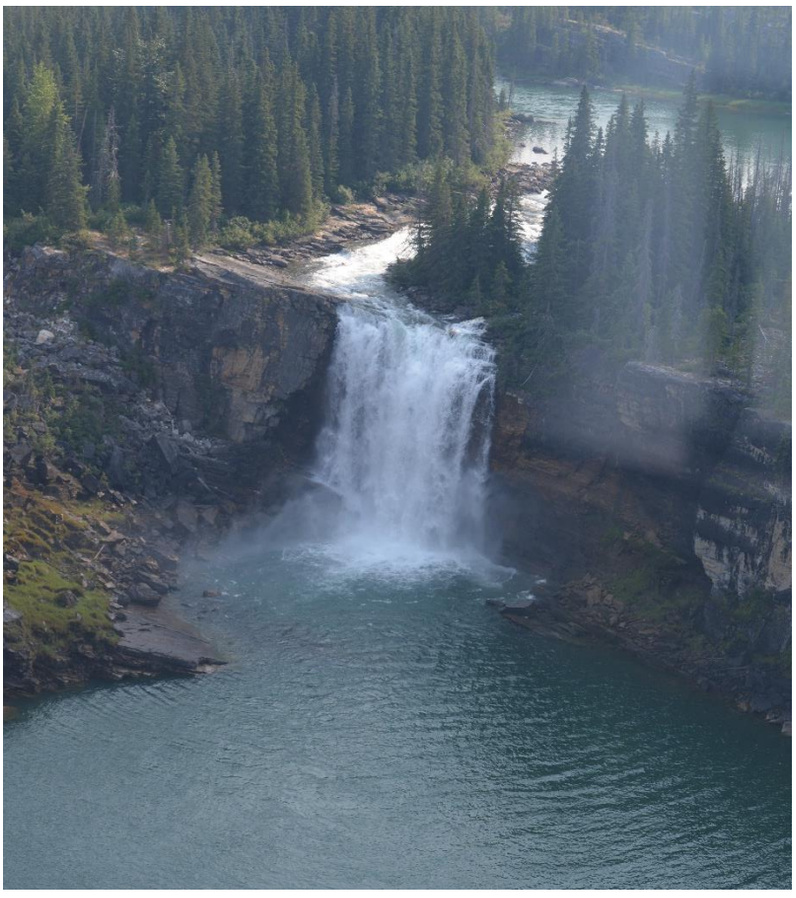
WATER QUALITY GUIDELINE SERIES

Nickel Water Quality Guideline for the Protection of Freshwater Aquatic Life

Technical Report

Ministry of Water, Land, and Resources Stewardship

Water Protection & Sustainability Branch



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BRITISH COLUMBIA

The Water Quality Guideline Series is a collection of British Columbia (B.C.) Ministry of Water, Land, and Resource Stewardship water quality guidelines. Water quality guidelines are developed to protect a variety of water values and uses: aquatic life, drinking water sources, recreation, livestock watering, irrigation, and wildlife. The Water Quality Guideline Series focuses on publishing water quality guideline technical reports and guideline summaries using the best available science to aid in the management of B.C.'s water resources. For additional information on B.C.'s approved water quality parameter specific guidelines, visit:

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EXECUTIVE SUMMARY

The B.C. Ministry of Water, Land, and Resources Stewardship (WLRS) develops province-wide ambient Water Quality Guidelines (WQGs) for substances or physical attributes that are important for managing both the fresh and marine surface waters of B.C. WQGs do not have direct legal standing but are used to provide a basis for evaluating the quality of water, sediment, and aquatic biota and environmental impact assessments, and to inform resource management decisions.

The development of WQGs for aquatic life is based on the principle that guideline values are protective of all forms of aquatic life and all aquatic life stages over indefinite exposure (ENV 2019a). For some substances, both a long-term chronic (30-day average) and a short-term acute (maximum) guideline are recommended as provincial WQGs, provided sufficient toxicological data are available. To meet a WQG, both of its components (i.e., chronic long-term and acute short-term) must be met. However, an exceedance of the WQGs does not imply that unacceptable risks are present, but that the potential for adverse effects may be increased and additional investigation and monitoring may be warranted.

This document provides information on the derivation of a nickel (Ni) WQG for the protection of aquatic life. Elevated concentrations of Ni can negatively affect aquatic life and exposure to Ni can decrease growth in aquatic plants. Acute exposure to Ni can cause mortality and chronic exposure can affect growth, reproduction and survival of fish, amphibians, and invertebrates.

Previously, B.C. used the CCME Ni WQG published in 1987 (CCREM 1987). Since 1987, many scientific studies have been published on the different aspects of Ni toxicity. The toxicity of Ni to aquatic organisms has made it a metal of concern for jurisdictions around the world, with guidelines published by the United States Environmental Protection Agency (U.S. EPA 1996), European Union (EC 2011), and Australian and New Zealand Environment and Conservation Council (ANZECC 2000a).

Nickel is a naturally occurring element that occurs in both dissolved and particulate states in freshwater ecosystems. Ambient Ni concentrations vary across the province depending upon the underlying geology. The median dissolved Ni concentration is 0.29 µg/L (90^{th} percentile is 1.23 µg/L) and the median total Ni concentration (dissolved + particulate) is also 0.29 µg/L (90^{th} percentile of 2.8 µg/L). These background concentrations were considered when deriving the Ni WQG.

The toxicity of Ni to aquatic organisms is modified by the water chemistry of the receiving water. The major factors influencing Ni toxicity are dissolved organic carbon (DOC) and water hardness. DOC binds to the toxic inorganic forms of Ni in water forming organic compounds that are not toxic to aquatic organisms. Increased water hardness can ameliorate Ni toxicity through the competition between calcium (Ca^{2+}) and Ni for biological uptake. pH also influences Ni toxicity, but to a lesser degree.

The B.C. Ni WQG was developed using a Biotic Ligand Model (BLM) which incorporates metal chemistry and the protective effects of competing cations, such as Ca^{2+}, to predict toxic concentrations. Using the BLM, Ni WQGs can be calculated that account for site-specific water chemistry. The BC Ni BLM requires 11 essential water chemistry parameters, however, not all of these water chemistry parameters are routinely measured. To overcome this issue, a simplified version of the BLM was incorporated in the BC Ni BLM to provide an estimate of the full WQG using only four water chemistry parameters: temperature, DOC, pH, and hardness. From these parameters, the other seven can be estimated.

The Ni toxicity database developed for this WQG meets the minimum requirements for the derivation of type A2 chronic and acute WQGs (ENV 2019a). The BC Ni BLM normalizes the toxicity datasets (chronic or acute) to the intended water chemistry and then fits the normalized data to species sensitivity distributions following the BC WQG derivation protocol (ENV, 2019a) and calculates a weighted HC_5

(hazard concentration). The BC Ni BLM then applies an assessment factor of two for chronic and three for acute, to the resultant HC_5 to address sources of uncertainty as the final step in calculating the WQG.

The long-term chronic and short-term acute Ni WQGs for the protection of aquatic life are based on dissolved Ni. Dissolved Ni provides a better estimate of bioavailable Ni since the Ni associated with suspended particulates is generally not available for biological uptake. Total metal concentrations still provide valuable information, especially in systems with high total Ni to dissolved Ni ratios, as changes in water chemistry (e.g., pH) can change the dynamics of particulate and dissolved Ni.

As the Ni WQGs are determined in relation to water chemistry conditions, Ni WQGs were calculated under several water chemistry scenarios and are provided below in Table E.1. Additional Ni WQGs, for specific scenarios, can be calculated using the BC Ni BLM software.

Table E.1. Long term chronic and short term acute WQGs calculated for 18 different water chemistry scenarios.

Scenario	Water Chemistry Conditions				Chronic WQG (µg/L)	Acute WQG (µg/L)
	Temperature (°C)	Hardness (mg/L)	DOC (mg/L)	pH		
1	15	20	5	5.5	0.8	12.5
2	15	20	5	7	1	15.5
3	15	20	5	8.5	1.8	20.1
4	15	20	10	5.5	1	16.7
5	15	20	10	7	1.5	23.9
6	15	20	10	8.5	3.7	36.8
7	15	20	20	5.5	1.6	26.8
8	15	20	20	7	2.6	43.8
9	15	20	20	8.5	7.9	76.4
10	15	150	5	5.5	2.6	50
11	15	150	5	7	2.9	52.9
12	15	150	5	8.5	3.4	48.5
13	15	150	10	5.5	2.8	53.3
14	15	150	10	7	3.4	61.4
15	15	150	10	8.5	5.9	66.7
16	15	150	20	5.5	3.1	60.1
17	15	150	20	7	4.5	78.9
18	15	150	20	8.5	11.4	105.0

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Appendix 1. Summary of toxicity data from short-term and long-term nickel toxicity tests. Data are presented in an Excel workbook.

