Environment and Climate Change Canada / Environnement et Changement climatique Canada

Canadian Environmental Protection Act, 1999

Federal Environmental Quality Guidelines

Lead

Environment and Climate Change Canada

July 2020

Introduction

Federal Environmental Quality Guidelines (FEQGs) provide thresholds of acceptable quality of the ambient environment. They are based solely on the toxicological effects or hazards of specific substances or groups of substances. FEQGs serve three functions: first they can be an aid to prevent pollution by providing targets for acceptable environmental quality; second, they can assist in evaluating the significance of concentrations of chemical substances currently found in the environment (monitoring of water, sediment, soil and biological tissue); and third, they can serve as performance measures to assess the effectiveness of risk management actions for the chemical substance. The use of FEQGs is voluntary unless prescribed in permits or other regulatory tools. Thus FEQGs, which apply to the ambient environment, are not effluent limits or "never-to-be-exceeded" values but may be used to derive effluent limits. The development of FEQGs is the responsibility of the Federal Minister of Environment under the *Canadian Environmental Protection Act, 1999* (CEPA 1999) (Government of Canada (GC) 1999). The intent is to develop FEQGs as an adjunct to risk assessment/risk management of priority chemicals identified in the Chemicals Management Plan (CMP) or other federal initiatives. Where data permit, FEQGs are derived following CCME protocols. FEQGs are developed where there is a federal need for a guideline (e.g., to support the federal Risk Management Strategy for lead or other monitoring activities) but where the CCME guidelines for the substance have not yet been developed or are not reasonably expected to be updated in the near future. More information on FEQGs is available on this page.

This factsheet describes the Federal Water Quality Guideline (FWQG) for the protection of aquatic life from adverse effects of lead (Table 1). The guideline is based on toxicity data identified up to June 2019. No FEQGs have been developed for the biological tissue, sediment or soil compartments at this time.

Table 1. Federal Freshwater Quality Guideline for Lead (dissolved).

Freshwater Aquatic Life	Guideline Value (µg/L)[a]
FWQG Equation	FWQG = exp(0.514[ln(DOC)] + 0.214[ln(Hardness)] + 0.4354)
Example FWQG	2.5

[a] FWQG is for dissolved lead for waters and is calculated using the equation above. As an example, the FWQG at a DOC concentration of 0.5 mg/L and hardness of 50 mg/L is 2.5 µg/L. The FWQG equation is valid for DOC 0.5-31.5 mg/L and hardness 4.7-511 mg/L.

Substance Identity

Lead (Pb) is a highly toxic naturally occurring element (CAS Number 7439-92-1) found in bedrock, soils, tills, sediments, surface waters, groundwaters and seawater (Reimann and de Caritat 1998; Health Canada 2013). Lead is one of the most abundant metals in the earth's crust (Adriano 2001), is highly resistant to chemical corrosion, and has no known biological function (Mager 2012). It is a class B, post-transition metal with a density of 11.34 g/cm^3 and a molecular weight of 207.2 g/mol. Lead has several oxidation states, but in nature the plumbous form predominates (ATSDR 2007). Lead can occur as stable organic compounds, such as tetraethyl lead, but this FWQG is for inorganic lead. Lead was one of the first substances added to the List of Toxic Substances (Schedule 1) of the *Canadian Environmental Protection Act* (CEPA), 1988 (GC 1988). It was determined that the scientific evidence of the adverse effects of lead on human health and the environment had been sufficiently demonstrated, and that determination of its adverse effects was consistent with the criteria set out in section 11 of CEPA 1988 (Health Canada 2013).

Canada is a major global producer and supplier of refined lead, ranked eighth in the world, producing 14,000 tonnes (t) of lead in concentrate and 277,000 t of refined lead production in 2017 (NRCan 2019). Canada operates two primary lead and four recycled lead smelters. Because of the significant recycling of lead-acid batteries, recycled lead accounted for 55% of Canada's total refined lead production. Canada exported

261,480 t of unwrought lead metal in 2017 and the majority of these exports were to the United States (NRCan 2019)

Uses

The primary use of lead is in the manufacturing of lead-acid batteries, but it is also used extensively in the manufacture of cable sheathing, circuit boards, lining for chemical baths and storage vessels, chemical transmission pipes, wheel weights, electrical components, polyvinyl chloride (as a chemical stabilizer) and radiation shielding (Health Canada 2013). Other uses include lead used in ammunition and fishing sinkers and jigs. Historically, leaded gasoline and lead-based paints were important sources of lead, but these products have been phased out (Health Canada 2013).

