	4.9-9.6	52-60	80-100	699	C,R,M	P	Rescan Environmental Services Ltd., 2007
Fathead minnow (*Pimephales promelas*) (embryos, <3 hr post-fertilization)	Mortality (34 d, LC10)	7.32-8.22	24-26	4.9-9.6	52-60	80-100	585	C,R,M	P	Rescan Environmental Services Ltd., 2007
Fathead minnow (*Pimephales promelas*) (embryos, <3 hr post-fertilization)	Growth (34 d, EC25, mean dry biomass)	7.32-8.22	24-26	4.9-9.6	52-60	80-100	704	C,R,M	P	Elphick et al 2011
Fathead minnow (*Pimephales promelas*) (embryos, <3 hr post-fertilization)	Growth (34 d, EC50, mean dry biomass)	7.32-8.22	24-26	4.9-9.6	52-60	80-100	958	C,R,M	P	Elphick et al 2011
Fathead minnow (*Pimephales promelas*) (embryos, <3 hr post-fertilization)	Growth (34 d, NOEC, mean dry weight)	7.32-8.22	24-26	4.9-9.6	52-60	80-100	1,058	C,R,M	P	Rescan Environmental Services Ltd., 2007
Fathead minnow (*Pimephales promelas*) (embryos, <3 hr post-fertilization)	Growth (34 d, LOEC, mean dry weight)	7.32-8.22	24-26	4.9-9.6	52-60	80-100	>1,058	C,R,M	P	Rescan Environmental Services Ltd., 2007
Fathead minnow (*Pimephales promelas*) (embryos, <3 hr post-fertilization)	Growth (34 d, EC25, mean dry weight)	7.32-8.22	24-26	4.9-9.6	52-60	80-100	>1,058	C,R,M	P	Rescan Environmental Services Ltd., 2007
Fathead minnow (*Pimephales promelas*) (embryos, <3 hr post-fertilization)	Growth (34 d, EC50, mean dry weight)	7.32-8.22	24-26	4.9-9.6	52-60	80-100	>1,058	C,R,M	P	Rescan Environmental Services Ltd., 2007

Golden shiners (*Notemigonus crysoleucas*)	Average survival time (148-h, 6.2d)		22-22.5				3,033	C	U (some fish died from fungal infection)	Wiebe et al. 1934
Goldfish (*Carassius auratus*)	Mortality (≤10-d)						6,066	C	U	Ellis 1937
Goldfish *(Carassius auratus)*	Mortality (17-154-h)		21				7,097	C	U (original reference was not obtained, and also quite dated)	Powers 1917 (In Hammer 1977; Doudoroff and Katz; in Evans and Frick 2001)
Goldfish (*Carassius auratus*)	Mortality (NOEC, 25-d) (Mississippi River water)						3,033	C	U	Ellis 1937
Goldfish *(Carassius auratus)* (0.38-4.02g)	Mortality (10 d or 240h, LC50)	7.95±0.02	23.5	7.2±0.1	100±1.6	148.8±1.8	2,623	C,S,M	S	Threader and Houston 1983
Spotfin shiner (*Notropis spilopterus*)	5d LOEC (minimum lethal concentration)		18	≥4			1,517	C	U	van Horn et al. 1949
Lake Emerald shiner (*Notropis atheriniodes*)	5d LOEC (minimum lethal concentration)		18	≥4			1,517	C	U	Van Horn et al. 1949
Largemouth black bass (*Micropterus salmoides)*	0% Mortality (8.3-10.4-d)		22-22.5				3,033	C	U (fungal infection)	Wiebe et al. 1934
Largemouth black bass (*Micropterus salmoides)*	100% Mortality (142-148h, 5.9-6.2d)		22-22.5				6,066	C	U (fungal infection)	Wiebe et al. 1934
Rainbow trout (*Oncorhynchus mykiss*) (dry fertilized gametes)	Mortality (54 d, NOEC)	7.12-7.76	13-15	6.6-10.6	36-60	40-76	1,104	C,R,M	P	Rescan Environmental Services Ltd., 2007
Rainbow trout (*Oncorhynchus mykiss*) (dry fertilized gametes)	Mortality (54 d, LOEC)	7.12-7.76	13-15	6.6-10.6	36-60	40-76	2,327	C,R,M	P	Rescan Environmental Services Ltd., 2007
Rainbow trout (*Oncorhynchus mykiss*) (dry fertilized gametes)	Mortality (54 d,MATC)	7.12-7.76	13-15	6.6-10.6	36-60	40-76	1,603	C,R,M	P	Rescan Environmental Services Ltd., 2007

Rainbow trout (*Oncorhynchus mykiss*) (dry fertilized gametes)	Growth (54 d, NOEC, mean dry weight)	7.12-7.76	13-15	6.6-10.6	36-60	40-76	1,104	C,R,M	P	Rescan Environmental Services Ltd., 2007
Rainbow trout (*Oncorhynchus mykiss*) (dry fertilized gametes)	Growth (54 d, LOEC, mean dry weight)	7.12-7.76	13-15	6.6-10.6	36-60	40-76	>1,104	C,R,M	P	Rescan Environmental Services Ltd., 2007
Rainbow trout (*Oncorhynchus mykiss*) (dry fertilized gametes)	Growth (54 d, EC25, mean dry weight)	7.12-7.76	13-15	6.6-10.6	36-60	40-76	>1,104	C,R,M	P	Rescan Environmental Services Ltd., 2007
Rainbow trout (*Oncorhynchus mykiss*) (dry fertilized gametes)	Growth (54 d, EC50, mean dry weight)	7.12-7.76	13-15	6.6-10.6	36-60	40-76	>1,104	C,R,M	P	Rescan Environmental Services Ltd., 2007
Rainbow trout (*Oncorhynchus mykiss*) (dry fertilized gametes)	Growth (54d, NOEC mean dry biomass)	7.12-7.76	13-15	6.6-10.6	36-60	40-76	1,104	C,R,M	P	Elphick et al 2011
Rainbow trout (*Oncorhynchus mykiss*) (dry fertilized gametes)	Growth (54d, LOEC mean dry biomass)	7.12-7.76	13-15	6.6-10.6	36-60	40-76	2,327	C,R,M	P	Elphick et al 2011
Rainbow trout (*Oncorhynchus mykiss*) (dry fertilized gametes)	Growth (54d, MATC mean dry biomass)	7.12-7.76	13-15	6.6-10.6	36-60	40-76	1,603	C,R,M	P	Rescan Environmental Services Ltd., 2007
Rainbow trout (*Oncorhynchus mykiss*) (dry fertilized gametes)	Growth (54d, EC25, mean dry biomass)	7.12-7.76	13-15	6.6-10.6	36-60	40-76	1,174	C,R,M	P	Elphick et al 2011
Rainbow trout (*Oncorhynchus mykiss*) (dry fertilized gametes)	Growth (54 d, EC50, mean dry biomass)	7.12-7.76	13-15	6.6-10.6	36-60	40-76	1,559	C,R,M	P	Elphick et al 2011
Rainbow trout (*Oncorhynchus mykiss*) (dry fertilized gametes)	Mortality (54 d, LC50)	7.12-7.76	13-15	6.6-10.6	36-60	40-76	1,511	C,R,M	P	Rescan Environmental Services Ltd., 2007
Rainbow trout (*Oncorhynchus mykiss)* (embryo-larval)	7d EC25 (embryo viability)	7.5-8.3	14±1	9.8-10.2	100	120	989	C,R	P	Beak International Inc. 1999
Rainbow trout (*Oncorhynchus mykiss)* (embryo-larval)	7d EC50 (embryo viability)	7.5-8.3	14±1	9.8-10.2	100	120	1,456	C,R	P	Beak International Inc. 1999
Rainbow trout (*Oncorhynchus mykiss)* (embryo-alevin)	27d EC25 (embryo viability)	7.3-8.5	14±1	>9	100	120	1,110	C,R	P	Beak International Inc. 1999
Rainbow trout (*Oncorhynchus mykiss)* (embryo-alevin)	27d EC50 (embryo viability)	7.3-8.5	14±1	>9	100	120	1,595	C,R	P	Beak International Inc. 1999

Rainbow trout (*Oncorhynchus mykiss)* (fingerlings approx 2 months old)	8d NOEC (Survival)	7.63	15-16	10.1	32.2	23	485	C,S,M	S (highest concentration tested produced no effect_see CCME 2007 protocol for direction)	Camargo and Tarazona 1991
River shiner (*Notropis blennius)*	Mortality (215-576h, 9-24d)						1,517	C	U (original reference was not obtained, and also quite dated)	Garrey 1916 (In Hammer 1977 and Doudoroff and Katz 1953, In Evans and Frick 2001)
Wood frog (*Rana sylvatica*) (tadpoles)	Survivorship (70d NOEC)						145	C	U (field data using road salt)	Karraker et al. 2008
Wood frog (*Rana sylvatica*) (tadpoles)	Survivorship (70d LOEC)						945	C	U (field data using road salt)	Karraker et al. 2008
Wood frog (*Rana sylvatica*) (tadpoles)	Survivorship (10d NOEC)						47	C, R, U	U (control mortality <10% at day 10 but range in test concentrations too wide)	Sanzo and Hecnar, 2006
Wood frog (*Rana sylvatica*) (tadpoles)	Survivorship (10d LOEC)						625	C, R, U	U (control mortality <10% at day 10 but range in test concentrations too wide)	Sanzo and Hecnar, 2006
Wood frog (*Rana sylvatica*) (tadpoles)	Survivorship (10d MATC)						171	C, R, U	U (control mortality <10% at day 10 but range in test concentrations too wide)	Sanzo and Hecnar, 2006
Wood frog (*Rana sylvatica*) (tadpoles)	Survivorship (70d NOEC)						47	C, R, U	U (>50% control mortality)	Sanzo and Hecnar, 2006
Wood frog (*Rana sylvatica*) (tadpoles)	Survivorship (70d LOEC)						628	C, R, U	U (>50% control mortality)	Sanzo and Hecnar, 2006
Wood frog (*Rana sylvatica)* (tadpoles, Gosner stage 25)	Survivorship (90-d, NOEC)		18.7-19.3				47	C, R, U	U (>50% control mortality)	Sanzo and Hecnar, 2006
Wood frog (*Rana sylvatica)* (tadpoles, Gosner stage 25)	Mean time to metamorphosis (90-d, NOEC)		18.7-19.3				47	C, R, U	U (>50% control mortality)	Sanzo and Hecnar, 2006