LIST OF ABBREVIATIONS

AEP: Alberta Environment and Parks

ANZECC: Australia and New Zealand Environment and Conservation Council

ASTM: American Society for Testing and Materials

BAF: Bioaccumulation Factor

B.C.: British Columbia

BCF: Bioconcentration Factor

BLM: Biotic Ligand Model

Ca: Calcium

CABIN: Canadian Aquatic Biomonitoring Network

CCC: Criteria Continuous Concentration

CCME: Canadian Council of Ministers of the Environment

Cd: Cadmium

CMC: Criteria Maximum Concentration

Cu: Copper

DL: Detection Limit

DOC: Dissolved Organic Carbon

EC: Effect Concentration

ECCC: Environment and Climate Change Canada

EMS: Environmental Management System

ENV: British Columbia Ministry of Environment and Climate Change Strategy

EQS: Environmental Quality Standards

Fe: Iron

IC: Inhibitory Concentration

LC: Lethal Concentration

LOEC: Lowest Observed Effect Concentration

MATC: Maximum Allowable Toxicant Concentration

MDL: Method Detection Limit

Mg: Magnesium

Na: Sodium

Ni: Nickel

NOEC: No Observed Effect Concentration

NOM: Natural Organic Matter

PQL: Practical Quantitation Limit

SSD: Species Sensitivity Distribution

TGV: Trigger Value

TOC: Total Organic Carbon

USEPA: United States Environmental Protection Agency

WLRS: B.C. Ministry of Water, Land, and Resources Stewardship

WQC: Water Quality Criteria

WQG: Water Quality Guideline

DEFINITIONS

Lowest Observed Effect Concentration (LOEC): The lowest tested concentration of a substance that has been reported to have statistically significant harmful effects on organisms tested.

No Observed Effect Concentration (NOEC): The highest tested concentration of a substance that has been reported to have no statistically significant harmful effects on organisms tested.

EC_X: The concentration affecting X percent of the population within a certain amount of time.

IC_X: The concentration causing an X% inhibition in exposed individuals within a certain amount of time.

LC_X: The concentration which causes X% mortality in the exposed population within a certain amount of time.

Assessment Factor: Mathematical adjustments to guideline values to account for incomplete knowledge.

1. INTRODUCTION

The B.C. Ministry of Water, Land, and Resources Stewardship (WLRS) develops province-wide ambient Water Quality Guidelines (WQGs) for substances or physical attributes that are important for managing both the fresh and marine surface waters of B.C. WQGs do not have direct legal standing but are used to provide a basis for the evaluation of data on water, sediment, and biota for water quality and environmental impact assessments.

The approach to develop WQGs for aquatic life is based on the guiding principle that guideline values are protective of all forms of aquatic life and all aquatic life stages over indefinite exposure (ENV, 2019a). For some substances both a long-term chronic (30-day average) and a short-term acute (maximum) guideline are recommended as provincial WQGs, provided sufficient toxicological data are available. To meet a WQG, both of its components (i.e., chronic long-term and acute short-term) must be met. However, an exceedance of the WQGs does not imply that unacceptable risks are present, but that the potential for adverse effects may be increased and additional investigation and monitoring may be warranted.

Water quality guidelines are based on toxicological studies conducted under laboratory conditions. There are several uncertainties associated with applying WQGs in field conditions including:

- Laboratory to field differences in conditions;
- Single contaminant tests in laboratories vs exposure to multiple contaminants in the field (additive, synergistic, antagonistic effects);
- Intra and inter-specific differences (laboratory studies are conducted on a limited number of individuals within a species and a limited number of species which may or may not include the most sensitive species);
- Indirect effects (e.g., behavioral responses, food web dynamics);
- Partial-life cycle studies (many toxicity studies are only conducted on partial life-cycles and may not include the most sensitive life stage);
- Delayed effects (effects may not occur within the life stage tested, or may occur across generations); and
- Cumulative effects (compared with laboratory studies, organisms in the field are faced with various stressors such as habitat loss and climate change).

Given the uncertainties, WQGs can be considered as predicted no effect concentrations. However, it is an estimated value based on laboratory studies. Therefore, ongoing ecological monitoring is generally recommended to ensure the WQG is protective under field conditions.

Elevated concentrations of Ni can negatively affect aquatic organisms. Exposure to Ni can decrease growth in algae and macrophytes (e.g., Schlekat et al. 2010). Acute exposure to Ni can cause mortality and chronic exposure can affect growth, reproduction and survival of fish, amphibians and invertebrates (e.g., Leonard and Wood 2013; ECCC 2018; Klemish et al. 2018).

Previously, B.C. used the CCME Ni WQG published in 1987 (CCREM 1987). Since 1987, many scientific studies have been published on the different aspects of Ni toxicity. The economic importance and high production of Ni, combined with its toxicity to aquatic organisms has made it a metal of concern for jurisdictions around the world, with updated guidelines published by the United States Environmental Protection Agency (U.S. EPA 1996), European Union (EC 2011), and Australian and New Zealand Environment and Conservation Council (ANZECC 2000a).

This report provides the scientific rationale for B.C.'s Ni WQG, including a review of the current scientific literature on the toxicity of Ni to freshwater aquatic life. Key studies were evaluated for their applicability in deriving long-term chronic and short-term acute WQGs. The Ni WQG is calculated using a Biotic Ligand Model (BLM), which incorporates specific water chemistry data using specialized software. The BC Ni BLM software and BC Ni BLM User's Manual accompany this technical report.

2. PHYSICAL AND CHEMICAL PROPERTIES OF NICKEL

Nickel (Ni; atomic number 28, atomic mass 58.7) is a hard, silvery-white, shiny metal that is highly ductile and heat resistant (melting point of 1453°C). It can form alloys with iron, copper, chromium, and zinc. Nickel can exist in the oxidation states 0, -1, +1, +2, +3 and +4, but the Ni^{2+} is the dominant form in the environment, including in aquatic systems under typical pH conditions (pH 5 to 9; Pyle and Couture 2011). In natural freshwaters, more than 99% of dissolved Ni is bound to organic ligands, with very low free ionic Ni^{2+} concentrations (in the 10 - 15 mol/L range; Xue et al. 2001). Under toxic conditions, Ni is either bound to dissolved organic matter (DOM) or tends to adsorb to iron oxyhydroxides [$Fe(OH)_3$], especially at higher pH, and to manganese oxyhydroxides (MnO_2) and montmorillonite at low ionic strength. Under lower pH conditions, there is a competition between H^+ and Ni^{2+} that causes dissociation of Ni from hydrous oxides (Green-Pedersen et al. 1997). Ni may also form inorganic complexes with the following anions in order of decreasing affinity: $OH^- < SO_4^{2-} < Cl^- < NH_3$ (Richter and Theis 1980). Ni forms complexes with carbonates such as $NiCO_3$, however the Ni carbonate species are less frequent in natural systems than for other metals such as copper and lead (Richter and Theis 1980). Under anoxic conditions, Ni forms insoluble sulfides (Pyle and Couture 2011).

3. INDUSTRIAL AND ECONOMICAL IMPORTANCE OF NICKEL

Nickel's physicochemical characteristics such as strength, high-temperature stability, corrosion resistance, malleability, and heat and electrical conductive properties make this metal and its compounds highly desirable for the production of many common products (Pyle and Couture 2011). Globally, the majority of Ni is used to make stainless steel (71% of Ni uses), followed by alloys (14%), electroplating (6%), batteries (4%), casting (3%), and other uses (Natural Resources Canada 2021). Ni is a component of nickel-cadmium batteries and is sometimes used in lithium-ion batteries, which are used in hybrid and electric vehicles. Canada was the fifth largest producer of Ni in the world in 2019 (6.8% of the total), with exports (including Ni and Ni-based products) valued at $4.1 billion (Natural Resources Canada 2021). Canada's Ni mines are in Manitoba, Ontario, Quebec, and Newfoundland and Labrador, while three refineries exist in Fort Saskatchewan, Alberta; Sudbury, Ontario; and Long Harbour, Newfoundland and Labrador (Natural Resources Canada 2021). The U.S. Geological Survey estimated that 2.7 million tonnes of Ni were mined globally in 2019, while the world reserves were approximately 89 million tonnes (Natural Resources Canada 2021). The average monthly price of Ni has fluctuated since 2010, with a peak of US$22,910 per tonne in 2011 and a low of US$9,595 per tonne in 2016. In 2019 the price had increased to US$13,914 per tonne (Natural Resources Canada 2021).