Total lead emissions to air in Canada in 2014 were 136 t, representing an increase of 5% (about 7 t) from 2013, and 89% (1153 t) lower than in 1990 (ECCC 2016). Emissions to air decreased between 1990 and 2014 as a result of reduced emissions from many industrial sectors, most notably the non-ferrous smelting and refining industry, and the mining industry. The national lead releases to water were 146 t of lead in 2014, more than 10 times the total releases reported in 2013, mainly due to 134.1 t released from the Mount Polley mine dam failure (ECCC 2016). Other categories contributing to most releases of lead to water were waste, accounting for approximately 5.1 t, followed by the pulp, paper and paperboard industry, representing 1.9 t of the national total. Further 500 t per year is released into aquatic environments from the unintentional loss of lead sinkers and jigs (Health Canada 2013). Historically, use of lead shot was a significant source of lead to aquatic environments. However, lead inputs from this source have declined in Canada since the 1997 national regulation prohibiting the use of lead shot for hunting migratory game birds (exempting American woodcock, mourning doves and band-tailed pigeons) within 200 m of any watercourse and the 1999 extension of ban to include all areas of land and water nationwide (Stevenson et al. 2005). Not accounted by this regulation, lead ammunitions annually release about 5,200 t of lead into environment and this source represents the single most significant source of lead releases in Canada (GC 2018).

Fate, Behaviour and Partitioning in the Environment

Atmospheric deposition, urban runoff and industrial discharge are the major sources of lead in surface waters (Health Canada 2013; USEPA 2006). In surface waters lead is predominantly found as PbO or $PbCO_3$. Fate, transport and the subsequent bioavailability of lead in both aquatic and terrestrial systems are primarily controlled by solubility (USEPA 2013). In aquatic environments at a pH of ≥7, lead is readily complexed and, with the exception of nitrate, chlorate and chloride salts, most inorganic lead salts are poorly soluble, however, the solubility of lead salts greatly increases under acidic conditions (Mager 2012). Lead speciation in freshwater environments is largely driven by pH, alkalinity and dissolved organic matter (DOM) (Mager 2012). Lead concentrations in surface water are largely controlled by exchange with sediments, and the cycling of lead between water and sediments is governed by chemical, biological and mechanical processes that are affected by many factors, including salinity, organic complexation, oxidation-reduction potential, and pH (USEPA 2006).

The majority of lead in surface waters occurs in undissolved forms, as colloidal particles or undissolved particles of lead carbonate, lead hydroxide, or other lead compounds, which precipitate to the sediment bed (Getz et al. 1977; Eisler 1988). Concentrations of dissolved lead are generally small in surface and groundwaters because lead forms complexes with sulphates, hydroxides, phosphates, carbonates and other anions in the water. Lead can occur as surface coatings on sediment mineral particles, sorbed ions or within suspended organic matter, living or non-living (LDAI 2008). The ratio of lead in suspended solids to dissolved lead varies between 4:1 in rural streams and 27:1 in urban streams (Getz et al. 1977).

The bioavailability of lead to freshwater aquatic organisms is influenced by a variety of water chemistry parameters (Van Sprang et al. 2016). It is generally assumed that the free Pb ion (i.e., Pb^{2+}) is the most toxic form of lead, although it still needs to be determined whether other chemical species (e.g., $PbOH^+$) are also important contributors to lead toxicity (Mager 2012). As low pH favours a greater proportion of Pb^{2+}, lead toxicity tends to be greater at low pH (although the higher concentration of protons $[H^+]$ at low pH can also compete with Pb^{2+} for uptake by aquatic organisms) (Mager 2012). Conversely, as pH increases, an increasing proportion of lead binds to carbonate and hydroxide ions, which are less bioavailable forms of lead. Very high hardness may also reduce lead solubility, but pH appears to be an important factor in determining lead solubility for the range of water chemistries typically used in toxicity testing (Mager 2012). In addition to pH, DOM is another water quality variable that strongly influences the bioavailability of lead to aquatic organisms (Mager et al. 2010). DOM can complex the majority of lead under most environmentally-relevant conditions, although the quality of the DOM also has an important influence on its binding capacity (Richards et al. 2001; Mager 2012). Finally, several water chemistry parameters co-vary, which can make it difficult to elucidate the relative influence of each water chemistry parameter on lead bioavailability. For example, hardness and pH often co-vary with alkalinity and it can be difficult to determine the relative influence of each (Mager 2012).

In general lead is relatively stable in sediments, with long residence times and limited mobility (Das et al. 2008). However, lead-containing sediment particles can be remobilized into the water column. As a result, trends in sediment concentration tend to follow those in overlying waters (LDAI 2008). Desorption, dissolution, precipitation, sorption and complexation processes can all occur concurrently and continuously, leading to transformations and redistribution of lead (USEPA 2013).

Lead is typically bound to organic matter and soil in terrestrial systems and the most important factors determining its solubility in soils are pH and cation exchange capacity (Smolders et al. 2009). Iron and manganese oxides are also known to play an important role in lead sequestration in soils (USEPA 2013). Because lead is strongly adsorbed to soil, it generally is retained in the upper layers of soil and does not leach appreciably into the subsoil and groundwater (ATSDR 2007). Organic matter decreases bioavailability of lead, but over time as it is broken down, pedogenic minerals become more important in sequestration of lead (Schroth et al. 2008). The binding of lead to organic matter is relatively weak and as the organic matter is broken down the lead may be released into soil solution (USEPA 2013). However, as lead ages in soils, through its incorporation into particulate solid-phase of the soil, the bioavailability of lead is reduced to plants and soil organisms.