Wood frog (*Rana sylvatica)* (tadpoles, Gosner stage 25)	Number of metamorphosed frogs (90-d NOEC)		18.7-19.3				47	C, R, U	U (>50% control mortality)	Sanzo and Hecnar, 2006
Wood frog (*Rana sylvatica)* (tadpoles, Gosner stage 25)	Body weight (90-d, NOEC)		18.7-19.3				625	C, R, U	U (>50% control mortality)	Sanzo and Hecnar, 2006
Wood frog (*Rana sylvatica)* (tadpoles, Gosner stage 25)	Decreased survival (90-d, LOEC, 17% decreased suvivorship compared 50% survivorship in controls and lower test concentrations)		18.7-19.3				625	C, R, U	U (>50% control mortality)	Sanzo and Hecnar, 2006
Wood frog (*Rana sylvatica)* (tadpoles, Gosner stage 25)	Mean time to metamorphosis (90-d, LOEC, decrease compared to controls)		18.7-19.3				625	C, R, U	U (>50% control mortality)	Sanzo and Hecnar, 2006
Wood frog (*Rana sylvatica)* (tadpoles, Gosner stage 25)	Number of metamorphosed frogs (90-d LOEC, decrease compared to controls)		18.7-19.3				625	C, R, U	U (>50% control mortality)	Sanzo and Hecnar, 2006
Yellow perch (*Perca flavescens*)	Survival (gradual increase in NaCl) (720h or 30d)						5,520-10,616	C	?	Young 1923 (In Hanes et al. 1970; in Evans and Frick 2001)
Yellow perch (*Perca flavescens*)	0% Mortality (14-d)						8,492	C	U	Black 1950 (In Hanes et al. 1970; in Evans and Frick 2001)
CHRONIC - INVERTEBRATES										
Freshwater mussel (*Villosa delumbis*)	24h EC10 (survival of glochidia)	8.32-8.61	20.1-21.9	>80% saturation	116-130	170-192	716	A,S,M	S	Bringolf et al 2007
Freshwater mussel (*Villosa delumbis*)	48h EC10 (survival of glochidia)	8.32-8.61	20.1-21.9	>80% saturation	116-130	170-192	825	A,S,M	S	Bringolf et al 2007
Freshwater mussel (*Villosa delumbis*)	96h EC10 (survival of juveniles)	8.32-8.61	20.1-21.9	>80% saturation	116-130	170-192	898	A,S,M	S	Bringolf et al 2007
Freshwater mussel (*Villosa constricta*)	24h EC10 (survival of glochidia)	8.32-8.61	20.1-21.9	>80% saturation	116-130	170-192	789	A,S,M	S	Bringolf et al 2007
Freshwater mussel (*Villosa constricta*)	48h EC10 (survival of glochidia)	8.32-8.61	20.1-21.9	>80% saturation	116-130	170-192	267	A,S,M	S	Bringolf et al 2007
Freshwater mussel (*Elliptio complananta*)	24h EC10 (survival of glochidia)	8.32-8.61	20.1-21.9	>80% saturation	116-130	170-192	406	A,S,M	S	Bringolf et al 2007
Freshwater mussel (*Elliptio complananta*)	48h EC10 (survival of glochidia)	8.32-8.61	20.1-21.9	>80% saturation	116-130	170-192	91	A,S,M	S	Bringolf et al 2007

Wavy-rayed lampmussel (*Lampsilis fasciola*) (change in status endangered to special concern, public comment period ending 7Jan11)	24h EC10 (survival of glochidia)	8.32-8.61	20.1-21.9	>80% saturation	116-130	170-192	24	A,S,M	S	Bringolf et al 2007
Wavy-rayed lampmussel (*Lampsilis fasciola*) (change in status endangered to special concern, public comment period ending 7Jan11)	48h EC10 (survival of glochidia)	8.32-8.61	20.1-21.9	>80% saturation	116-130	170-192	2	A,S,M	S	Bringolf et al 2007
Wavy-rayed lampmussel (*Lampsilis fasciola*) (change in status endangered to special concern, public comment period ending 7Jan11)	96h EC10 (survival of juveniles)	8.32-8.61	20.1-21.9	>80% saturation	116-130	170-192	601	A,S,M	S	Bringolf et al 2007
Wavy-rayed lampmussel (*Lampsilis fasciola*) (change in status endangered to special concern, public comment period ending 7Jan11)	24h EC30 (2008) (survival of glochidia)		21			95-115 (ASTM moderately hard water)	8.6	A,S,M	S	Gillis 2011
Freshwater mussel (*Lampsilis siliquoidea*)	96h EC10 (survival of juveniles)	8.32-8.61	20.1-21.9	>80% saturation	116-130	170-192	1,474	A,S,M	S	Bringolf et al 2007
Freshwater mussel (*Lampsilis siliquoidea*) (collected from Cox Creek in 2008)	24h EC30 (2008) (survival of glochidia)		21			95-115 (ASTM moderately hard water)	35	A,S,M	A,S,M	Gillis 2011
Freshwater mussel (*Lampsilis siliquoidea*) (collected from Cox Creek in 2008???)	24h EC30 (2007) (survival of glochidia)		21			95-115 (ASTM moderately hard water)	117	A,S,M	A,S,M	Gillis 2011
Freshwater mussel (*Lampsilis siliquoidea*) (collected from Cox Creek in 2008???)	24h EC20 (2007) (survival of glochidia)		21			95-115 (ASTM moderately hard water)	20	A,S,M	A,S,M	Gillis 2011

Northern Riffleshell Mussel (*Epioblasma torulosa rangiana)* (glochidia) (COSEWIC endangered, Canadian occurrence in Ontario)	24h EC30 (survival of glochidia)		21			95-115 (ASTM moderately hard water)	161	A,S,M	S	Gillis 2011
Northern Riffleshell Mussel (*Epioblasma torulosa rangiana)* (glochidia) (COSEWIC endangered, Canadian occurrence in Ontario)	24h EC20 (survival of glochidia)		21			95-115 (ASTM moderately hard water)	111	A,S,M	S	Gillis 2011
Northern Riffleshell Mussel (*Epioblasma torulosa rangiana)* (glochidia) (COSEWIC endangered, Canadian occurrence in Ontario)	24h EC10 (survival of glochidia)		21			95-115 (ASTM moderately hard water)	42	A,S,M	S	Gillis 2011
Water flea (*Ceriodaphnia dubia*) (neonates, < 24 hr old)	Reproduction (7-d, NOEC, mean number offspring per female)	7.64-8.32	23-27	7.45-9.27	56-62	58-72	267	C, R, M	S	Harmon et al., 2003
Water flea (*Ceriodaphnia dubia*) (neonates, < 24 hr old)	Reproduction (7-d, LOEC, mean number offspring per female)	7.64-8.32	23-27	7.45-9.27	56-62	58-72	516	C, R, M	S	Harmon et al., 2003
Water flea (*Ceriodaphnia dubia*) (16-24 hrs)	Reproduction (7 d, NOEC)	7.2-7.6		7		40-48	303	C,R,U	S	Aragao and Pereira, 2003
Water flea (*Ceriodaphnia dubia*) (16-24 hrs)	Reproduction (7 d, NOEC)	7.2-7.6		7		40-48	303	C,R,U	S	Aragao and Pereira, 2003
Water flea (*Ceriodaphnia dubia*) (16-24 hrs)	Reproduction (7 d, NOEC)	7.2-7.6		7		40-48	303	C,R,U	S	Aragao and Pereira, 2003
Water flea (*Ceriodaphnia dubia*) (16-24 hrs)	Reproduction (7 d, NOEC)	7.2-7.6		7		40-48	303	C,R,U	S	Aragao and Pereira, 2003
Water flea (*Ceriodaphnia dubia*) (16-24 hrs)	Reproduction (7 d, NOEC)	7.2-7.6		7		40-48	152	C,R,U	S	Aragao and Pereira, 2003
Water flea (*Ceriodaphnia dubia*) (16-24 hrs)	Reproduction (7 d, NOEC)	7.2-7.6		7		40-48	<152	C,R,U	S	Aragao and Pereira, 2003
Water flea (*Ceriodaphnia dubia*) (6-30 hrs)	Reproduction (7 d, NOEC)	7.2-7.6		7		40-48	152	C,R,U	S	Aragao and Pereira, 2003
Water flea (*Ceriodaphnia dubia*) (6-30 hrs)	Reproduction (7 d, NOEC)	7.2-7.6		7		40-48	303	C,R,U	S	Aragao and Pereira, 2003