4. ENVIRONMENTAL FATE AND TRANSPORT OF NICKEL

Ni is the 24th most abundant element on earth, with an average concentration of 0.0086%, or 75 µg/g in the earth's crust (Chau and Kulikovsky-Cordeiro 1995, ATSDR 2005). It is found in ultrabasic igneous rocks ranging in content from 0.016% in basalt to 0.20% in periodotite (Birge and Black 1980). In Canada, it is commonly found in sulfide ores (Cornwall 1966). Nickel concentrations in soils are measured in the range of 4 to 80 mg/kg (ATSDR 2005). Nickel is released to the atmosphere by windblown dust, oil-, coal-, or sewage sludge-burning power plants, industries involved in Ni refining, steel or alloy production,

municipal incineration, and volcanoes (ATSDR 2005). Forest fires can also cause an intense, albeit short-term, increase of Ni in the atmosphere (Chau and Kulikovsky-Cordeiro 1995). Ni will enter aquatic systems via atmospheric deposition, surface runoff, or industrial or municipal effluents (Chau and Kulikovsky-Cordeiro 1995).

In natural waters, Ni is relatively mobile. Redox potential, pH, ionic strength, and chemical composition of the water (including the type and concentration of organic and inorganic matter and solid particulates) will affect the fate, transport, and bioavailability of Ni. In aerobic conditions, Ni can partition into the sediment after forming nickel ferrite, a stable compound, and also through coprecipitation with hydrous iron and manganese oxides (ATSDR 2005). In the sediment, Ni can be strongly bound to particles, but can also be remobilized due to microbial action, especially in anaerobic conditions or under low pH (ATSDR 2005). Ni does not accumulate significantly in aquatic organisms such as fish and mollusks (Birge and Black 1980, McGeer et al. 2003).

The natural concentrations of Ni in freshwater ecosystems in Canada are generally in the 0.1 to 10 µg/L range, although concentrations can go up to 10,000 µg/L in areas naturally enriched with Ni. Further information on background levels of Ni in B.C. waters is presented in Section 6. Ni concentrations typically range between 50 and 2,000 µg/L in waters affected by industrial discharges (Chau and Kulikovsky-Cordeiro 1995).

5. ANALYSIS OF NICKEL IN ENVIRONMENTAL SAMPLES

Both dissolved and total Ni can be analysed in water samples. Dissolved Ni analysis typically refers to the fraction that passes through a 0.45 µm filter, while total Ni analysis includes the dissolved fraction plus any Ni associated with particulate material (e.g., suspended sediments).

The B.C. Environmental Laboratory Manual (ENV 2020) describes four common laboratory techniques for the measurement of Ni in environmental samples:

- Atomic Absorption Spectroscopy (AAS);
- Graphite Furnace Atomic Absorption Spectrometry (GFAAS);
- Inductively Coupled Plasma Atomic Emission Spectrometry (ICP-AES); and,
- Inductively Coupled Plasma Mass Spectrometry (ICP-MS).

Sample preparation, in particular the type of acid digestion conducted, can modify the Ni detection limit. The B.C. Environmental Laboratory Manual (ENV 2020) recommends digesting total metals (including Ni) in a mixture of nitric and hydrochloric acids. Samples for dissolved Ni analysis should be filtered in the field using a 0.45 µm filter and then preserved with nitric acid (ENV 2020). The typical method detection limit (MDL) for Ni using AAS (direct aspiration method) is 40 µg/L, while it is 1 µg/L for GFAAS, and 10-30 µg/L for ICP-AES in the absence of interferences (ATSDR 2005, ENV 2020). Detection limits for Ni with ICP-MS instruments typically range from 0.03 to 0.5 µg/L EPA 1994). The Practical Quantitation Limit (PQL), defined as “the lowest achievable level of analytical quantitation during routine laboratory operating conditions within specified limits of precision and accuracy” (U.S. EPA 1985), is in practice about 5-10 times higher than the MDL (U.S. EPA 1985). Therefore, MDLs should be, at a minimum, five times below the ambient WQG to ensure a high level of precision and accuracy in the laboratory results. In cases where laboratories have defined PQLs for the substance of interest, it is recommended that the PQL be at or below the ambient WQG.

6. BACKGROUND CONCENTRATIONS OF NICKEL IN BRITISH COLUMBIA WATERS

Nickel is naturally present in both aquatic and terrestrial ecosystems; hence, background concentrations must be considered in the derivation of the provincial Ni water quality guideline (ENV 2019a).

6.1 Methods for Estimating Background Concentrations

Data to characterize background Ni concentrations in B.C. were taken from two sources: the B.C. Environmental Management System (EMS) database and the Canadian Aquatic Biomonitoring Network (CABIN) database. To do this, water quality sampling stations that were sampled at least three times over the last 20 years for any water quality parameter (2001/01/01 to 2021/11/02) were extracted. Since EMS does not identify reference stations, the database had to be screened to create a sub-set of water quality stations known to be minimally impacted. For this purpose, the list of reference stations prepared by ENV environmental impact assessment biologists to identify sites that were considered minimally impacted by human activities was used.

The resultant data set was augmented with samples collected by ENV and Environment and Climate Change Canada (ECCC) at B.C. reference stations as part of the CABIN program. CABIN reference stations are located on stream reaches minimally impacted by anthropogenic activities and are generally sampled once during the late summer/early fall low flow period. Only total Ni values were available in the CABIN database. The combined EMS and CABIN dataset had 627 minimally impacted stations with data for dissolved and/or total Ni (Figure 6.1).

The dataset underwent several additional automated and manual data cleaning steps summarized below:

- For lake samples, if surficial depth was not provided, we considered it equal to the lower depth, unless the lower depth was also not provided, in which case both depths were treated as 0 m;
- For lake samples, if discrete samples were collected from multiple depths, only samples from the surface (upper depth < 10 m) were included;
- For lake samples, if integrated samples were taken at multiple integrated depths (e.g., 0–5 m and 5–15 m), only the most surficial integrated sample was included;
- If the same data were included in both the CABIN and EMS databases for the same site on the same day, EMS data were discarded and only CABIN data were kept for that site and that day;
- Non-detect results with an MDL of 5 µg/L or higher were removed as these would influence the results of the analysis;
- Samples were excluded where results were missing or reported as 0; and,
- Data were visually inspected, and samples were removed where results appeared to be obvious errors, assumed to be attributed to either data entry, or analytical errors.

Arithmetic means were calculated for laboratory replicates (analytical replicates taken from one field sample), with MDL substituted for values below detection. All field replicates were included as independent samples.

The results from each station were given equal weight within a Natural Resource Region by calculating the mean Ni concentrations for each station. Station means were calculated using four different approaches depending on the number of samples above (detects) and below (non-detects) the MDL (Table 6.1). A value of ½ the minimum MDL was used to represent station means when all samples were below the MDL (Group 1). The minimum MDL was chosen to account for decreasing MDLs over time. For stations with less than three detects, ½ of the MDL was substituted for non-detect values and the arithmetic mean of all station results was calculated (Group 2). Regression on order statistics (ROS) was used to calculate an estimate of the mean for stations that had a mixture of non-detects and detects with

at least three detected values (Huston and Juarez-Colunga 2009; Group 3). Although Huston and Juarez-Colunga (2009) state that ROS can be used on sample sizes >0, a minimum of three detects is required to calculate a valid regression using the NADA package (Lee 2017) in R (R Core Team 2022). The arithmetic mean was calculated for stations where all samples were above the MDL (Group 4). Statistics to summarize the distribution of station means (median, the 10th and 90th percentile) were calculated for each Natural Resource region. All analyses were conducted in R (R Core Team 2022).