Ambient Concentrations

Environment and Climate Change Canada (ECCC) monitoring data (unpublished), along with data from Alberta's Regional Aquatics Monitoring Program (RAMP) and British Columbia Ministry of Environment for lead concentrations in surface waters are summarized in Table 2. These data collected between 1997 and 2019 ranged from 0.002 µg/L to 309 µg/L for total lead and from 0.003 µg/L to 121 µg/L for dissolved lead.

Table 2. Concentrations of total and dissolved lead in Canadian surface waters.

Location	Sampling Years	Total or Dissolved	Mean (µg/L)	Min (µg/L)	Max (µg/L)
Great Lakes	2004-2014	Total	0.11	0.005	1.29
Great Lakes Connecting Channels	2003-2014	Total	0.38	0.005	10.5
St. Lawrence River	2007-2014	Total	0.38	0.02	6.95
	2002-2014	Dissolved	0.03	0.007	0.24
Newfoundland	2003-2006	Total	0.44	0.005	22.8
New Brunswick	2010-2013	Total	0.09	0.03	0.28
Nova Scotia	2013	Total	0.26	0.03	1.34
Saskatchewan	2003-2014	Total	0.54	0.005	27.3
	2003-2014	Dissolved	0.05	0.005	1.09
Manitoba	2003-2014	Total	1.42	0.01	21.5
	2003-2014	Dissolved	0.05	0.005	1.06
Athabasca Region	1997-2015	Total	0.9	0.003	309
	1997-2015	Dissolved	0.3	0.004	121

Location	Sampling Years	Total or Dissolved	Mean (µg/L)	Min (µg/L)	Max (µg/L)
Alberta	2003-2015	Total	0.85	0.005	60.1
	2003-2015	Dissolved	0.04	0.005	2.23
British Columbia	1998-2019	Total	0.04	0.002	11.7
	1998-2019	Dissolved	0.03	0.003	0.88
Northwest Territories	2003-2014	Total	2.2	0.005	222

Mode of Action

Mechanisms of acute lead toxicity in fish include a smothering effect at very high lead concentrations (e.g., 20 to >100 mg/L) and an ionoregulatory effect at lower lead concentrations that may be more typically observed in the environment (Mager 2012). As demonstrated in a series of studies with rainbow trout (*Oncorhynchus mykiss*), acute lead exposures can result in the disruption of Ca^{2+}, Na^{+}, and Cl^{-} homeostasis, with hypocalcemia being the primary cause of lead toxicity in fish (Rogers et al. 2003, 2005; Rogers and Wood 2004). In chronic lead exposures to fish, lead can adversely affect the growth and development of fish, including spinal deformity (Davies et al. 1976; Holcombe et al. 1976; Hodson et al. 1978). Although the mechanisms of how lead may affect growth in fish are not fully understood, it could be related to reduced feeding ability or appetite due to neurological effects of lead (Mager 2012).

The mechanisms of lead toxicity in invertebrates are less understood than in fish. However, where Pb^{2+} competes with Ca^{2+} at a common uptake site, it appears, at least for the sensitive cladoceran *C. dubia* at low aqueous lead concentrations, that Pb^{2+} is taken up via a channel or transporter that has a low affinity for Ca^{2+} (Mager et al. 2011a,b). For the snail *L. stagnalis*, also a chronically-sensitive species to lead, Ca^{2+} homeostasis can be affected by lead exposure, but this appears to be a secondary effect and does not explain the sensitivity of snail to lead (Brix et al. 2012). In plants, excess lead can lead to reduced mitosis, photosynthesis and water absorption, as well as growth inhibition (Eisler 1988).

Aquatic Toxicity

A detailed review of studies was performed following the CCME (2007) guidance for data quality. Determinants of test acceptability included, but were not limited to, exposure duration, analytical determination of lead exposure concentrations and other water quality parameters, documentation of the control response, the use of suitable biological endpoints and the inclusion of appropriate statistical analyses of the data collected in the study.

The chronic freshwater toxicity studies for lead were identified from existing data compilations (ECCC 2020). It is now generally accepted that the dissolved rather than the total recoverable fraction of most metals, including lead, reflects the fraction that is potentially bioavailable to aquatic organisms (Reiley et al. 2003). As summarized in Diamond et al. (1997), the relationship between total recoverable and dissolved lead is variable because it is heavily influenced by the amount of carbonate and hence alkalinity (Davies et al. 1976; Sprague 1995). Van Sprang et al. (2016) assessed dissolved versus total lead measurements in toxicity studies and concluded that in all studies they examined estimated inorganic lead concentration was lower than the inorganic solubility limit, and therefore precipitation was considered unlikely to have occurred in studies where only total lead concentrations were reported. Therefore, the dissolved lead concentration was assumed to be equal to the total lead concentration.

Acceptable chronic toxicity data were available for 28 species (11 fish, 13 invertebrates and 4 plants/algae) (ECCC 2020) and the dataset met the CCME's (2007) minimum data requirements for developing the SSD-based guideline (i.e., Type A guideline). Within the acceptable dataset, several studies evaluated the influence of varying water chemistry on the bioavailability of lead and related chronic toxicity to aquatic species. The toxicity data for these species were used to evaluate toxicity modifying factors (TMFs) to develop FWQGs for lead that could be adjusted for the site-specific water chemistry (ECCC 2020).