Water flea (*Ceriodaphnia dubia*) (6-30 hrs)	Reproduction (7 d, NOEC)	7.2-7.6		7		40-48	<152	C,R,U	S	Aragao and Pereira, 2003
Water flea (*Ceriodaphnia dubia*) (6-30 hrs)	Reproduction (7 d, NOEC)	7.2-7.6		7		40-48	152	C,R,U	S	Aragao and Pereira, 2003
Water flea (*Ceriodaphnia dubia*) (6-30 hrs)	Reproduction (7 d, NOEC)	7.2-7.6		7		40-48	152	C,R,U	S	Aragao and Pereira, 2003
Water flea (*Ceriodaphnia dubia*) (6-30 hrs)	Reproduction (7 d, NOEC)	7.2-7.6		7		40-48	303	C,R,U	S	Aragao and Pereira, 2003
Water flea (*Ceriodaphnia dubia*) (6-30 hrs)	Reproduction (7 d, NOEC, mean value calculated from above 12 tests - excluded <152 results from calculation of mean)	7.2-7.6		7		40-48	243	C,R,U	S	Aragao and Pereira, 2003
Water flea (*Ceriodaphnia dubia*) (16-24 hrs)	Reproduction (7 d, LOEC)	7.2-7.6		7		40-48	607	C,R,U	S	Aragao and Pereira, 2003
Water flea (*Ceriodaphnia dubia*) (16-24 hrs)	Reproduction (7 d, LOEC)	7.2-7.6		7		40-48	607	C,R,U	S	Aragao and Pereira, 2003
Water flea (*Ceriodaphnia dubia*) (16-24 hrs)	Reproduction (7 d, LOEC)	7.2-7.6		7		40-48	303	C,R,U	S	Aragao and Pereira, 2003
Water flea (*Ceriodaphnia dubia*) (16-24 hrs)	Reproduction (7 d, LOEC)	7.2-7.6		7		40-48	607	C,R,U	S	Aragao and Pereira, 2003
Water flea (*Ceriodaphnia dubia*) (16-24 hrs)	Reproduction (7 d, LOEC)	7.2-7.6		7		40-48	607	C,R,U	S	Aragao and Pereira, 2003
Water flea (*Ceriodaphnia dubia*) (16-24 hrs)	Reproduction (7 d, LOEC)	7.2-7.6		7		40-48	607	C,R,U	S	Aragao and Pereira, 2003
Water flea (*Ceriodaphnia dubia*) (6-30 hrs)	Reproduction (7 d, LOEC)	7.2-7.6		7		40-48	152??	C,R,U	S	Aragao and Pereira, 2003
Water flea (*Ceriodaphnia dubia*) (6-30 hrs)	Reproduction (7 d, LOEC)	7.2-7.6		7		40-48	303	C,R,U	S	Aragao and Pereira, 2003
Water flea (*Ceriodaphnia dubia*) (6-30 hrs)	Reproduction (7 d, LOEC)	7.2-7.6		7		40-48	303	C,R,U	S	Aragao and Pereira, 2003
Water flea (*Ceriodaphnia dubia*) (6-30 hrs)	Reproduction (7 d, LOEC)	7.2-7.6		7		40-48	152??	C,R,U	S	Aragao and Pereira, 2003
Water flea (*Ceriodaphnia dubia*) (6-30 hrs)	Reproduction (7 d, LOEC)	7.2-7.6		7		40-48	303	C,R,U	S	Aragao and Pereira, 2003

Water flea (*Ceriodaphnia dubia*) (6-30 hrs)	Reproduction (7 d, LOEC)	7.2-7.6		7		40-48	607	C,R,U	S	Aragao and Pereira, 2003
Water flea (*Ceriodaphnia dubia*) (6-30 hrs)	Reproduction (7 d, LOEC, mean value calculated from above 12 tests - excluded same 2 test results from calculation of mean NOEC)	7.2-7.6		7		40-48	485	C,R,U	S	Aragao and Pereira, 2003
Water flea (*Ceriodaphnia dubia*) (16-24 hrs)	Reproduction (7 d, IC50)	7.2-7.6		7		40-48	685	C,R,U	S	Aragao and Pereira, 2003
Water flea (*Ceriodaphnia dubia*) (16-24 hrs)	Reproduction (7 d, IC50)	7.2-7.6		7		40-48	558	C,R,U	S	Aragao and Pereira, 2003
Water flea (*Ceriodaphnia dubia*) (16-24 hrs)	Reproduction (7 d, IC50)	7.2-7.6		7		40-48	667	C,R,U	S	Aragao and Pereira, 2003
Water flea (*Ceriodaphnia dubia*) (16-24 hrs)	Reproduction (7 d, IC50)	7.2-7.6		7		40-48	594	C,R,U	S	Aragao and Pereira, 2003
Water flea (*Ceriodaphnia dubia*) (16-24 hrs)	Reproduction (7 d, IC50)	7.2-7.6		7		40-48	346	C,R,U	S	Aragao and Pereira, 2003
Water flea (*Ceriodaphnia dubia*) (16-24 hrs)	Reproduction (7 d, IC50)	7.2-7.6		7		40-48	370	C,R,U	S	Aragao and Pereira, 2003
Water flea (*Ceriodaphnia dubia*) (6-30 hrs)	Reproduction (7 d, IC50)	7.2-7.6		7		40-48	455	C,R,U	S	Aragao and Pereira, 2003
Water flea (*Ceriodaphnia dubia*) (6-30 hrs)	Reproduction (7 d, IC50)	7.2-7.6		7		40-48	582	C,R,U	S	Aragao and Pereira, 2003
Water flea (*Ceriodaphnia dubia*) (6-30 hrs)	Reproduction (7 d, IC50)	7.2-7.6		7		40-48	431	C,R,U	S	Aragao and Pereira, 2003
Water flea (*Ceriodaphnia dubia*) (6-30 hrs)	Reproduction (7 d, IC50)	7.2-7.6		7		40-48	412	C,R,U	S	Aragao and Pereira, 2003
Water flea (*Ceriodaphnia dubia*) (6-30 hrs)	Reproduction (7 d, IC50)	7.2-7.6		7		40-48	437	C,R,U	S	Aragao and Pereira, 2003
Water flea (*Ceriodaphnia dubia*) (6-30 hrs)	Reproduction (7 d, IC50)	7.2-7.6		7		40-48	406	C,R,U	S	Aragao and Pereira, 2003
Water flea (*Ceriodaphnia dubia*) (0-4, 20-24 & 0-24 hrs)	Survival (7 d, NOEC)	7.0-8.5	25±1		110-120	160-180	1,092	C,R,U	S	Cooney *et al.*, 1992

Water flea (*Ceriodaphnia dubia*) (0-4, 20-24 & 0-24 hrs)	Survival (7 d, LOEC)	7.0-8.5	25±1		110-120	160-180	1,456	C,R,U	S	Cooney et al., 1992
Water flea (*Ceriodaphnia dubia*) (0-4, 20-24 & 0-24 hrs)	Survival (7 d, MATC)	7.0-8.5	25±1		110-120	160-180	1,261	C,R,U	S	Cooney et al., 1992
Water flea (*Ceriodaphnia dubia*) (0-4 hrs)	Reproduction (7 d, NOEC)	7.0-8.5	25±1		110-120	160-180	455	C,R,U	S	Cooney et al., 1992
Water flea (*Ceriodaphnia dubia*) (0-4 hrs)	Reproduction (7 d, NOEC)	7.0-8.5	25±1		110-120	160-180	607	C,R,U	S	Cooney et al., 1992
Water flea (*Ceriodaphnia dubia*) (0-4 hrs)	Reproduction (7 d, NOEC)	7.0-8.5	25±1		110-120	160-180	819	C,R,U	S	Cooney et al., 1992
Water flea (*Ceriodaphnia dubia*) (0-4 hrs)	Reproduction (7 d, NOEC)	7.0-8.5	25±1		110-120	160-180	<455	C,R,U	S	Cooney et al., 1992
Water flea (*Ceriodaphnia dubia*) (0-4 hrs)	Reproduction (7 d, NOEC)	7.0-8.5	25±1		110-120	160-180	<455	C,R,U	S	Cooney et al., 1992
Water flea (*Ceriodaphnia dubia*) (0-4 hrs)	Reproduction (7 d, NOEC)	7.0-8.5	25±1		110-120	160-180	<455	C,R,U	S	Cooney et al., 1992
Water flea (*Ceriodaphnia dubia*) (20-24 hrs)	Reproduction (7 d, NOEC)	7.0-8.5	25±1		110-120	160-180	819	C,R,U	S	Cooney et al., 1992
Water flea (*Ceriodaphnia dubia*) (20-24 hrs)	Reproduction (7 d, NOEC)	7.0-8.5	25±1		110-120	160-180	607	C,R,U	S	Cooney et al., 1992
Water flea (*Ceriodaphnia dubia*) (20-24 hrs)	Reproduction (7 d, NOEC)	7.0-8.5	25±1		110-120	160-180	819	C,R,U	S	Cooney et al., 1992
Water flea (*Ceriodaphnia dubia*) (20-24 hrs)	Reproduction (7 d, NOEC)	7.0-8.5	25±1		110-120	160-180	607	C,R,U	S	Cooney et al., 1992
Water flea (*Ceriodaphnia dubia*) (20-24 hrs)	Reproduction (7 d, NOEC)	7.0-8.5	25±1		110-120	160-180	607	C,R,U	S	Cooney et al., 1992
Water flea (*Ceriodaphnia dubia*) (20-24 hrs)	Reproduction (7 d, NOEC)	7.0-8.5	25±1		110-120	160-180	607	C,R,U	S	Cooney et al., 1992
Water flea (*Ceriodaphnia dubia*) (0-24 hrs)	Reproduction (7 d, NOEC)	7.0-8.5	25±1		110-120	160-180	455	C,R,U	S	Cooney et al., 1992
Water flea (*Ceriodaphnia dubia*) (0-24 hrs)	Reproduction (7 d, NOEC)	7.0-8.5	25±1		110-120	160-180	819	C,R,U	S	Cooney et al., 1992