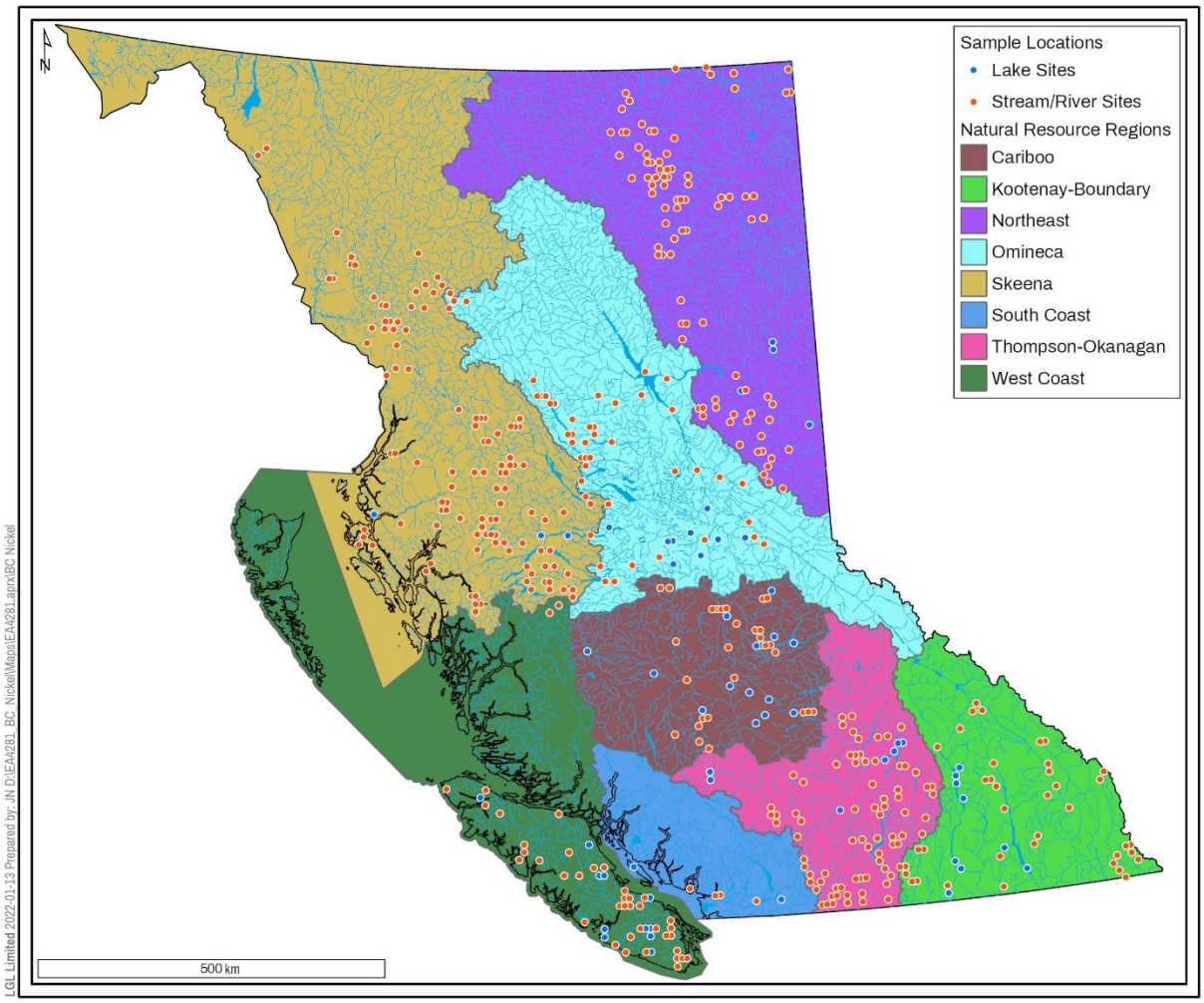

Figure 6.1. Sampling stations in each B.C. Natural Resource Region with dissolved or total nickel data.

Table 6.1. Statistical approach used to calculate station means.

Group	Conditions	Approach	Dissolved Nickel		Total Nickel	
			Total Stations	Total Samples	Total Stations	Total Samples
1	% non-detects = 100	½ of minimum station MDL	8	57	20	80
2	0 < % non-detects < 100 AND # detects < 3	Substitute ½ MDL for non-detects and calculate arithmetic mean for all samples	25	229	21	198
3	0 < % non-detects < 100 AND # detects ≥ 3	Regression on order statistics	67	2,577	94	4,615
4	% non-detects = 0	Arithmetic mean	67	577	491	1,878

6.2 Background Concentration Results

Concentrations of dissolved and total Ni were analyzed separately, and the results are given below. Dissolved Ni is the more bioavailable fraction (see section 8). However, total Ni concentrations were also examined as more total Ni samples have been collected across the province and therefore, they provide a better geographical representation. A plot of total vs. dissolved paired Ni samples revealed that 18% of paired samples had dissolved values that were higher than total values (Figure 6.2). This may be the result of lower analytical precision at very low concentrations. Given the variability in the total compared to dissolved Ni concentrations, no attempt was made to estimate the proportion of dissolved Ni under field conditions, nor were total and dissolved estimates directly compared. Regional summaries were used to minimize the effect of individual sampling errors.

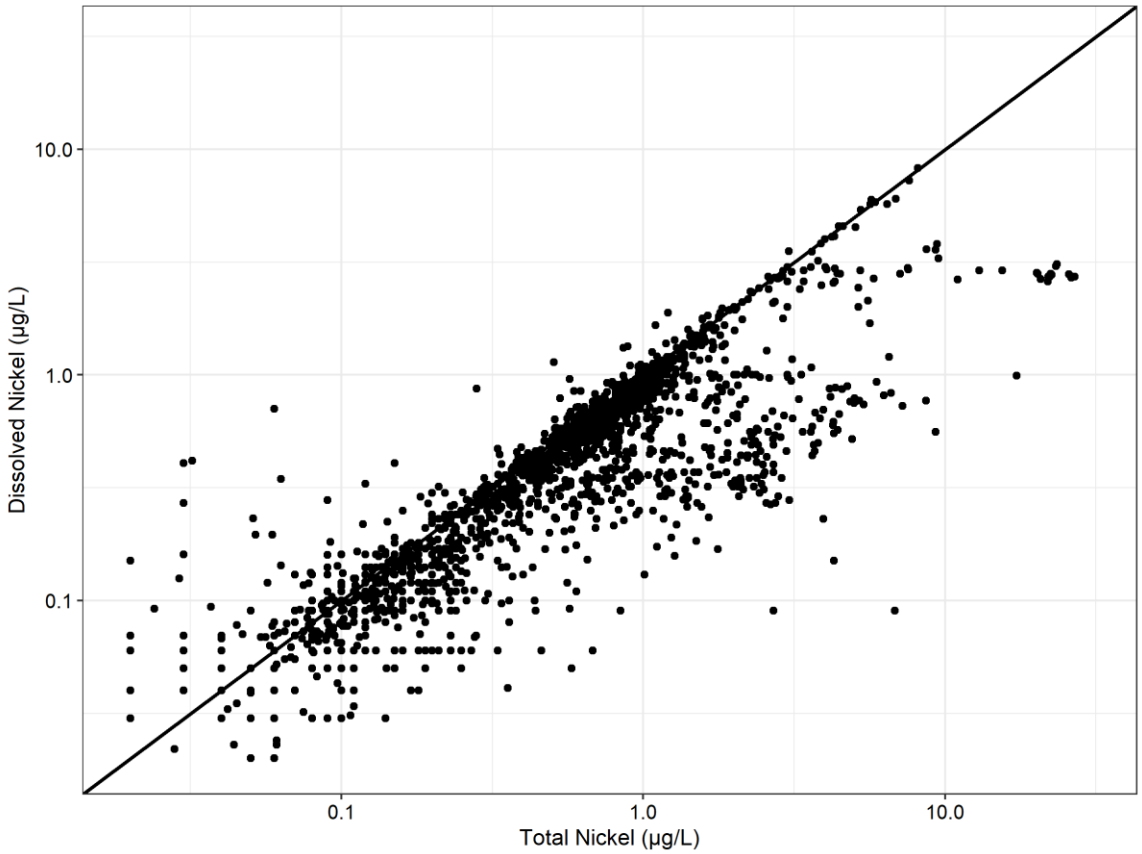

Figure 6.2. Total vs. dissolved Ni from paired field samples where both total and dissolved Ni were above detection. Solid line is 1:1. n = 1,924.

6.2.1 *Dissolved Nickel*

Dissolved Ni data were summarized by Natural Resource Region in Table 6.2 and Figure 6.3. There were 167 stations with dissolved Ni data, which included 3,440 samples. The number of stations within each region with dissolved Ni data ranged from 3 (South Coast) to 61 (Cariboo). Three regions had less than 10 stations (Omineca, Northeast, South Coast), making the characterization of dissolved Ni difficult. The Cariboo, West Coast, and Skeena regions had the most stations with dissolved Ni data (n = 61, 43, and 27, respectively). The median of station means in each region was <1 µg/L, except for the Northeast, with a median of 1.86 µg/L.

The median dissolved Ni concentration across the province, based on station means, was 0.288 µg/L. This was about five times higher than the lowest regional median (0.06 µg/L; South Coast) and six times lower than the highest regional median (1.86 µg/L; Northeast). The 90th percentile of station means was 1.23 µg/L, which was higher than the 90th percentile for the individual regions, except for Omineca, Northeast, and Cariboo (3.11, 1.98, and 1.40 µg/L, respectively). Regionally, Northeast, Omineca, Cariboo, and Kootenay-Boundary had a higher median of station means (but not always a higher 90th percentile) than other regions. Sampling at more stations is needed to better characterize dissolved Ni concentrations across the province, but especially in the Omineca, Northeast, South Coast, and Kootenay-Boundary Regions, as well as in the northern part of Skeena Region and in Haida Gwaii and the Central Coast (within the West Coast Region).

Of the 167 stations with data for dissolved Ni, 61 stations were on lakes and 106 were on rivers. The median of the distribution of station means was slightly lower for rivers (0.254 µg/L) than for lakes (0.314 µg/L); however, the range of concentrations were relatively consistent (see Figure 6.4).

There is uncertainty in the results of some of the station means calculated using ROS where >80% of the data at a station were non-detects. The number of stations where >80% of the data were non-detects was highest in the Cariboo Region (8 stations), followed by Thompson-Okanagan (6 stations), Kootenay-Boundary (5 stations), and Skeena (3 stations). Only two stations had >80% non-detects in the South Coast Region, while no stations had this many non-detects in the West Coast, Northeast, and Omineca regions. The regions with the highest proportion of non-detect samples were Kootenay-Boundary (80.4%), Thompson-Okanagan (72.6%), and Omineca (44.7%).