The CCME (2007) protocol states that, where possible, it is important to account for exposure and toxicity modifying factors in guideline derivation. This may be done through single or multi-factor equations, matrices or models (CCME 2007). Multiple linear regression (MLR) analysis was explored as an approach to account for the simultaneous effect of multiple water chemistry variables on lead toxicity. In conducting forward step-wise MLR (e.g., Neter et al. 1990), the independent variable (in this case water hardness, DOC or pH) that explains the greatest amount of the variability in the dependent variable (in this case lead toxicity) is entered first. If the relationship between this independent variable and the dependent variable is not significant, the modelling process would be considered complete (that no MLR model could be developed). If the relationship is significant, the variable is retained and the independent variable that explains the greatest proportion of the remaining variability is entered next. If this second variable does not explain a significant additional percentage of the variability, the second variable is removed, and the final model contains only the first independent variable that was entered. If the relationship is significant, the second variable is retained, and the independent variable explaining the next highest proportion of the remaining variability is entered, and so on.

The increased understanding of how various factors influence the bioavailability of metals has led to the development of the biotic ligand model (BLM) (Di Toro et al. 2001; Santore et al. 2001; Paquin et al. 2002). The BLM is a mechanistic model that predicts metal bioavailability to aquatic organisms by considering competition for metal ions at the "biotic ligand" of the organism (e.g., fish gills) and other ligands in the water (e.g., DOC), as well as competition between uptake of metals and other cations in the water (e.g., calcium) (Di Toro et al. 2001). The BLM has been used to develop water quality criteria/guidelines (USEPA 2007; EU 2008a,b,c, 2010; DeForest et al 2017). The BLM for lead and other metals served as an important basis for identifying the key variables considered in the MLR-based approach considered in the current evaluation.

Multiple linear regression analysis was evaluated as a potential tool for deriving the FWQG for two primary reasons. First, MLR models are a reliable tool for predicting metal toxicity (DeForest et al. 2018), including lead toxicity (Esbaugh et al. 2011, 2012) over a range of water chemistries. Second, it is a linear model that is conceptually similar to the already accepted hardness-based model. As such, the procedure for developing potential MLR-based WQGs would follow the same basic steps that have already been accepted for developing hardness-based WQGs (CCME 2007) and the resulting equation would have the same basic structure as a hardness-based WQG, but instead of just a hardness slope it may, for example, contain slopes for hardness, DOC and pH. For several metals (e.g., aluminium, lead, nickel, zinc), it has been observed that MLR models are able to predict toxicity as well as BLMs (Esbaugh et al. 2012; Brix et al. 2017; DeForest et al. 2018; USEPA 2018).

Forward stepwise multiple linear regression (MLR) analyses were conducted using SYSTAT (Version 13) statistical software. The MLR analyses determines whether a significant portion of variability in toxicity could be explained by hardness, pH and/or DOC. MLR analysis was conducted for a given species if toxicity data were available from tests in which the range of hardness exceeded 100 mg $CaCO_3$/L (with the highest hardness being three times the lowest), the range of DOC exceeded 5 mg/L (with the highest DOC being three times the lowest) and the range of pH spanned at least 1.5 units. The acceptable toxicity data could include data combined from multiple studies for the same species. MLR analyses were conducted on a species-by-species basis, whereby toxicity values for a given species were the dependent variables and the water chemistry values were the independent variables. Individual species MLRs were conducted for the following species: *Brachionus calyciflorus, Ceriodaphnia dubia, Lymnaea stagnalis, Pimephales promelas,* and *Pseudokirchneriella subcapitata* (Grosell et al. 2006; Nys et al. 2016; Cooper et al. 2009; Mager et al. 2011b; Esbaugh et al. 2012; Parametrix 2010a,b; De Schamphelaere et al. 2014). A pooled MLR analyses was also conducted including the invertebrates and fish stated previously with the addition of *Philodina rapida*. *Philodina rapida* was not included in the individual species MLR analyses because the range of pH did not meet the minimum requirement but was included in the pooled MLR since it has been tested over a wide range of DOC and hardness conditions. The green algae *P. subcapitata* could not be included in the pooled MLR since the MLR analyses indicated the TMF relationships were significantly different compared with the fish and invertebrates. All MLR models were derived using EC_{10} effect concentrations.

The results of MLR analysis are presented in Table 3. None of the variables were significant for *B. calyciflorus*. The *P. subcapitata* MLR was the only MLR to retain pH. Both DOC and hardness were significant in the MLR analysis for the pooled invertebrate and fish model, whereas pH was not significant. These MLR relationships are referred to as MLR models hereon in this factsheet.

Table 3. Summary results of MLR analysis.