Water flea (*Ceriodaphnia dubia*) (0-24 hrs)	Reproduction (7 d, NOEC)	7.0-8.5	25±1		110-120	160-180	455	C,R,U	S	Cooney et al., 1992
Water flea (*Ceriodaphnia dubia*) (0-24 hrs)	Reproduction (7 d, NOEC)	7.0-8.5	25±1		110-120	160-180	<455	C,R,U	S	Cooney et al., 1992
Water flea (*Ceriodaphnia dubia*) (0-24 hrs)	Reproduction (7 d, NOEC)	7.0-8.5	25±1		110-120	160-180	455	C,R,U	S	Cooney et al., 1992
Water flea (*Ceriodaphnia dubia*) (0-24 hrs)	Reproduction (7 d, NOEC)	7.0-8.5	25±1		110-120	160-180	455	C,R,U	S	Cooney et al., 1992
Water flea (*Ceriodaphnia dubia*) (0-24 hrs)	Reproduction (7 d, NOEC) (mean of above 18 tests, not including the <values)	7.0-8.5	25±1		110-120	160-180	613	C,R,U	S	Cooney et al., 1992
Water flea (*Ceriodaphnia dubia*) (0-4 hrs)	Reproduction (7 d, LOEC)	7.0-8.5	25±1		110-120	160-180	607	C,R,U	S	Cooney et al., 1992
Water flea (*Ceriodaphnia dubia*) (0-4 hrs)	Reproduction (7 d, LOEC)	7.0-8.5	25±1		110-120	160-180	819	C,R,U	S	Cooney et al., 1992
Water flea (*Ceriodaphnia dubia*) (0-4 hrs)	Reproduction (7 d, LOEC)	7.0-8.5	25±1		110-120	160-180	1,092	C,R,U	S	Cooney et al., 1992
Water flea (*Ceriodaphnia dubia*) (0-4 hrs)	Reproduction (7 d, LOEC)	7.0-8.5	25±1		110-120	160-180	455	C,R,U	S	Cooney et al., 1992
Water flea (*Ceriodaphnia dubia*) (0-4 hrs)	Reproduction (7 d, LOEC)	7.0-8.5	25±1		110-120	160-180	455	C,R,U	S	Cooney et al., 1992
Water flea (*Ceriodaphnia dubia*) (0-4 hrs)	Reproduction (7 d, LOEC)	7.0-8.5	25±1		110-120	160-180	455	C,R,U	S	Cooney et al., 1992
Water flea (*Ceriodaphnia dubia*) (20-24 hrs)	Reproduction (7 d, LOEC)	7.0-8.5	25±1		110-120	160-180	1,092	C,R,U	S	Cooney et al., 1992
Water flea (*Ceriodaphnia dubia*) (20-24 hrs)	Reproduction (7 d, LOEC)	7.0-8.5	25±1		110-120	160-180	819	C,R,U	S	Cooney et al., 1992
Water flea (*Ceriodaphnia dubia*) (20-24 hrs)	Reproduction (7 d, LOEC)	7.0-8.5	25±1		110-120	160-180	1,092	C,R,U	S	Cooney et al., 1992
Water flea (*Ceriodaphnia dubia*) (20-24 hrs)	Reproduction (7 d, LOEC)	7.0-8.5	25±1		110-120	160-180	819	C,R,U	S	Cooney et al., 1992
Water flea (*Ceriodaphnia dubia*) (20-24 hrs)	Reproduction (7 d, LOEC)	7.0-8.5	25±1		110-120	160-180	819	C,R,U	S	Cooney et al., 1992

Water flea (*Ceriodaphnia dubia*) (20-24 hrs)	Reproduction (7 d, LOEC)	7.0-8.5	25±1		110-120	160-180	819	C,R,U	S	Cooney et al., 1992
Water flea (*Ceriodaphnia dubia*) (0-24 hrs)	Reproduction (7 d, LOEC)	7.0-8.5	25±1		110-120	160-180	607	C,R,U	S	Cooney et al., 1992
Water flea (*Ceriodaphnia dubia*) (0-24 hrs)	Reproduction (7 d, LOEC)	7.0-8.5	25±1		110-120	160-180	1,092	C,R,U	S	Cooney et al., 1992
Water flea (*Ceriodaphnia dubia*) (0-24 hrs)	Reproduction (7 d, LOEC)	7.0-8.5	25±1		110-120	160-180	607	C,R,U	S	Cooney et al., 1992
Water flea (*Ceriodaphnia dubia*) (0-24 hrs)	Reproduction (7 d, LOEC)	7.0-8.5	25±1		110-120	160-180	455	C,R,U	S	Cooney et al., 1992
Water flea (*Ceriodaphnia dubia*) (0-24 hrs)	Reproduction (7 d, LOEC)	7.0-8.5	25±1		110-120	160-180	607	C,R,U	S	Cooney et al., 1992
Water flea (*Ceriodaphnia dubia*) (0-24 hrs)	Reproduction (7 d, LOEC)	7.0-8.5	25±1		110-120	160-180	607	C,R,U	S	Cooney et al., 1992
Water flea (*Ceriodaphnia dubia*) (0-24 hrs)	Reproduction (7 d, LOEC) (mean of above 18 tests)	7.0-8.5	25±1		110-120	160-180	740	C,R,U	S	Cooney et al., 1992
Water flea (*Ceriodaphnia dubia*) (<24 hr neonates)	Reproduction (7 d, IC25)	7.4-7.8 (MHRW)	24-26		57-64 (MHRW)	80-100 (MHRW)	454	C,R,M	P	Elphick et al 2011
Water flea (*Ceriodaphnia dubia*) (<24 hr neonates)	Reproduction (7 d, IC50)	7.4-7.8 (MHRW)	24-26		57-64 (MHRW)	80-100 (MHRW)	697	C,R,M	P	Elphick et al 2011
Water flea (*Ceriodaphnia dubia*) (<24 hr neonates)	7d IC25 (Reproduction)	6.8	24-26			10	117	C,R,M	P	Elphick et al 2011
Water flea (*Ceriodaphnia dubia*) (<24 hr neonates)	7d IC50 (Reproduction)	6.8	24-26			10	161	C,R,M	P	Elphick et al 2011
Water flea (*Ceriodaphnia dubia*) (<24 hr neonates)	7d LC50 (Survival)	6.8	24-26			10	132	C,R,M	P	Elphick et al 2011
Water flea (*Ceriodaphnia dubia*) (<24 hr neonates)	7d IC25 (Reproduction)	7	24-26			20	264	C,R,M	P	Elphick et al 2011
Water flea (*Ceriodaphnia dubia*) (<24 hr neonates)	7d IC50 (Reproduction)	7	24-26			20	301	C,R,M	P	Elphick et al 2011
Water flea (*Ceriodaphnia dubia*) (<24 hr neonates)	7d LC50 (Survival)	7	24-26			20	316	C,R,M	P	Elphick et al 2011

Water flea (*Ceriodaphnia dubia*) (<24 hr neonates)	7d IC25 (Reproduction)	7.2	24-26			40	146	C,R,M	P	Elphick et al 2011
Water flea (*Ceriodaphnia dubia*) (<24 hr neonates)	7d IC50 (Reproduction)	7.2	24-26			40	481	C,R,M	P	Elphick et al 2011
Water flea (*Ceriodaphnia dubia*) (<24 hr neonates)	7d LC50 (Survival)	7.2	24-26			40	540	C,R,M	P	Elphick et al 2011
Water flea (*Ceriodaphnia dubia*) (<24 hr neonates)	7d IC25 (Reproduction)	7.8	24-26			80	454	C,R,M	P	Elphick et al 2011
Water flea (*Ceriodaphnia dubia*) (<24 hr neonates)	7d IC50 (Reproduction)	7.8	24-26			80	697	C,R,M	P	Elphick et al 2011
Water flea (*Ceriodaphnia dubia*) (<24 hr neonates)	7d LC50 (Survival)	7.8	24-26			80	1,134	C,R,M	P	Elphick et al 2011
Water flea (*Ceriodaphnia dubia*) (<24 hr neonates)	7d IC25 (Reproduction)	8.2	24-26			160	580	C,R,M	P	Elphick et al 2011
Water flea (*Ceriodaphnia dubia*) (<24 hr neonates)	7d IC50 (Reproduction)	8.2	24-26			160	895	C,R,M	P	Elphick et al 2011
Water flea (*Ceriodaphnia dubia*) (<24 hr neonates)	7d LC50 (Survival)	8.2	24-26			160	1,240	C,R,M	P	Elphick et al 2011
Water flea (*Ceriodaphnia dubia*) (<24 hr neonates)	7d IC25 (Reproduction)	8.3	24-26			320	521	C,R,M	P	Elphick et al 2011
Water flea (*Ceriodaphnia dubia*) (<24 hr neonates)	7d IC50 (Reproduction)	8.3	24-26			320	700	C,R,M	P	Elphick et al 2011
Water flea (*Ceriodaphnia dubia*) (<24 hr neonates)	7d LC50 (Survival)	8.3	24-26			320	1,303	C,R,M	P	Elphick et al 2011
Water flea (*Ceriodaphnia dubia*) (neonates, < 24 hr old)	Reduced Reproduction (7 d, 12.8% decrease in reproduction compared to controls)	8.1	25		69	100	342	C,R,U	S	Lasier et al., 2006
Water flea (*Ceriodaphnia dubia*) (neonates, < 24 hr old)	Reduced Reproduction (7 d, 21.9% decrease in reproduction compared to controls)	8.2	25		99	45	342	C,R,U	S	Lasier et al., 2006
Water flea (*Ceriodaphnia dubia*) (neonates, < 24 hr old)	Reduced Reproduction (7 d, 34.8% decrease in reproduction compared to controls)	8	25		44	46	342	C,R,U	S	Lasier et al., 2006
Water flea (*Ceriodaphnia dubia*) (neonates, < 24 hr old)	Reduced Reproduction (7 d, 17.1% decrease in reproduction compared to controls)	8.3	25		96	99	342	C,R,U	S	Lasier et al., 2006