Table 6.2. Summary statistics for dissolved nickel concentrations at selected minimally impacted stations in British Columbia by region.

Region	Number of Stations	Number of Samples	Date Range	Concentration Range Across all Samples (µg/L)	MDL Range Across all Samples (µg/L)	% Samples < MDL	Distribution of Station Means (µg/L)		
							Median	10th Percentile	90th Percentile
Cariboo	61	1,471	2001 – 2021	0.02 – 25.4	0.02 – 1	29.3	0.516	0.191	1.40
Kootenay-Boundary	10	179	2003 – 2021	0.125 – 1.1	0.05 – 1	80.4	0.401	0.130	0.469
Northeast	4	55	2012 – 2018	0.032 – 1.98	ND	0	1.86	0.746	1.98
Omineca	8	38	2005 – 2018	0.314 – 6.04	1	44.7	0.818	0.539	3.11
Skeena	27	380	2001 – 2021	0.02 – 13.7	0.02 – 1	25.5	0.108	0.0397	0.581
South Coast	3	21	2002 – 2005	0.05 – 0.33	0.05 – 0.05	33.3	0.0600	0.0320	0.143
Thompson-Okanagan	11	731	2001 – 2021	0.05 – 5.72	0.05 – 3	72.6	0.288	0.0250	0.643
West Coast	43	565	2001 – 2021	0.02 – 1.76	0.02 – 0.4	28.7	0.095	0.0427	0.355

ND = no data; MDL = method detection limit.

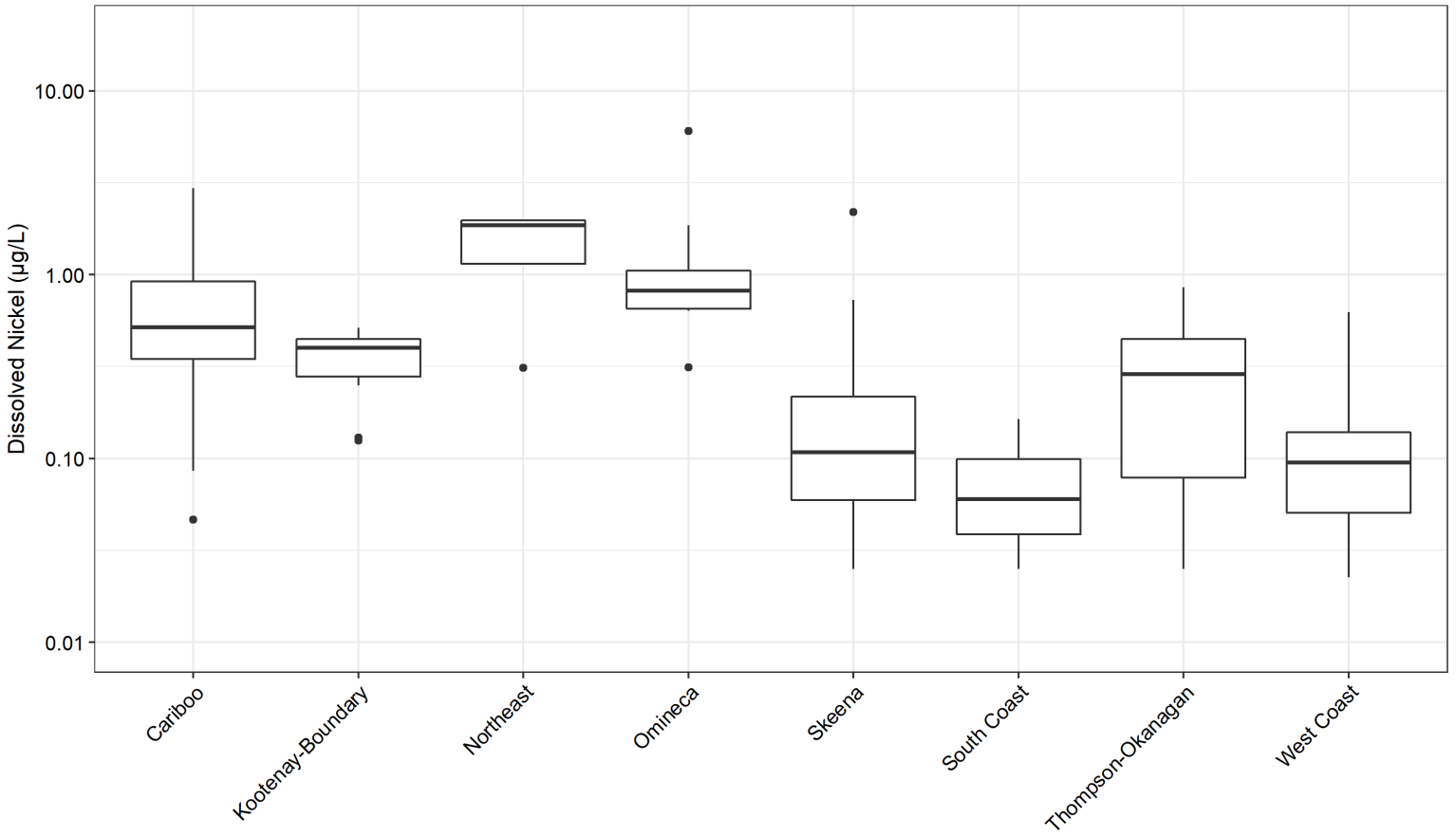

Figure 6.3. Distribution of station mean dissolved nickel concentrations at selected minimally impacted stations in British Columbia by region. Note y-axis is on a log scale. Solid horizontal bar within each box represents median of station means.

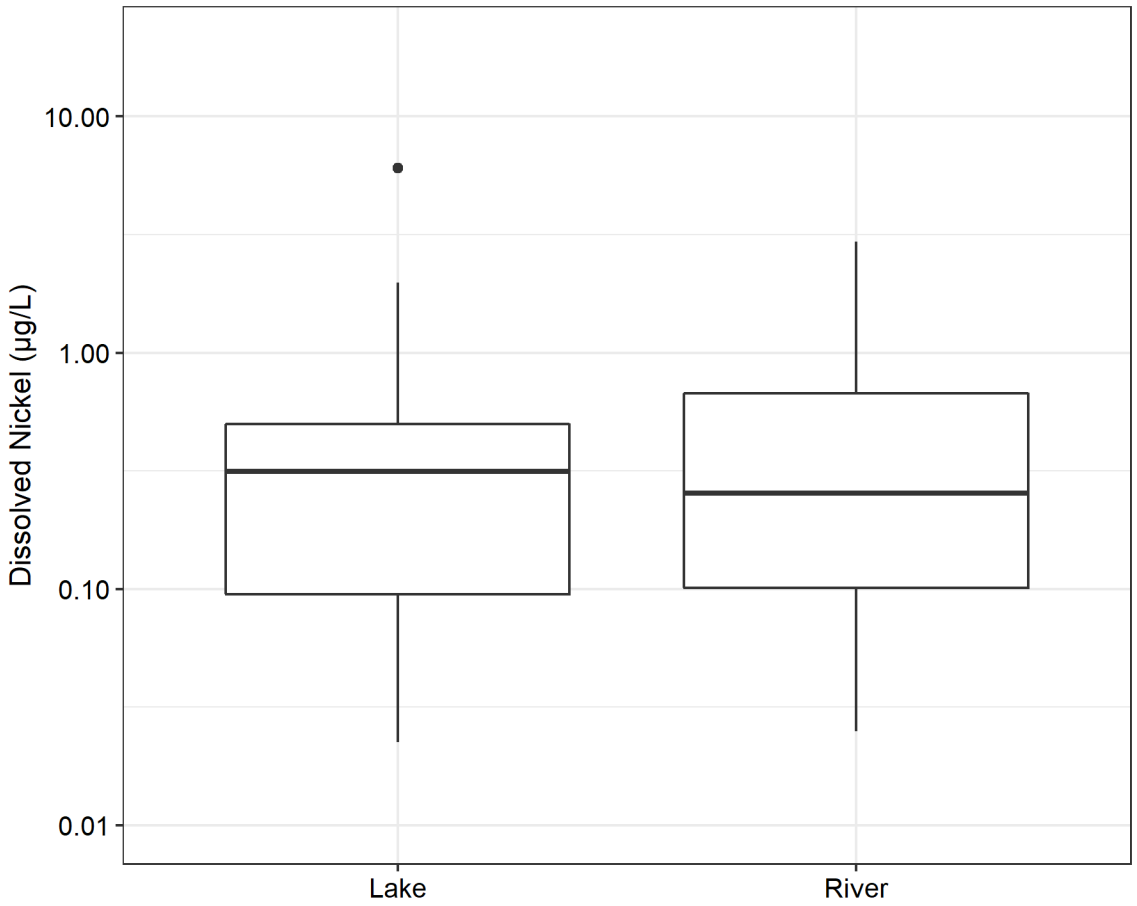

Figure 6.4. Distribution of station means for lakes and rivers for dissolved nickel.
Note y-axis is on a log scale.