			Model Coefficients			
Species	n	r^2	Ln DOC	pH	Ln Hardness	Intercept
Brachionus calyciflorus	18	0	-	-	-	-
Ceriodaphnia dubia	28	0.2	0.682	-	-	2.649
Pimephales promelas	8	0.68	-	-	0.984	1.989
Lymnaea stagnalis	6	0.69	1.259	-	-	-0.229
Pseudokirchneriella subcapitata	15	0.81	0.473	-1.542	-	12.629
Pooled (invertebrates and fish)[a]	66	0.72	0.514	-	0.214	2.156

[a]Includes data from 5 species (*B. calyciflorus, C. dubia, P. promelas, L. stagnalis* and *P. rapida). P. rapida* was not included in the individual species MLR analyses because the range of pH did not meet the minimum requirement but was included in the pooled MLR because it has been tested over a wide range of DOC and hardness conditions.

The pooled MLR model incorporates 66 data points from 5 different species, retained both DOC and hardness variables with r^2 value of 0.72 and wide water chemistry ranges (DOC range 0.5-31.5 mg/L and hardness range 4.7-511 mg/L). The pooled (invertebrate and fish) MLR model was therefore chosen for deriving the guideline.

Federal Water Quality Guideline Derivation

Federal Water Quality Guidelines (FWQGs) are preferably developed using the CCME (2007) protocol. In the case of lead, there were sufficient acceptable chronic toxicity data to meet the minimum data requirements for the preferred CCME Type A approach. A Type A guideline is a statistical approach that uses species sensitivity distributions (SSD) to calculate a hazard concentration of 5% of species (HC_5), which in turn is the final guideline value (CCME 2007).

The first step in developing the FWQG for lead was to normalize the toxicity values to a common water chemistry using the pooled MLR. The chronic toxicity data for all 28 species were normalized to water with a DOC concentration of 0.5 mg/L and hardness of 50 mg/L. All data in the toxicity dataset were within the acceptable ranges of the pooled MLR (DOC range 0.5-31.5 mg/L and hardness range 4.7-511 mg/L) and therefore were able to be considered in guideline derivation. Where multiple comparable endpoints were available for the same species, effect, life stage and exposure duration, a geometric mean was calculated. In an effort to include data for preferred endpoints, if studies did not report an EC_{10}, but reported sufficient information to develop a concentration-response curve, EC_{10} values were calculated using the USEPA Toxicity Relationship Analysis Program (TRAP v. 1.3) (USEPA 2015).

The most sensitive and preferred endpoint (or geometric mean) was then selected for each species following CCME (2007). A total of 82 endpoints (79 EC_{10s}, 1 EC_{20}, 1 NOEC and 1 MATC) for 28 species were included in the SSD dataset and summarized in Table 4. *Lymnaea stagnalis* was the most sensitive species with a normalized species mean toxicity value of 1.8 µg/L and northern pike (*Esox lucius*) was the least sensitive species with a normalized effect concentration of 376.4 µg/L.

Table 4. Pooled multiple linear regression based normalized chronic lead toxicity endpoints for toxicity data used in deriving the FWQG for lead.

Species	Group	Endpoint	Effect Concentration (µg/L)	Normalized Effect Concentration (µg/L)[a]	Reference
Lymnaea stagnalis (great pond snail)	Invertebrate	14-d EC10 (growth)	Geomean (n=6)	1.8	Esbaugh et al. 2012; personal communication
Philodina rapida (rotifer)	Invertebrate	4-d EC10 (population growth)	Geomean (n=6)	3.8	Esbaugh et al. 2012
Ceriodaphnia dubia (cladoceran)	Invertebrate	6-d EC10 (survival and reproduction)	Geomean (n=3)	4.1	AquaTox 2012
Hyalella azteca (amphipod)	Invertebrate	42-d EC10 (survival, growth and reproduction)	5	4.1	Besser et al. 2016
Pseudokirchneriella subcapitata (alga)	Plant	72-h EC10 (Mean growth rate)	Geomean (n=15)	5.9	De Schamphelaere et al. 2014
Lampsilis siliquoidea (fatmucket)	Invertebrate	28-d EC10 (survival and growth)	6	6.5	Wang et al. 2010
Oncorhynchus mykiss (rainbow trout)	Fish	62-d EC10 (weight)	7	7.0	Mebane et al. 2008; personal communication
Lymnaea palustris (marsh snail)	Invertebrate	120-d EC20 (survival and growth)	23	10.2	Borgmann et al. 1978; Van Sprang et al. 2016
Diaphanosoma birgei (cladoceran)	Invertebrate	25-d EC10 (net reproductive rate)	13	11.9	Garcia-Garcia et al. 2006; Van Sprang et al. 2016
Chironomus dilutus (midge)	Invertebrate	21-d EC10 (emergence)	15	15.0	Mebane et al. 2008; personal communication
Daphnia magna (cladoceran)	Invertebrate	21-d EC10 (survival and reproduction)	Geomean (n=3)	15.4	Chapman et al. 1980
Acipenser transmontanus (white sturgeon)	Fish	53-d EC10 (survival)	26	25.0	Ingersoll and Mebane 2014
Alona rectangula (cladoceran)	Invertebrate	25-d EC10 (gross reproductive rate)	40	35.9	Garcia-Garcia et al. 2006; Van Sprang et al. 2016
Pimephales promelas (fathead minnow)	Fish	30-d EC10 (survival and growth)	Geomean (n=3)	36.0	Mager et al. 2011b
Baetis tricaudatus (mayfly)	Invertebrate	10-d EC10 (molting)	37	40.9	Mebane et al. 2008; personal communication
Brachionus calyciflorus (rotifer)	Invertebrate	2-d EC10 (population size)	Geomean (n=18)	43.1	Nys et al. 2016
Chlaydomonas reinhardtii (green alga)	Plant	48-h EC10 (growth rate)	82	45.7	De Schamphelaere et al. 2014
Salvelinus namaycush (lake trout)	Fish	60-d EC10 (weight)	50	54.7	Sauter et al. 1976
Lemna minor (duckweed)	Plant	7-d EC10 (root growth)	Geomean (n=7)	58.3	Antunes and Kreager 2014
Lepomis macrochirus (bluegill)	Fish	60-d EC10 (survival)	57	59.1	Sauter et al. 1976
Salvelinus fontinalis (brook trout)	Fish	12-wk EC10 (growth)	88	60.2	Holcombe et al. 1976
Chlorella kesslerii (green alga)	Plant	48-h EC10 (growth rate)	120	66.6	De Schamphelaere et al. 2014
Chironomus riparius (midge)	Invertebrate	14-d EC10 (survival and growth)	Geomean (n=2)	76.5	Nguyen et al. 2012
Ictalurus punctatus (channel catfish)	Fish	60-d EC10 (survival)	76	81.3	Sauter et al. 1976
Catostomus	Fish	60-d EC10 (weight)	101	107.4	Sauter et al. 1976