Water flea (*Ceriodaphnia dubia*) (neonates, < 24 hr old)	Reduced Reproduction (7 d, 32.9% decrease in reproduction compared to controls)	8.1	25		69	100	565	C,R,U	S	Lasier et al., 2006
Water flea (*Ceriodaphnia dubia*) (neonates, < 24 hr old)	Reduced Reproduction (7 d, 53.5% decrease in reproduction compared to controls)	8.2	25		99	45	565	C,R,U	S	Lasier et al., 2006
Water flea (*Ceriodaphnia dubia*) (neonates, < 24 hr old)	Reduced Reproduction (7 d, 58.5% decrease in reproduction compared to controls)	8	25		44	46	565	C,R,U	S	Lasier et al., 2006
Water flea (*Ceriodaphnia dubia*) (neonates, < 24 hr old)	Reduced Reproduction (7 d, 43.9% decrease in reproduction compared to controls)	8.3	25		96	99	565	C,R,U	S	Lasier et al., 2006
Water flea *(Ceriodaphnia dubia)* (<24h old)	Reproduction (IC50, 7-d)					dilute mineral water	394	C,R	P	Degreave et al. 1992
Water flea *(Ceriodaphnia dubia)* (<24h old)	Reproduction (IC50, 7-d)					dilute mineral water	437	C,R	P	Degreave et al. 1992
Water flea *(Ceriodaphnia dubia)* (<24h old)	Reproduction (IC50, 7-d)					dilute mineral water	443	C,R	P	Degreave et al. 1992
Water flea *(Ceriodaphnia dubia)* (<24h old)	Reproduction (IC50, 7-d)					dilute mineral water	449	C,R	P	Degreave et al. 1992
Water flea *(Ceriodaphnia dubia)* (<24h old)	Reproduction (IC50, 7-d)					dilute mineral water	182	C,R	P	Degreave et al. 1992
Water flea *(Ceriodaphnia dubia)* (<24h old)	Reproduction (IC50, 7-d)					dilute mineral water	843	C,R	P	Degreave et al. 1992
Water flea *(Ceriodaphnia dubia)* (<24h old)	Reproduction (IC50, 7-d)					dilute mineral water	783	C,R	P	Degreave et al. 1992
Water flea *(Ceriodaphnia dubia)* (<24h old)	Reproduction (IC50, 7-d)					dilute mineral water	813	C,R	P	Degreave et al. 1992
Water flea *(Ceriodaphnia dubia)* (<24h old)	Reproduction (IC50, 7-d)					dilute mineral water	916	C,R	P	Degreave et al. 1992
Water flea *(Ceriodaphnia dubia)* (<24h old)	Reproduction (IC50, 7-d)					dilute mineral water	971	C,R	P	Degreave et al. 1992
Water flea *(Ceriodaphnia dubia)* (<24h old)	Reproduction (IC50, 7-d)					dilute mineral water	1,025	C,R	P	Degreave et al. 1992
Water flea *(Ceriodaphnia dubia)* (<24h old)	Reproduction (IC50, 7-d)					dilute mineral water	1,068	C,R	P	Degreave et al. 1992

Water flea *(Ceriodaphnia dubia)* (<24h old)	Reproduction (IC50, 7-d)					dilute mineral water	1,153	C,R	P	Degreave et al. 1992
Water flea *(Ceriodaphnia dubia)* (<24h old)	Reproduction 7d IC50 (mean group 1)					dilute mineral water	813	C,R	P	Degreave et al. 1992
Water flea *(Ceriodaphnia dubia)* (<24h old)	Reproduction 7d IC50 (mean group 3)					dilute mineral water	582	C,R	P	Degreave et al. 1992
Water flea (*Ceriodaphnia dubia*) (<12h old)	Reproduction (EC50, 9-d, mean brood size)	8.2±0.2	23-27	8.0±1.5	55-75	90-110 (L Huron water)	1,068	C	S	Cowgill and Milazzo, 1990
Water flea (*Ceriodaphnia dubia*) (<12h old)	Reproduction (Total Progeny, EC50, 9-d)	8.2±0.2	23-27	8.0±1.5	55-75	90-110 (L Huron water)	1,088	C	S	Cowgill and Milazzo, 1990
Water flea (*Ceriodaphnia dubia*) (<12h old)	Reproduction (Mean number of broods, 9-d, EC50)	8.2±0.2	23-27	8.0±1.5	55-75	90-110 (L Huron water)	1,208	C	S	Cowgill and Milazzo, 1990
Water flea (*Ceriodaphnia dubia*) (<12h old)	Reproduction (NOEC, 9-d, mean brood size)	8.2±0.2	23-27	8.0±1.5	55-75	90-110 (L Huron water)	786	C	S	Cowgill and Milazzo, 1990
Water flea (*Ceriodaphnia dubia*) (<12h old)	Reproduction (NOEC, 9-d, mean number of broods)	8.2±0.2	23-27	8.0±1.5	55-75	90-110 (L Huron water)	786	C	S	Cowgill and Milazzo, 1990
Water flea (*Ceriodaphnia dubia*) (<12h old)	Reproduction (NOEC, 9-d, total progeny)	8.2±0.2	23-27	8.0±1.5	55-75	90-110 (L Huron water)	786	C	S	Cowgill and Milazzo, 1990
Water flea (*Ceriodaphnia dubia*) (neonates, < 24 hr old)	Mortality and reproduction (7-d, median EC50)	7.64-8.32	23-27	7.45-9.27	56-76	54-72	819	C, R, M	S	Harmon et al., 2003
Water flea (*Ceriodaphnia dubia*) (neonates, < 24 hr old)	Mortality (7-d, NOEC)	7.64-8.32	23-27	7.45-9.27	56-76	54-72	1,031	C, R, M	S	Harmon et al., 2003
Water flea (*Ceriodaphnia dubia*) (neonates, < 24 hr old)	Mortality (7-d, LOEC)	7.64-8.32	23-27	7.45-9.27	56-76	54-72	1,335	C, R, M	S	Harmon et al., 2003
Water flea *(Ceriodaphnia dubia)* (<24h old)	Mortality (NOEC, 7-d)					dilute mineral water	910	C,R	P	Degreave et al. 1992
Water flea *(Ceriodaphnia dubia)* (<24h old)	Mortality (LC50, 7-d)					dilute mineral water	170	C,R	P	Degreave et al. 1992
Water flea *(Ceriodaphnia dubia)* (<24h old)	Mortality (LC50, 7-d)					dilute mineral water	552	C,R	P	Degreave et al. 1992
Water flea *(Ceriodaphnia dubia)* (<24h old)	Mortality (LC50, 7-d)					dilute mineral water	710	C,R	P	Degreave et al. 1992

Water flea *(Ceriodaphnia dubia)* (<24h old)	Mortality (LC50, 7-d)					dilute mineral water	867	C,R	P	Degreave et al. 1992
Water flea *(Ceriodaphnia dubia)* (<24h old)	Mortality (LC50, 7-d)					dilute mineral water	995	C,R	P	Degreave et al. 1992
Water flea *(Ceriodaphnia dubia)* (<24h old)	Mortality (LC50, 7-d)					dilute mineral water	1,037	C,R	P	Degreave et al. 1992
Water flea *(Ceriodaphnia dubia)* (<24h old)	Mortality (LC50, 7-d)					dilute mineral water	1,055	C,R	P	Degreave et al. 1992
Water flea *(Ceriodaphnia dubia)* (<24h old)	7d LC50 (mean)					dilute mineral water	1,074	C,R	P	Degreave et al. 1992
Water flea *(Ceriodaphnia dubia)* (<24h old)	7d LC50 (mean)					dilute mineral water	808	C,R	P	Degreave et al. 1992
Water flea *(Ceriodaphnia dubia)* (<24h old)	7d LC50	8.2±0.2	23-27	8.0±1.5	55-75	90-110 (L Huron water)	1,088	C	S	Cowgill and Milazzo, 1990
Water flea *(Ceriodaphnia dubia)* (<24h old)	7d NOEC (survival)						182	C	S (US EPA ref tox test data)	Diamond et al (1992)
Water flea *(Ceriodaphnia dubia)* (<24h old)	7d LOEC (survival)						455	C	S (US EPA ref tox test data)	Diamond et al (1992)
Water flea *(Ceriodaphnia dubia)* (<24h old)	7d MATC (survival)						288	C	S (US EPA ref tox test data)	Diamond et al (1992)
Water flea *(Ceriodaphnia dubia)* (<24h old)	7d NOEC (reporoduction)						121	C	S (US EPA ref tox test data)	Diamond et al (1992)
Water flea *(Ceriodaphnia dubia)* (<24h old)	7d LOEC (reporoduction)						455	C	S (US EPA ref tox test data)	Diamond et al (1992)
Water flea *(Ceriodaphnia dubia)* (<24h old)	7d MATC (reporoduction)						235	C	S (US EPA ref tox test data)	Diamond et al (1992)
Water flea (*Daphnia magna*) (<24 hr neonate)	Reproduction (21 d, EC25)	7.4-8.1	19-21	7.6-8.8	60	80-100	421	C,R,M	P	Elphick et al 2011
Water flea (*Daphnia magna*) (<24 hr neonate)	Reproduction (21 d, NOEC)	7.4-8.1	19-21	7.6-8.8	60	80-100	<506	C,R,M	P	Elphick et al 2011
Water flea (*Daphnia magna*) (<24 hr neonate)	Reproduction (21 d, LOEC)	7.4-8.1	19-21	7.6-8.8	60	80-100	506	C,R,M	P	Elphick et al 2011