6.2.1 *Total Nickel*

Total Ni data were summarized by ENV Natural Resource Region in Table 6.3 and Figure 6.5. There were 626 stations with total Ni data, which included 6,771 samples. All regions had more than 50 stations except the South Coast, which had 7 stations. The median of station means in each region was <1 µg/L, except for the Northeast, with a median of 1.91 µg/L. The median total Ni across the province was calculated based on station means and was 0.289 µg/L. This was about three times higher than the lowest regional median (0.102 µg/L; West Coast) and six times lower than the highest regional median (1.91 µg/L; Northeast). The 90th percentile of station means across the province was 2.78 µg/L, which was higher than the 90th percentile for the individual regions, except for Omineca and Northeast (5.58 and 5.18 µg/L, respectively). Regionally, Northeast, Omineca, and Cariboo had a higher median of station means (but not always a higher 90th percentile) compared with other regions.

Of the 626 stations with total Ni data, 75 stations were on lakes and 551 were on rivers. For total Ni, the median of the distribution of station means in rivers (0.270 µg/L) was slightly lower than that of lakes (0.349 µg/L) (see Figure 6.6).

There is uncertainty in the results of some of the station means calculated using ROS where >80% of the data at a station were non-detects. The number of stations where >80% of the data were non-detects was highest in the Thompson-Okanagan Region (13 stations), followed by Skeena (7 stations), Cariboo (6 stations), and Kootenay-Boundary (5 stations). Only one station had >80% non-detects in the South Coast and West Coast regions, while no stations had this many non-detects in the Northeast and Omineca regions. The regions with the highest proportion of non-detect samples were Thompson-Okanagan (52.4%) and Kootenay-Boundary (48.4%).

Table 6.3. Summary statistics for total nickel concentrations at selected minimally impacted stations in British Columbia by region.

Region	Number of Stations	Number of Samples	Date Range	Concentration Range Across all Samples (µg/L)	MDL Range Across all Samples (µg/L)	% Samples < MDL	Distribution of Station Means (µg/L)		
							Median	10th Percentile	90th Percentile
Cariboo	64	2,283	2001 – 2021	0.02 – 44.3	0.02 – 1	19.4	0.703	0.251	1.60
Kootenay-Boundary	55	312	2001 – 2021	0.05 – 5	0.05 – 1	48.4	0.144	0.062	0.552
Northeast	109	217	2008 – 2021	0.01 – 49.9	0.01 – 0.02	0.461	1.91	0.508	5.18
Omineca	59	687	2001 – 2021	0.02 – 47	0.02 – 0.5	1.31	0.764	0.0822	5.58
Skeena	144	1,329	2001 – 2021	0.02 – 40.2	0.02 – 1	8.58	0.170	0.0500	1.26
South Coast	7	58	2002 – 2021	0.05 – 9.61	0.05 – 1	31	0.172	0.0459	2.39
Thompson-Okanagan	108	1,144	2001 – 2021	0.05 – 20	0.05 – 3	52.4	0.192	0.0454	0.917
West Coast	80	741	2001 – 2021	0.02 – 2.28	0.02 – 0.4	26.5	0.102	0.0407	0.284

MDL = method detection limit.

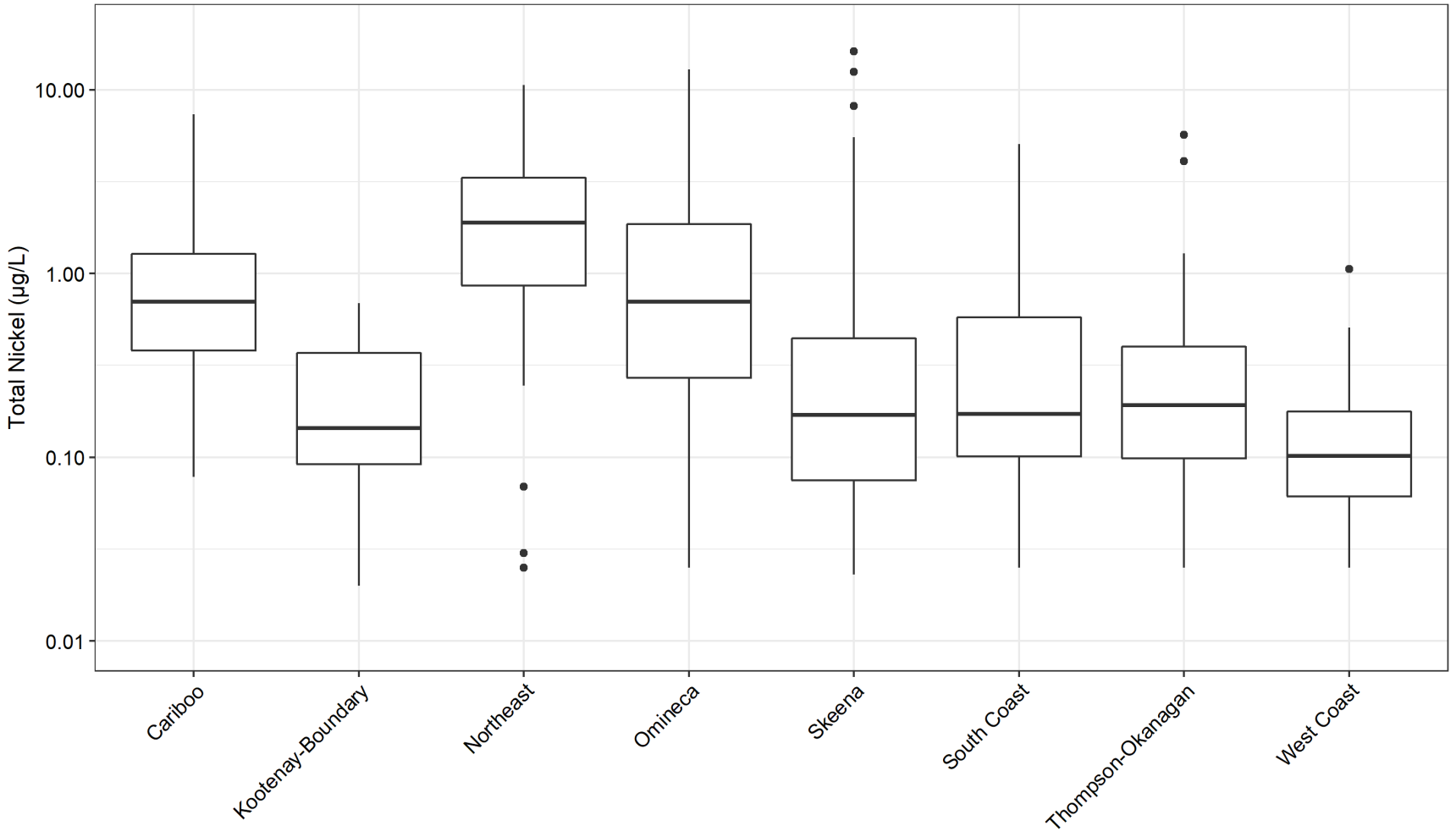

Figure 6.5. Distribution of station mean total nickel concentrations at selected minimally impacted stations in British Columbia by region. Note y-axis is on a log scale. Solid horizontal bar within each box represents median of station means.