Species	Group	Endpoint	Effect Concentration (µg/L)	Normalized Effect Concentration (µg/L)[a]	Reference
commersoni (white sucker)					
Stizostedion vitreum (walley)	Fish	30-d EC10 (survival)	148.62	158.0	Sauter et al. 1976
Micropterus dolomieui (smallmouth bass)	Fish	90-d NOEC (growth)	308	242.4	Coughlan et al. 1986
Esox lucius (Northern pike)	Fish	20- d MATC (survival)	349.57	376.4	Sauter et al. 1976

[a]Normalized to a hardness of 50 mg/L and DOC of 0.5 mg/L using the pooled invertebrate and fish MLR

The R package (R version 4.3.1) 'ssdtools' (version 1.0.6) (Thorley and Schwarz 2018) as well as the corresponding user friendly "Shiny App" (shinyssdtools version 0.1.1) (Dalgarno 2018) were used to create SSDs from the dataset. The package fit several cumulative distribution functions (CDFs) (log-normal, log-logistic, log-Gumbel, log-normal_log-normal, Gamma and Weibull) to the data using maximum likelihood estimation (MLE) as the regression method. Akaike information criterion (AIC), which is a measure of the relative quality of fit to the data set, was calculated for each distribution (Burnham and Anderson 2002). Using AICc, AIC corrected for small sample size, a model averaged HC_5 can be established. The smaller the AICc the better the distribution fits the data set. Each model was then weighted, models with high value weight fit the data well compared to the others. The SSD and accompanying summary statistics at water hardness 50 mg/L and DOC 0.5 mg/L are presented in Figure 1 and Table 5, respectively.

Table 5. SSD summary statistics at 50 mg/L hardness and 0.5 mg/L DOC.

Distribution	AICc	Predicted HC_5 (µg/L)	95% LCL and UCL (µg/L)	Weight	Weighted HC_5 (µg/L)	Weighted 95% LCL and UCL (µg/L)
Log-normal	-46.9	3.04	(1.49-6.64)	0.4	1.22	(0.6-2.67)
Log-logistic	-45.2	2.81	(1.15-6.83)	0.17	0.49	(0.2-1.18)
Log-Gumbel	-42.9	3.38	(2.11-6.34)	0.06	0.19	(0.12-0.35)
Log-normal_Log-normal	-42.6	3.06	(1.92-5.91)	0.05	0.15	(0.09-0.29)
Gamma	-44.6	1.45	(0.292-6.2)	0.13	0.19	(0.04-0.81)
Weibull	-45.4	1.42	(0.402-4.89)	0.19	0.27	(0.08-0.94)
				Guideline =	2.5	(1.13-6.33)