Water flea (*Daphnia magna*) (<24 hr neonate)	Reproduction (21 d, EC50)	7.4-8.1	19-21	7.6-8.8	60	80-100	1,037	C,R,M	P	Elphick et al 2011
Water flea (*Daphnia magna*) (<24 hr neonate)	Mortality (21 d, NOEC)	7.4-8.1	19-21	7.6-8.8	60	80-100	1,980	C,R,M	P	Rescan Environmental Services Ltd., 2007
Water flea (*Daphnia magna*) (<24 hr neonate)	Mortality (21 d, LOEC)	7.4-8.1	19-21	7.6-8.8	60	80-100	4,070	C,R,M	P	Rescan Environmental Services Ltd., 2007
Water flea (*Daphnia magna*) (<24 hr neonate)	Mortality (21 d, LC50)	7.4-8.1	19-21	7.6-8.8	60	80-100	2,311	C,R,M	P	Rescan Environmental Services Ltd., 2007
Water flea *(Daphnia magna)*	50% Mortality (7-d)	7.9-8.2	23-27	8.1-8.8	54-58	166-172	3,504	C	S	Cowgill and Milazzo, 1990
Water flea *(Daphnia magna)*	Reproduction (NOEC, 10-d, mean brood size)	7.9-8.2	23-27	8.1-8.8	54-58	166-172	786	C	S	Cowgill and Milazzo, 1990
Water flea *(Daphnia magna)*	Reproduction (NOEC, 10-d, mean number of broods)	7.9-8.2	23-27	8.1-8.8	54-58	166-172	786	C	S	Cowgill and Milazzo, 1990
Water flea *(Daphnia magna)*	Reproduction (Total progeny, 10-d, NOEC)	7.9-8.2	23-27	8.1-8.8	54-58	166-172	2,184	C	S	Cowgill and Milazzo, 1990
Water flea *(Daphnia magna)*	Growth (Dry weight, 10-d, NOEC)	7.9-8.2	23-27	8.1-8.8	54-58	166-172	786	C	S	Cowgill and Milazzo, 1990
Water flea (*Daphnia magna)*	Reproductive impairment (21-d)	7.74	18±1	9	42.3	45.3	1,035	C,R,M	S	Biesinger and Christensen 1972
Water flea *(Daphnia magna)*	Reproduction (Mean brood size, 10-d, EC50)	7.9-8.2	23-27	8.1-8.8	54-58	166-172	2,451	C	S	Cowgill and Milazzo, 1990
Water flea *(Daphnia magna)*	Reproduction (Total progeny, 10-d, EC50)	7.9-8.2	23-27	8.1-8.8	54-58	166-172	2,597	C	S	Cowgill and Milazzo, 1990
Water flea *(Daphnia magna)*	Growth (Dry weight, 10-d, EC50)	7.9-8.2	23-27	8.1-8.8	54-58	166-172	2,614	C	S	Cowgill and Milazzo, 1990
Water flea *(Daphnia magna)*	Reproduction (Mean number of broods, 10-d, EC50)	7.9-8.2	23-27	8.1-8.8	54-58	166-172	3,504	C	S	Cowgill and Milazzo, 1990
Water flea (*Daphnia ambigua*) (neonates, < 24 hr old)	Mortality (10-d, NOEC)	7.64-8.32	19-23	7.45-9.27	56-76	54-72	267	C, R, M	S	Harmon et al., 2003
Water flea (*Daphnia ambigua*) (neonates, < 24 hr old)	Mortality (10-d, LOEC)	7.64-8.32	19-23	7.45-9.27	56-76	54-72	516	C, R, M	S	Harmon et al., 2003
Water flea (*Daphnia ambigua*) (neonates, < 24 hr old)	Mortality (10-d, MATC)	7.64-8.32	19-23	7.45-9.27	56-76	54-72	371	C, R, M	S	Harmon et al., 2003

Water flea (*Daphnia ambigua*) (neonates, < 24 hr old)	Mortality and reproduction (10-d median EC10)	7.64-8.32	19-23	7.45-9.27	56-76	54-72	259	C, R, M	S	Harmon et al., 2003
Water flea (*Daphnia ambigua*) (neonates, < 24 hr old)	Mortality and reproduction (10-d median EC50)	7.64-8.32	19-23	7.45-9.27	56-76	54-72	394	C, R, M	S	Harmon et al., 2003
Water flea (*Daphnia ambigua*) (neonates, < 24 hr old)	Reproduction (10-d, NOEC, mean number offspring per female)	7.64-8.32	19-23	7.45-9.27	56-76	54-72	267	C, R, M	S	Harmon et al., 2003
Water flea (*Daphnia ambigua*) (neonates, < 24 hr old)	Reproduction (10-d, LOEC, mean number offspring per female)	7.64-8.32	19-23	7.45-9.27	56-76	54-72	516	C, R, M	S	Harmon et al., 2003
Water flea (*Daphnia pulex*)	21d NOEC (Reproduction)	7.94±0.24	20±0.1	8.8±0.3	58.8±10.5	96.3±9.9 (ASTM recon water)	314	C,S,M	S	Birge et al. 1985
Water flea (*Daphnia pulex*)	21d LOEC (Reproduction)	7.94±0.24	20±0.1	8.8±0.3	58.8±10.5	96.3±9.9 (ASTM recon water)	441	C,S,M	S	Birge et al. 1985
Water flea (*Daphnia pulex*)	21d MATC (Reproduction)	7.94±0.24	20±0.1	8.8±0.3	58.8±10.5	96.3±9.9 (ASTM recon water)	372	C,S,M	S	Birge et al. 1985
Water flea (*Daphnia pulex*)	21d IC10 (Reproduction)	7.94±0.24	20±0.1	8.8±0.3	58.8±10.5	96.3±9.9 (ASTM recon water)	368	C,S,M	S	Birge et al. 1985 (Point estimates were calculated by Elphick et al 2011 using linear interpolation based on original data from Birge et al 1985)
Water flea (*Daphnia pulex*)	21d NOEC (Growth)	7.94±0.24	20±0.1	8.8±0.3	58.8±10.5	96.3±9.9 (ASTM recon water)	314	C,S,M	S	Birge et al. 1985
Water flea (*Daphnia pulex*)	21d LOEC (Growth)	7.94±0.24	20±0.1	8.8±0.3	58.8±10.5	96.3±9.9 (ASTM recon water)	441	C,S,M	S	Birge et al. 1985
Water flea (*Daphnia pulex*)	21d MATC (Growth)	7.94±0.24	20±0.1	8.8±0.3	58.8±10.5	96.3±9.9 (ASTM recon water)	372	C,S,M	S	Birge et al. 1985
Aquatic sowbug *(Asellus communis)*	7d LC50						3,731	C	U (control survival not reported)	Wurtz and Bridges 1961
Amphipod (*Gammarus pseudopinmaeus*)	60d NOEC (Survival)		7			spring water	1,000	C	S	Williams et al 1999
Amphipod (*Gammarus pseudopinmaeus*)	60d NOEC (Survival)		7			spring water	2,000	C	S (highest concentration tested produced no effect_see CCME 2007 protocol for direction)	Williams et al 1999

Amphipod (*Gammarus pseudopinmaeus*)	60d NOEC (Reproduction) (reproduction in control group)		7			spring water	10	C	S	Williams et al 1999
Amphipod (*Gammarus pseudopinmaeus*)	60d LOEC (Reproduction) (no reproduction in 2 test concs of 1,000 and 2,000 mg Cl/L)		7			spring water	1,000	C	S	Williams et al 1999
Amphipod (*Gammarus pseudopinmaeus*)	60d MATC (Reproduction)		7			spring water	100	C	S	Williams et al 1999
Snail (*Physa* sp.)	60d NOEC (Survival)		7			spring water	1,000	C	S	Williams et al 1999
Snail (*Physa* sp.)	60d NOEC (Survival)		7			spring water	2,000	C	S (highest concentration tested produced no effect_see CCME 2007 protocol for direction)	Williams et al 1999
Caddisfly (*Hydropsyche betteni*)	survival and pupate (10-d)						800	C	?	Kersey 1981 (In Evans and Frick 2001)
Caddisfly *(Hydropsyche betteni)*	80% Mortality (6-d)						5,999	C	?	Kersey 1981 (In Evans and Frick 2001)
Caddisfly *(Hydropsyche bronta*)	survival and pupate (10-d)						800	C	?	Kersey 1981 (In Evans and Frick 2001)
Caddisfly *(Hydropsyche slossonae*)	survival and pupate (10-d)						800	C	?	Kersey 1981 (In Evans and Frick 2001)
Chironomid (*Chironomus tentans*) (<u><</u>24 hr post-hatch)	Growth (20 d, NOEC, mean AF biomass)	7.6-8.1	22-24	7.2-8.4	60	80-100	2,133	C,R,M	P (sand + peat used as substrate)	Elphick et al 2011
Chironomid (*Chironomus tentans*) (<u><</u>24 hr post-hatch)	Growth (20 d, LOEC, mean AF weight)	7.6-8.1	22-24	7.2-8.4	60	80-100	>2,133	C,R,M	P (sand + peat used as substrate)	Rescan Environmental Services Ltd., 2007
Chironomid (*Chironomus tentans*) (<u><</u>24 hr post-hatch)	Growth (20 d, LOEC, mean AF biomass)	7.6-8.1	22-24	7.2-8.4	60	80-100	3,960	C,R,M	P (sand + peat used as substrate)	Elphick et al 2011
Chironomid (*Chironomus tentans*) (<u><</u>24 hr post-hatch)	Survival (20 d, NOEC)	7.6-8.1	22-24	7.2-8.4	60	80-100	2,133	C,R,M	P (sand + peat used as substrate)	Rescan Environmental Services Ltd., 2007
Chironomid (*Chironomus tentans*) (<u><</u>24 hr post-hatch)	Survival (20 d, LOEC)	7.6-8.1	22-24	7.2-8.4	60	80-100	3,960	C,R,M	P (sand + peat used as substrate)	Rescan Environmental Services Ltd., 2007