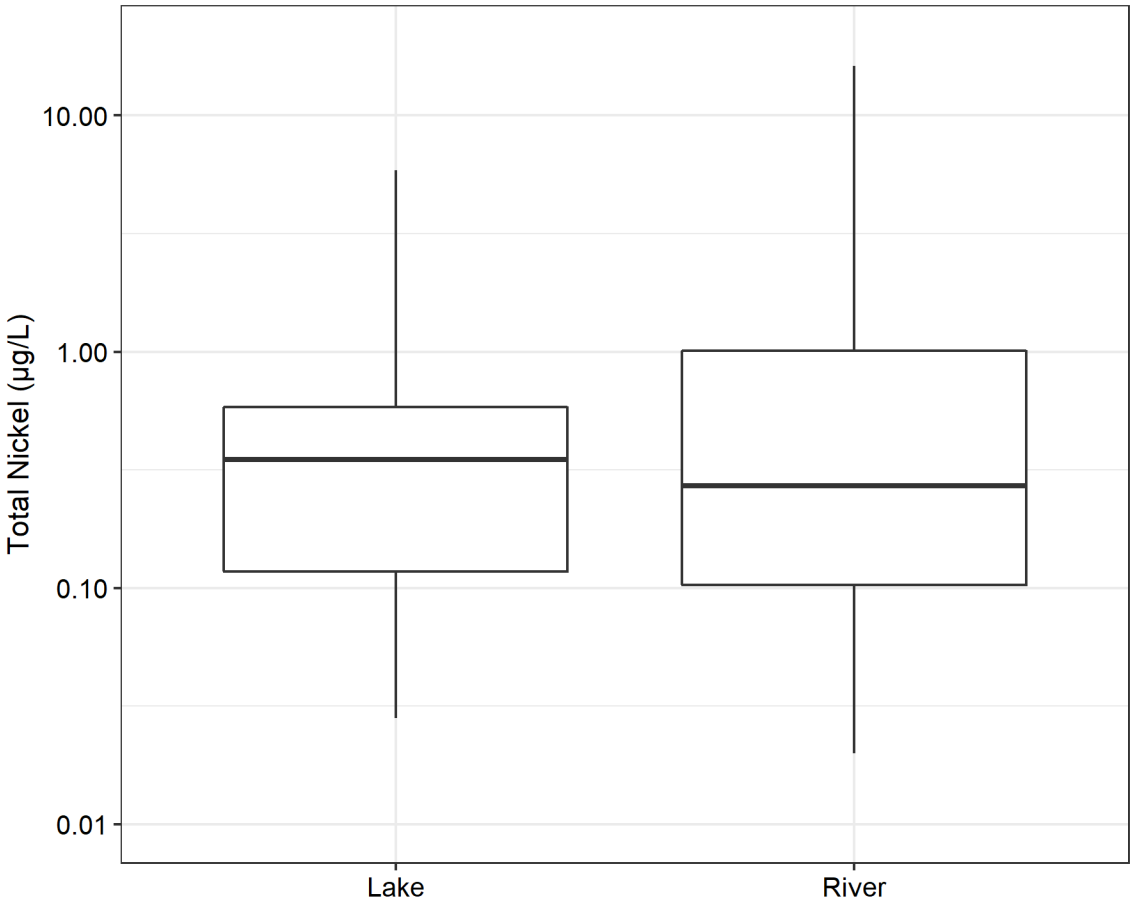

Figure 6.6. Distribution of station means for lakes and rivers for total nickel.
Note y-axis is on a log scale.

7. ESSENTIALITY

Nickel is an essential nutrient for microorganisms, algae, and aquatic plants, acting as a co-factor in Ni-dependent enzymes involved in nitrogen fixation and metabolism, and carbon cycling (Muyssen et al. 2004, Ragsdale 2011, DalCorso 2012). For example, Ni plays a role in cyanobacterial urease and hydrogenase metabolism (Ragsdale 1998). Ni is also considered a possible essential metal in higher vertebrates, based on symptoms observed in animals fed with a Ni-deficient diet (Phipps et al. 2002). In addition, homeostatic regulation of uptake and excretion of Ni, storage of excess Ni in hair, fur or feathers, and binding of Ni to metallothioneins are presented as signs of Ni essentiality in terrestrial animals (Phipps et al. 2002). Some studies are also showing that Ni could be essential in fish due to observed homeostatic regulation (Muyssen et al. 2004, Chowdhury et al. 2008). For instance, Rainbow Trout (*Oncorhynchus mykiss*) and Fathead Minnow (*Pimephales promelas*) can keep relatively constant Ni concentration in their bodies or organs (e.g., gill, kidney, liver) despite being fed different concentrations of Ni-enriched diets (Chowdhury et al. 2008, Lapointe and Couture 2010). Similarly, the inverse relationship observed between the bioconcentration factor (BCF) and exposure concentration for Ni may indicate active regulation of Ni by crustaceans, suggesting Ni is essential to crustaceans. However, further research is needed to determine whether Ni is essential to aquatic invertebrates (Muyssen et al. 2004).

8. NICKEL TOXICITY-MODIFYING FACTORS

The toxicological risk of several metals (e.g., copper, cadmium, and nickel) is highly dependent on their bioavailability to aquatic organisms. Bioavailability, in aquatic toxicology, is defined as the rate and extent to which a substance can reach its site of action (Adams et al. 2020). The factors that can reduce metal bioavailability, and by extension its toxicity, are often related to changes in solubility, sorption to solid or humic substances, or partition to inaccessible phases such as minerals. The presence of elements that can compete for binding sites, such as Ca and Mg, is often the most important toxicity-modifying factor (TMF) for divalent metals (Adams et al. 2020).

In the case of Ni, hardness, pH, and DOC act as TMF as they reduce toxicity to fish and aquatic invertebrates in both acute and chronic studies (Santore et al. 2021). The TMF effects of these factors are discussed in sections 8.1 to 8.3.

8.1 Hardness

In general, Ni toxicity decreases as hardness (i.e., Ca and Mg ion concentrations) increases (Deleebeeck et al. 2009a, Schlekat et al. 2010). These ions might reduce the membrane permeability to Ni, hence reducing its uptake and toxicity. Calcium and magnesium both have a protective effect on rainbow trout, increasing by about 4-fold the LC_{50} between 0.12 and 1.0 mM Ca, and 0.12 and 2.0 mg Mg, respectively (Deleebeeck et al. 2007a). In acute tests conducted on *Ceriodaphnia dubia*, 48-h LC_{50} estimates increased from 81 to 400 µg/L as water hardness was increased from 50 to 253 mg/L $CaCO_3$ (Keithly et al. 2004). However, only an increase in Mg, not Ca, was found to decrease the toxicity of Ni to *Pseudokirchneriella subcapitata* (Deleebeeck et al. 2009b).

The four Ca^{2+} channel blockers tested by Niyogi et al. (2014) did not have an effect on the Ni uptake rate of the great pond snail (*Lymnaea stagnalis*), so it appears that Ni does not directly compete with Ca^{2+} at the uptake pathways. However, this does not rule out non-competitive inhibition of parts of the Ca^{2+} uptake pathways by Ni (Niyogi et al. 2014). Similarly, Gopalapillai et al. (2013) concluded that major cations do not act by competitive inhibition to reduce Ni toxicity to the duckweed *Lemna minor*. Interestingly, Gopalapillai et al. (2013) found that Ca^{2+} did not consistently decrease the toxicity of Ni to *L. minor*. They ran 7-day tests assessing *L. minor* growth using three different calcium salts: $CaSO_4$, $Ca(NO_3)_2$, and $CaCl_2$. Whether Ca^{2+} affected root growth inhibition by Ni depended on the identity of the counter-anion. $CaCl_2$ decreased Ni toxicity, $Ca(NO_3)_2$ did not affect toxicity, and $CaSO_4$ increased toxicity (Gopalapillai et al. 2013). However, it appeared that Ca and the anions affected *L. minor*'s physiological response to Ni but not the uptake of Ni, suggesting that Ca^{2+} does not competitively inhibit the uptake of Ni (Gopalapillai et al. 2013).

In fish, as accumulation of Ni on the gills is the main driver of acute toxicity, the presence of Ca ions decreases the toxic potential of Ni by saturating the binding sites on the gill surface. For instance, fathead minnow EC_{50} estimates increased by 10 times when hardness was increased from 23.5 to 249.4 mg/L $CaCO_3$ (Meyer et al. 1999). A protective effect from Ca from 5 to 100 mg/L was also observed in fathead minnow in the study of Hoang et al. (2004).

8.2 Dissolved Organic Carbon

Dissolved organic carbon, in the form of natural organic matter (NOM), reduced the toxicity of Ni to *D. pulex* and *H. azteca* in tests conducted with two natural sources of organic matter (Kozlova et al. 2009). DOC binds to Ni^{2+} ions, decreasing the bioavailability of Ni and therefore decreasing its toxicity (Deleebeeck et al. 2008a, Schlekat et al. 2010). One study however showed that increasing water hardness, measured as higher concentrations of Ca^{2+} and Mg^{2+}, caused the dissociation of DOC-bound Ni,

increasing the free ion Ni^{2+} in water, which is more toxic to microalgae such as *P. subcapitata* (Mandal et al. 2002).

DOC has been found to decrease acute Ni toxicity by up to 50% in juvenile fathead minnow when DOC was increased from 0.5 mg/L to 5 mg/L, after which the protective effect plateaued (Hoang et al. 2004). The protective effect of DOC is expected to be important for more sensitive organisms or sensitive life stages (Santore et al. 2021). For instance, the reduction of Ni toxicity with NOM was less important for *H. azteca* than for *D. pulex*, even though the range of DOC used (0 - 30 mg/L) was similar in both studies (Doig and Liber 2006, Kozlova et al. 2009). Doig and Liber (2006) also found that the concentration of DOC was more important than the NOM source or fraction (e.g., whole peat, peat fulvic acid, Suwannee River humic acid) when tested on *H. azteca* (the more Ni-tolerant organism) exposed to sublethal concentrations of Ni. In contrast, when using *D. pulex* as test organism, Suwannee River NOM (of 0.5 to 41 mg/L) reduced by 3-fold the toxicity of Ni, while a smaller range of Nordic Reservoir NOM (1.5 to 35 mg/L) reduced Ni toxicity by 5 times (Kozlova et al. 2009). In that latter study, the source of NOM was important. De Schamphelaere et al. (2006) found that increased DOC decreased both acute and chronic toxicity of Ni to *C. dubia*.