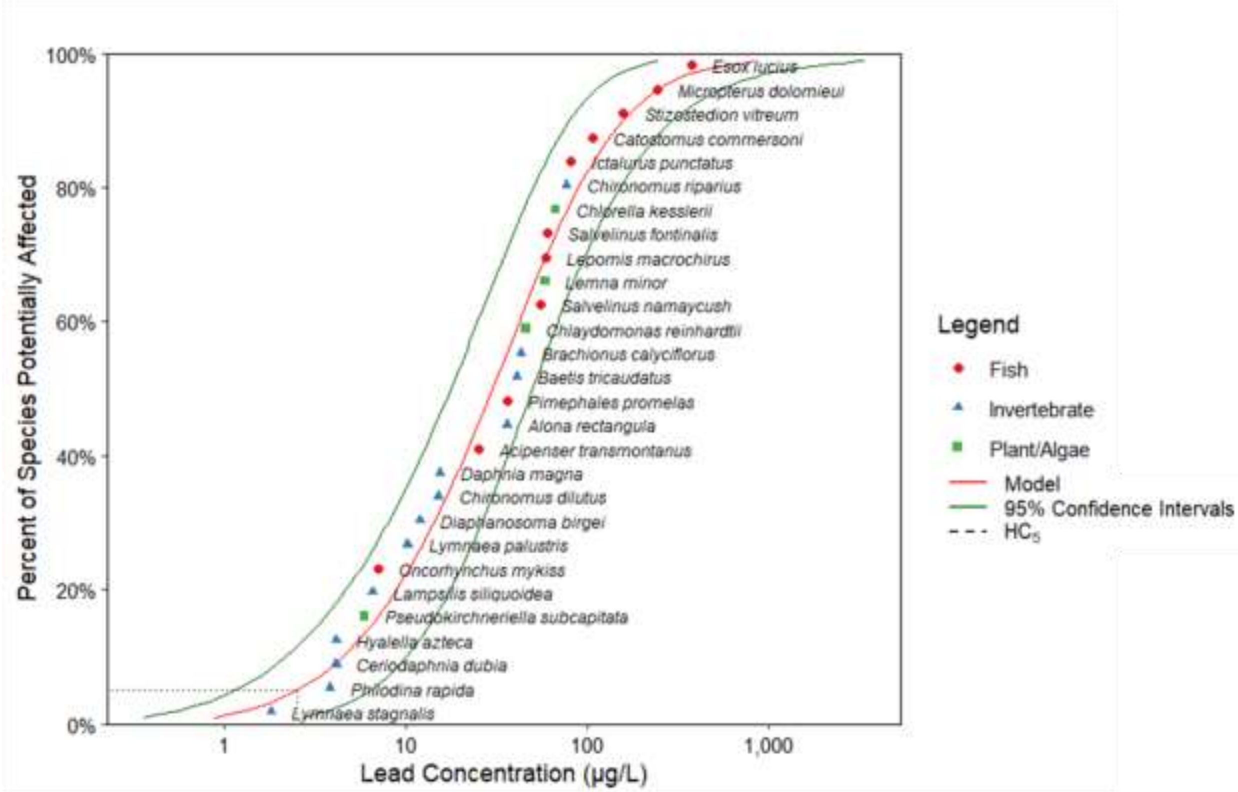

Figure 1. Species sensitivity distribution (SSD) for the chronic toxicity of lead at DOC of 0.5 mg/L and hardness of 50 mg/L. The HC_5 is 2.5 µg/L.

Because DOC and hardness were identified as significant toxicity modifying factors, the FWQG is expressed as an equation where the local water DOC and hardness are entered in order to calculate a site-specific FWQG. The equation is based on the pooled MLR model slopes of 0.514 and 0.214, respectively, and the 5th percentile value of 2.5 µg/L derived from the SSD at DOC of 0.5 mg/L and hardness of 50 mg/L.

Based on the pooled MLR model and the HC_5 from the SSD, the equation to derive FWQG for lead is:

$$\begin{aligned}\text{y-intercept} &= \ln(5^{th}\text{ percentile}) - [\text{DOC slope} \times \ln(\text{DOC})] - [\text{hardness slope} \times \ln(\text{hardness})] \\ &= \ln(2.50) - [0.514 \times \ln(0.5)] - [0.214 \times \ln(50)] \\ &= 0.4354\end{aligned}$$

$$\text{FWQG } (\mu g/L) = \exp(0.514\,[\ln(\text{DOC})] + 0.214[\ln(\text{hardness})] + 0.4354)$$

The FWQG equation was derived for dissolved lead and is found by using the equation above or by using the FWQG calculator (Appendix). Users can input site-specific DOC and hardness measurements to calculate a FWQG for the specific water chemistry. The FWQG equation is valid between DOC 0.5 and 31.5 mg/L and hardness 4.7 and 511 mg/L, which are the ranges of data used to derive the MLR slopes. Only values within these DOC and hardness ranges should be entered into the guideline equation to ensure the equation is accurate and the FWQG is protective. Users should be extremely cautious if extrapolating beyond the recommended ranges of DOC and hardness and should contact their local authority for advice. Although the hardness is regularly measured during monitoring, DOC may not always be routinely measured. In the absence of site-specific data, a DOC concentration of 0.5 mg/L and a hardness concentration of 4.7 mg/L may be assumed, which are the lower limits of the FWQG equation. For water bodies where lead concentrations are potentially of concern, it is recommended that both DOC and hardness be measured. It is to be also noted that the FWQG for lead is for dissolved concentration of lead. When guideline users only have total lead concentrations for their site, it is recommended that they first compare their total lead concentration to dissolved lead guideline, and where there is an exceedance, re-sample the waterbody for the dissolved lead. Examples of FWQGs for lead for selected DOC and hardness values are given in Table 6.