Chironomid (*Chironomus tentans*) (≤24 hr post-hatch)	Growth (20 d, IC10, mean AF biomass)	7.6-8.1	22-24	7.2-8.4	60	80-100	2,316	C,R,M	P (sand + peat used as substrate)	Elphick et al 2011
Chironomid (*Chironomus tentans*) (≤24 hr post-hatch)	Growth (20 d, IC25, mean AF biomass)	7.6-8.1	22-24	7.2-8.4	60	80-100	2,590	C,R,M	P (sand + peat used as substrate)	Elphick et al 2011
Chironomid (*Chironomus tentans*) (≤24 hr post-hatch)	Growth (20 d, IC50, mean AF biomass)	7.6-8.1	22-24	7.2-8.4	60	80-100	3,047	C,R,M	P (sand + peat used as substrate)	Elphick et al 2011
Chironomid (*Chironomus tentans*) (≤24 hr post-hatch)	Survival (20d LC50)	7.6-8.1	22-24	7.2-8.4	60	80-100	2,812	C,R,M	P (sand + peat used as substrate)	Rescan Environmental Services Ltd., 2007
Fingernail clam *(Musculium securis)* (newborn)	60-80d NOEC reduced natality (mean number newborns per number of parents, 60-80-d)						0 (control)	C	S	Mackie 1978
Fingernail clam (Musculium securis) (newborn)	60-80d LOEC reduced natality (mean number newborns per number of parents, 60-80-d)						121 (first highest test concentration)	C	S	Mackie 1978
Flatly coiled gyraulus *(Gyraulus circumstriatus)*	10d LC50						1,941	C	U (control survival not reported)	Wurtz and Bridges 1961
Mayfly *(Stenonema modestum)*	Development (NOEC, 14-d)		12±1				1,213	C,R,M	S	Diamond et al. 1992
Mayfly *(Stenonema modestum)*	Development (NOEC, 14-d)		12±1				2,426	C,R,M	S	Diamond et al. 1992
Mayfly *(Stenonema modestum)*	Development (LOEC, 14-d)		12±1				1,638	C,R,M	S	Diamond et al. 1992
Mayfly *(Stenonema modestum)*	Development (LOEC, 14-d)		12±1				3,640	C,R,M	S	Diamond et al. 1992
Mayfly *(Stenonema modestum)*	Development (MATC, 14-d)		12±1				2,047	C,R,M	S	Diamond et al. 1992
Mayfly *(Stenonema modestum)*	Growth (NOEC, 14-d)		12±1				1,638	C,R,M	S	Diamond et al. 1992
Mayfly *(Stenonema modestum)*	Growth (NOEC, 14-d)		12±1				2,426	C,R,M	S	Diamond et al. 1992
Mayfly *(Stenonema modestum)*	Growth (LOEC, 14-d)		12±1				2,123	C,R,M	S	Diamond et al. 1992
Mayfly *(Stenonema modestum)*	Growth (LOEC, 14-d)		12±1				4,246	C,R,M	S	Diamond et al. 1992

Mayfly *(Stenonema modestum)*	Growth (MATC, 14-d)		12±1				2,446	C,R,M	S	Diamond et al. 1992
Mayfly *(Stenonema modestum)*	Mortality (NOEC, 14-d)		12±1				1,638	C,R,M	S	Diamond et al. 1992
Mayfly *(Stenonema modestum)*	Mortality (NOEC, 14-d)		12±1				3,397	C,R,M	S	Diamond et al. 1992
Mayfly *(Stenonema modestum)*	Mortality (LOEC, 14-d)		12±1				4,246	C,R,M	S	Diamond et al. 1992
Mayfly *(Stenonema modestum)*	Mortality (LOEC, 14-d)		12±1				2,123	C,R,M	S	Diamond et al. 1992
Mayfly *(Stenonema modestum)*	Mortality (MATC, 14-d)		12±1				2,661	C,R,M	S	Diamond et al. 1992
Mayfly *(Stenonema modestum)*	Development (NOEC, 7-d)		12±1				2,426	C,R,M	S	Diamond et al. 1992
Mayfly *(Stenonema modestum)*	Development (NOEC, 7-d)		12±1				2,426	C,R,M	S	Diamond et al. 1992
Mayfly *(Stenonema modestum)*	Development (LOEC, 7-d)		12±1				3,640	C,R,M	S	Diamond et al. 1992
Mayfly *(Stenonema modestum)*	Development (LOEC, 7-d)		12±1				4,246	C,R,M	S	Diamond et al. 1992
Mayfly *(Stenonema modestum)*	Development (LOEC, 7-d)		12±1				3,088	C,R,M	S	Diamond et al. 1992
Oligochaete (*Lumbriculus variegatus*) (adult)	Reproduction (28 d, NOEC)	7.3-7.8	22-24	5.1-8.2	60	80-100	<366	C,R,M	P (sand + peat used as substrate)	Elphick et al 2011
Oligochaete (*Lumbriculus variegatus*) (adult)	Reproduction (28 d, LOEC)	7.3-7.8	22-24	5.1-8.2	60	80-100	366	C,R,M	P (sand + peat used as substrate)	Elphick et al 2011
Oligochaete (*Lumbriculus variegatus*) (adult)	Reproduction (28 d, EC25)	7.3-7.8	22-24	5.1-8.2	60	80-100	825	C,R,M	P (sand + peat used as substrate)	Elphick et al 2011
Oligochaete (*Lumbriculus variegatus*) (adult)	Reproduction (28 d, EC50)	7.3-7.8	22-24	5.1-8.2	60	80-100	1,366	C,R,M	P (sand + peat used as substrate)	Elphick et al 2011
Oligochaete (*Tubifex tubifex*) (adult)	Reproduction (28 d, NOEC, number of young produced)	7.2-7.8	22-24	5.5-7.5	60	80-100	462	C,R,M	P (sand + peat used as substrate)	Elphick et al 2011

Oligochaete (*Tubifex tubifex*) (adult)	Reproduction (28 d, LOEC, number of young produced)	7.2-7.8	22-24	5.5-7.5	60	80-100	964	C,R,M	P (sand + peat used as substrate)	Elphick et al 2011
Oligochaete (*Tubifex tubifex*) (adult)	Reproduction (28 d, IC50, number of young produced)	7.2-7.8	22-24	5.5-7.5	60	80-100	752	C,R,M	P (sand + peat used as substrate)	Elphick et al 2011
Oligochaete (*Tubifex tubifex*) (adult)	Reproduction (28 d, IC25, number of young produced)	7.2-7.8	22-24	5.5-7.5	60	80-100	606	C,R,M	P (sand + peat used as substrate)	Elphick et al 2011
Oligochaete (*Tubifex tubifex*) (adult)	Reproduction (28 d, IC10, number of young produced)	7.2-7.8	22-24	5.5-7.5	60	80-100	519	C,R,M	P (sand + peat used as substrate)	Elphick et al 2011
Oligochaete (*Tubifex tubifex*) (adult)	Cocoon Formation (28 d, EC25)	7.2-7.8	22-24	5.5-7.5	60	80-100	620	C,R,M	P (sand + peat used as substrate)	Rescan Environmental Services Ltd., 2007
Oligochaete (*Tubifex tubifex*) (adult)	Cocoon Formation (28 d, EC50)	7.2-7.8	22-24	5.5-7.5	60	80-100	809	C,R,M	P (sand + peat used as substrate)	Rescan Environmental Services Ltd., 2007
Oligochaete (*Tubifex tubifex*) (adult)	Cocoon Formation (28 d, NOEC)	7.2-7.8	22-24	5.5-7.5	60	80-100	964	C,R,M	P (sand + peat used as substrate)	Rescan Environmental Services Ltd., 2007
Oligochaete (*Tubifex tubifex*) (adult)	Cocoon Formation (28 d, LOEC)	7.2-7.8	22-24	5.5-7.5	60	80-100	2,138	C,R,M	P (sand + peat used as substrate)	Rescan Environmental Services Ltd., 2007
Oligochaete (*Tubifex tubifex*) (adult)	Survival (28 d, NOEC)	7.2-7.8	22-24	5.5-7.5	60	80-100	2,138	C,R,M	P (sand + peat used as substrate)	Rescan Environmental Services Ltd., 2007
Oligochaete (*Tubifex tubifex*) (adult)	Survival (28 d, LOEC)	7.2-7.8	22-24	5.5-7.5	60	80-100	4,065	C,R,M	P (sand + peat used as substrate)	Rescan Environmental Services Ltd., 2007
Oligochaete (*Tubifex tubifex*) (adult)	Survival (28 d, EC25)	7.2-7.8	22-24	5.5-7.5	60	80-100	2,167	C,R,M	P (sand + peat used as substrate)	Rescan Environmental Services Ltd., 2007
Oligochaete (*Tubifex tubifex*) (adult)	Survival (28 d, EC50)	7.2-7.8	22-24	5.5-7.5	60	80-100	3,597	C,R,M	P (sand + peat used as substrate)	Rescan Environmental Services Ltd., 2007
Oligochaete (*Tubifex tubifex*) (adult)	Survival (28 d, EC50)	7.2-7.8	22-24	5.5-7.5	60	80-100	4,460	C,R,M	P (sand + peat used as substrate)	Rescan Environmental Services Ltd., 2007
Ramshorn snail *(Helisoma campanulatum)*	10d LC50						3,731	C	U (control survival not reported)	Wurtz and Bridges 1961
Red leech *(Erpobdella punctata)*	4d LC50						4,550	C	U (control survival not reported)	Wurtz and Bridges 1961