Table 6. FWQGs (µg/L) for lead for the protection of aquatic life for selected DOC and hardness values.

	Hardness (mg/L)					
DOC (mg/L)	**50**	**100**	**200**	**300**	**400**	**500**
0.5	2.5	2.9	3.4	3.7	3.9	4.1
2	5.1	5.9	6.9	7.5	8.0	8.3
5	8.2	9.5	11.0	12.0	12.7	13.4
10	11.7	13.5	15.7	17.1	18.2	19.1
20	16.6	19.3	22.4	24.4	26.0	27.3
30	20.5	23.8	27.6	30.1	32.0	33.6

To assess whether the FWQG for lead is sufficiently protective, a protectiveness assessment was conducted following CCME (2007). FWQGs were calculated for each of the 134 acceptable endpoints in the toxicity dataset. The FWQGs were then compared to measured toxicity values at their associated water chemistry (Figure 2). Values that plot above the 1 to 1 line indicate that the FWQG is protective of the toxicity value in that particular test, while values below the 1 to 1 line indicate that the FWQG is lower than the observed toxicity, and hence may require further evaluation. This protectiveness assessment resulted in 11 out of 134 (92%) acceptable toxicity data points being above the site-specific guideline. These 11 endpoints include 8 endpoints for invertebrates (3 for *C. dubia*, 4 for *L. stagnalis*, 1 for *Philodina rapida*) and 3 endpoints for plant species *P. subcapitata*. It is important to note that an equal or greater number of endpoints for these 4 species were above the 1 to 1 line (31 for *C. dubia*; 4 for *L. stagnalis*, 5 for *Philodina rapida* and 12 for *P. subcapitata*). None of the endpoints below the guideline were for a species at risk, or for lethal effects equal to or above a level of 15% (CCME 2007). Overall examination of the available data suggests the lead FWQG is protective.

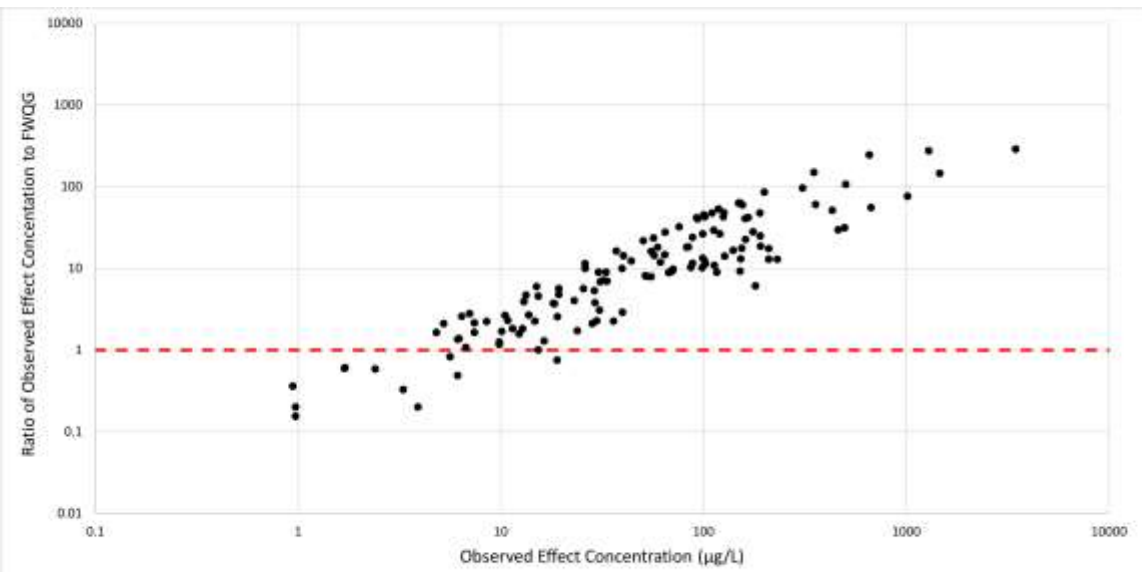

Figure 2. Ratio of chronic effect concentration for lead to FWQG calculated using the 5 species pooled MLR model containing hardness and DOC.

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List of Acronyms and Abbreviations

AIC – Akaike information criterion
ATSDR – Agency for Toxic Substances and Disease Registry
BLM – biotic ligand model
CAS – Chemical Abstracts Service
CCME – Canadian Council of Ministers of Environment
CDF – Cumulative Distribution Function
CEPA – Canadian Environmental Protection Act
CMP – Chemicals Management Plan
DOC – dissolved organic carbon
DOM – dissolved organic matter
EC – effect concentration
EU – European Union
ECCC – Environment and Climate Change Canada
FEQG – Federal Environmental Quality Guideline
FWQG – Federal Water Quality Guideline
GC – Government of Canada
MATC – maximum acceptable toxicant concentration

MLE – maximum likelihood estimation
MLR – multiple linear regression
NOEC – no observed effect concentration
NRCan – Natural Resources Canada
SSD – species sensitivity distribution
TMF – toxicity modifying factor
USEPA – United States Environmental Protection Agency