Red leech *(Erpobdella punctata)*	10d LC50						4,550	C	U (control survival not reported)	Wurtz and Bridges 1961
Rotifer (*Brachionus calyciflorus)* (mixed: young and non-egg bearing adults)	Rate of population increase (14-d, NOEC)	7.2-7.5	23-27				1,213	C, R, U	S	Peredo-Alvarez et al., 2003
Rotifer (*Brachionus calyciflorus)* (mixed: young and non-egg bearing adults)	Negatively affected population density (14-d)	7.2-7.5	23-27				1,820	C, R, U	S	Peredo-Alvarez et al., 2003
Rotifer (*Brachionus calyciflorus*) (<4 hr old)	Reproduction (48 hr, NOEC)	7.88-8.16	24-25	7.8-8.4	60	76	1,120	A,S,M	P	Elphick et al 2011
Rotifer (*Brachionus calyciflorus*) (<4 hr old)	Reproduction (48 hr, LOEC)	7.88-8.16	24-25	7.8-8.4	60	76	2,330	A,S,M	P	Elphick et al 2011
Rotifer (*Brachionus calyciflorus*) (<4 hr old)	Reproduction (48 hr, IC10)	7.88-8.16	24-25	7.8-8.4	60	76	1,241	A,S,M	P	Elphick et al 2011
Rotifer (*Brachionus calyciflorus*) (<4 hr old)	Reproduction (48 hr, IC25)	7.88-8.16	24-25	7.8-8.4	60	76	1,505	A,S,M	P	Elphick et al 2011
Rotifer (*Brachionus calyciflorus*) (<4 hr old)	Reproduction (48 hr, IC50)	7.88-8.16	24-25	7.8-8.4	60	76	1,945	A,S,M	P	Elphick et al 2011
Rotifer (*Brachionus patulus)* (mixed: young and non-egg bearing adults)	Negatively affected peak population density (20-d)	7.2-7.5	23-27				1,213	C, R, U	S	Peredo-Alvarez et al., 2003
Rotifer (*Brachionus patulus)* (mixed: young and non-egg bearing adults)	Negatively affected rate of population increase (20-d)	7.2-7.5	23-27				1,213	C, R, U	S	Peredo-Alvarez et al., 2003
Rotifer (*Brachionus patulus)* (mixed: young and non-egg bearing adults)	Negatively affected day of maximum population density (20-d)	7.2-7.5	23-27				1,213	C, S, U	S	Peredo-Alvarez et al., 2003
Amphibpod (*Hyalella azteca*) (7-8 d)	Growth (28d, NOEC, mean dry weight)	7.5-8.1	22-24	5.6-9.0	60	80-100 (MHSW)	2,210	C,R,M	U (control survival was 62.5% & conducted using sediment & peat moss as substrate)	Elphick et al 2011

Amphipod (*Hyalella azteca*) (7-8 d)	Growth (28d, LOEC, mean dry weight)	7.5-8.1	22-24	5.6-9.0	60	80-100 (MHSW)	4,237	C,R,M	U (control survival was 62.5% & conducted using sediment & peat moss as substrate)	Elphick et al 2011
Amphipod (*Hyalella azteca*) (7-8 d)	Growth (28d, IC25, mean dry weight)	7.5-8.1	22-24	5.6-9.0	60	80-100 (MHSW)	1705	C,R,M	U (control survival was 62.5% & conducted using sediment & peat moss as substrate)	Elphick et al 2011
Amphipod (*Hyalella azteca*) (7-8 d)	Growth (28d, IC50, mean dry weight)	7.5-8.1	22-24	5.6-9.0	60	80-100 (MHSW)	2298	C,R,M	U (control survival was 62.5% & conducted using sediment & peat moss as substrate)	Elphick et al 2011
Amphipod (*Hyalella azteca*) (7-8 d)	Survival (28d, NOEC)	7.5-8.1	22-24	5.6-9.0	60	80-100 (MHSW)	2,210	C,R,M	U (control survival was 62.5% & conducted using sediment & peat moss as substrate)	Rescan Environmental Services Ltd., 2007
Amphipod (*Hyalella azteca*) (7-8 d)	Survival (28 d, LOEC)	7.5-8.1	22-24	5.6-9.0	60	80-100 (MHSW)	4,238	C,R,M	U (control survival was 62.5% & conducted using sediment & peat moss as substrate)	Rescan Environmental Services Ltd., 2007
Amphipod (*Hyalella azteca*) (7-8 d)	Survival (28d, EC50)	7.5-8.1	22-24	5.6-9.0	60	80-100 (MHSW)	2,453	C,R,M	U (control survival was 62.5% & conducted using sediment & peat moss as substrate)	Rescan Environmental Services Ltd., 2007
Amphipod (*Hyalella azteca*) (0-7 d)	Survival (28d LC10)					130 (dechlor Lake Ontario tap water)	733	C,S,M	S	Bartlett 2009 (unpublished)
Amphipod (*Hyalella azteca*) (0-7 d)	Survival (28d LC25)					130 (dechlor Lake Ontario tap water)	954	C,S,M	S	Bartlett 2009 (unpublished)

Amphipod (*Hyalella azteca*) (0-7 d)	Survival (28d LC50)					130 (dechlor Lake Ontario tap water)	1,200	C,S,M	S	Bartlett 2009 (unpublished)
Amphipod (*Hyalella azteca*) (0-7 d)	Growth (28d EC25 dry weight)					130 (dechlor Lake Ontario tap water)	421	C,S,M	S	Bartlett 2009 (unpublished)
Tubificid worm, Oligochaete (*Limnodrilus hoffmeisteri)*	Mortality (LC50, 10.9-d)						3,518	C	U (control survival not reported)	Wurtz and Bridges 1961
CHRONIC - AQUATIC PLANTS AND ALGAE										
Alga *(Chlamydomonas reinhardtii)*	Growth Inhibition (3-6 d, EC49, 49% decrease)						3,014	C	S	Reynoso et al. 1982
Alga *(Chlorella emersonii)*	Growth Inhibition (8-14 d MATC)		25-30				6,824	C	S	Setter et al. 1982
Alga *(Chlorella minutissimo)*	Growth (28d MATC)						6,066	C	S	Kessler (1974)
Alga *(Chlorella zofingiensis)*	Growth (28d MATC)						6,066	C	S	Kessler (1974)
Alga *(Anabaena variabilis)*	Growth (4d MATC)						14,300	C	S (salt tolerant)	Schiewer (1974)
Alga *(Chlorella fusca)*	Growth (28d MATC)						18,200	C	S (salt tolerant)	Kessler (1974)
Alga *(Chlorella kessleri)*	Growth (28d MATC)						18,200	C	S (salt tolerant)	Kessler (1974)
Alga *(Chlorella vulgaris)*	Growth (28d MATC)						18,200	C	S (salt tolerant)	Kessler (1974)
Alga *(Chlorella protothecoides)*	Growth (28d MATC)						30,300	C	S (salt tolerant)	Kessler (1974)
Alga *(Chlorella saccharophilia)*	Growth (28d MATC)						30,300	C	S (salt tolerant)	Kessler (1974)
Alga *(Chlorella luteoviridis)*	Growth (28d MATC)						36,400	C	S (salt tolerant)	Kessler (1974)
Alga *(Anacystis nidulans)*	Growth (4d MATC)						>24,300	C	S (salt tolerant)	Schiewer (1974)

Diatom *(Nitzschia linearis)*	Growth (5d or 120h EC50) 50% Reduction in number of cells			5-9			1,474	C	U (control survival not reported)	Patrick et al., 1968
Duckweed *(Lemna minor)*	Population (EC50, 7-d)						2,960	C,S	S	Buckley et al. 1996
Duckweed *(Lemna minor)*	Population (EC50, 7-d)						3,033	C,S	S	Buckley et al. 1996
Duckweed *(Lemna minor)*	Population (EC50, 7-d)						3,270	C,S	S	Buckley et al. 1996
Duckweed *(Lemna minor)*	Population (EC50, 7-d)						3,336	C,S	S	Buckley et al. 1996
Duckweed (*Lemna minor*)	96h MATC Frond production	7.3-7.6	24.5-25.6		2	39	1,171	C,U	S	Taraldson and Norberg-King (1990)
Eurasian millfoil *(Myriophyllum spicatum)*	Population (EC50, 32-d)						3,617	C	?	Stanley 1974 (In Bright and Addison 2002)
Eurasian millfoil *(Myriophyllum spicatum)*	Population (EC50, 32-d)						4,965	C	?	Stanley 1974 (In Bright and Addison 2002)
Eurasian millfoil *(Myriophyllum spicatum)*	Growth (EC50, 32-d)						4,504	C	?	Stanley 1974 (In Bright and Addison 2002)
Eurasian millfoil *(Myriophyllum spicatum)*	Growth (EC50, 32-d)						4,859	C	?	Stanley 1974 (In Bright and Addison 2002)
Freshwater green alga *(Scendesmus obliquus)*	Decrease in dry matter, photosynthetic pigment and oxygen production; increases in respiration, soluble saccharides and proteins, as well as lipid and proline content (7-d)		24-26				4,255	C	?	Mohammed and Shafea 1992
Freshwater green alga *(Scendesmus obliquus)*	Decrease in dry matter, photosynthetic pigment and oxygen production; increases in respiration, soluble saccharides and proteins, as well as lipid and proline content (7-d)		24-26				7,091	C	?	Mohammed and Shafea 1992
Freshwater green alga *(Scendesmus obliquus)*	Decrease in cell number to 43.1% of control (7-d)		24-26				7,091	C	?	Mohammed and Shafea 1992
Pondweed *(Potamogeton pectinatus)*	Reduced germination (28-d)						1,820	C	?	Teeter 1965 (In Evans and Frick 2001)

Pondweed *(Potamogeton pectinatus)* (13-week old plant)	Reduced shoots and dry weight (32-d)						1,820	C	?	Teeter 1965 (In Evans and Frick 2001)
Pondweed *(Potamogeton pectinatus)* (9-week old plant)	Reduced dry weight (32-d)						1,820	C	?	Teeter 1965 (In Evans and Frick 2001)

Assign 3 data codes, one from each of the following rows:
A-acute C-chronic F-flowthrough
S-static R-static renewal M-measured conc.
U-unmeasured nominal conc.

Data Quality
U- Unacceptable
P- Primary
S- Secondary
? - Unclassified (original document could not be obtained for review)

MATC: The Maximum Acceptable Toxicant Concentration