In contrast, higher concentrations of DOC did not have a consistent protective effect on the toxicity of Ni to the green microalgae *Chlorella* sp. which was in part dependent upon the source of the DOC (Macoustra et al. 2021). In tests using DOC from Appletree Creek and Crystal Cascades, the high DOC treatment (12 mg/L or 14 mg/L, respectively) significantly decreased Ni toxicity to *Chlorella* sp. by two times or less compared to the control with no DOC added, but the high DOC treatment using Lake Eacham water (12 mg/L) had a similar 72-hour EC_{50} for growth (130 µg/L) to the control (120 µg/L). However, the low DOC treatment using Appletree Creek or Lake Eacham DOC (3.3 mg/L or 1.9 mg/L, respectively) had significantly higher toxicity than the control, while the Crystal Cascades low DOC treatment (5.2 mg/L) resulted in a similar 72-hour EC_{50} for growth (140 µg/L) as the control (Macoustra et al. 2021). In another study, Peters et al. (2018) found that toxicity of Ni to a different strain of *Chlorella* sp. decreased as DOC concentration increased from 4.5 to 11 mg/L (72-hour EC_{50} estimates for growth ranged from 282 µg/L to 2,410 µg/L), though the 2.2 mg/L DOC concentration resulted in an EC_{50} of 434 µg/L.

8.3 pH

In general, Ni toxicity is reduced as pH decreases (Schlekat et al. 2010). Deleebeeck et al. (2009b) found that the 72-hour EC_{50} for dissolved Ni increased from 81.5 to 145 µg/L as pH decreased from 7.92 to 6.45 in tests conducted with the green alga *P. subcapitata*. The study of Deleebeeck et al. (2007a) showed that a pH increase from 5.5 to 8.5 led to a decrease of the 17-day LC_{50} for rainbow trout from 2,440 µg/L to 558 µg/L. The authors described it as a potential protective effect from H^+ ions against Ni toxicity. On larval fathead minnows, Ni was more toxic at the highest pH (pH 8 – 8.5, 96-hour LC_{50} of 3,100 µg/L) than at low pH (pH 6 – 6.5, 96-hour LC_{50} > 4,000 µg/L). Nys et al. (2016) found the toxicity of Ni to *Daphnia magna,* great pond snail (*L. stagnalis*), *Brachionus calyciflorus*, and *P. subcapitata* to be higher (i.e., lower effect concentration) at pH 8.7 than pH 8.2, though effect concentrations for *L. minor* were not significantly different between these two pH values. However, when the effect concentrations were expressed as Ni^{2+} activity, the EC_{10} and EC_{20} values for *L. minor* were significantly lower at pH 8.7 than pH 8.2, and the differences in effect concentrations between pH scenarios for the other species were larger than before (Nys et al. 2016). Increasing pH also increased the Ni toxicity to *C. dubia, Hyalella azteca*, and *Lumbriculus variegatus* (Schubauer-Berigan et al. 1993).

Different explanations for increased toxicity of Ni at high pH values were suggested by different studies. Puttaswamy and Liber (2012) studied the effect of bicarbonate (HCO_3^-) on the chronic toxicity of Ni to *C. dubia* and found that reproduction IC_{50} estimates decreased (i.e., greater toxicity) at high levels of

bicarbonate. They suggested that the $NiCO_3$ complex could be more bioavailable to daphnids; however, this increase in toxicity could also be due to higher pH and hence less competition of Ni with H^+. Santore et al. (2021) argued that this effect is only observed in *C. dubia*, and that bicarbonate itself can cause toxic effects on *C. dubia* reproduction. Further studies are needed to determine if the presence of bicarbonate ions is increasing the toxicity of nickel to aquatic invertebrates.

9. TOXICITY OF NICKEL TO FRESHWATER AQUATIC ORGANISMS

Despite its essentiality, Ni will cause toxicity at elevated concentrations from acute and chronic exposure. Like several other metals, the free ion is considered the most toxic chemical species (e.g., Macoustra et al. 2021). While in fish, the mechanism of Ni toxicity is ascribed to be respiratory impairment (Pane et al. 2003a, 2004a), in aquatic invertebrates such as *D. magna*, Ni is hypothesized to act as an ion regulatory toxicant by disrupting Mg and Na homeostasis (Pane et al. 2003b, Leonard and Wood 2013). In fact, Ni and Mg share uptake transporters, hence the Mg depletion seen in organisms exposed to high concentrations of Ni (Pane et al. 2003b). It is believed that Ni not only interacts with Mg and Na, but also with Ca in aquatic and terrestrial organisms (e.g., Pane et al. 2006, Deleebeeck et al. 2009b, Niyogi et al. 2014). Niyogi et al. (2014) found that Ni disrupted Ca^{2+} and Mg^{2+} homeostasis in the freshwater pulmonate snail *L. stagnalis.* Interaction between Ni and the divalent metal transporter DMT-1, involved in iron absorption, has been found in the guts of the rainbow trout, leading to iron deficiency in exposed fish (Kwong and Niyogi 2009). In plants, high levels of Ni inhibit root growth, disrupt the uptake of nutrients, and interfere with their water balance and photosynthesis (DalCorso 2012). There is also increasing evidence that Ni triggers the production of reactive oxygen species, which consequently leads to depletion of enzymatic and non-enzymatic antioxidants, lipid peroxidation, and DNA damages in tissues of exposed organisms (Zheng et al. 2014, Palermo et al. 2015). A recent review of the potential pathways by which Ni may exert toxicity on aquatic organisms was presented in Brix et al. (2017).

9.1 Effects on Algae

Nickel has been shown to negatively affect freshwater algae by decreasing growth and growth rate (Deleebeeck et al. 2009a 2009b, AECOM Technical Services, Inc. 2011, Martínez-Ruiz and Martínez-Jerónimo 2015, Nys et al. 2016, Nagai and De Schamphelaere 2016, Nys et al. 2017, Van Regenmortel and De Schamphelaere 2018, Santos et al. 2019, Macoustra et al. 2021), decreasing the concentration of photosynthetic pigments, decreasing cell protein, carbohydrate, and lipid concentrations, increasing the catalase and glutathione peroxidase (antioxidant enzymes) activity, and causing morphological and ultrastructural changes indicative of deformed and damaged cells (Martínez-Ruiz and Martínez-Jerónimo 2015). The Ni concentrations causing toxic effects in the green microalga *Ankistrodesmus falcatus* (4, 5, 8, and 17 µg/L, depending on the effect) were lower than the Canadian WQG (CCREM, 1987) for Ni at the test hardness of 15 mg/L $CaCO_3$ (i.e., 25 µg/L; Martínez-Ruiz and Martínez-Jerónimo 2015).

A. falcatus appears to be more sensitive than other green microalgae, with a 96-hour IC_{50} for growth of 17 µg/L (hardness = 15 mg/L $CaCO_3$, pH = 8.8, dissolved organic carbon [DOC] = 0.3 mg/L; Martínez-Ruiz and Martínez-Jerónimo 2015), compared to 72-hour EC_{50} estimates for growth ranging from 81.5 µg/L for *P. subcapitata* (hardness = 21 mg/L $CaCO_3$, pH = 7.9, DOC = 0.2 mg/L; Deleebeeck et al. 2009b) to 2,410 µg/L for *Chlorella* sp. (hardness = 32 mg/L $CaCO_3$, pH = 6.8, DOC = 11 mg/L; Peters et al. 2018). Deleebeeck et al. (2009a) reported that *P. subcapitata* appears to be more sensitive to Ni than other green microalgae they collected from the field (i.e., *A. falcatus, Chlamydomonas* sp., *Chlorella* sp., *Coelastrum microporum*, *Desmodesmus* sp., *Desmodesmus spinosus*, *Scenedesmus accuminatus*, and *Spermatozopsis exsultans*). Based on the literature compiled for this report, the diatom *Navicula pelliculosa* appears to be another sensitive algae species, with a 72-hour EC_{50} for growth of 55.2 µg/L (hardness = 24 mg/L $CaCO_3$, pH = 7,

DOC = 0.3 mg/L; Nagai and De Schamphelaere 2016). Nagai and De Schamphelaere (2016) reported that *P. subcapitata* seems to be more tolerant of Ni than *N. pelliculosa*.

9.2 Effects on Macrophytes

Fewer studies have looked at the effects of Ni on aquatic macrophytes than on algae. As part of a review summarizing different mechanisms of nickel toxicity, Brix et al. (2017) explained that Ni competitively inhibits the transport of iron from the root to the shoot of terrestrial plants, causing oxidative damage in roots via the Fenton reaction, and decreased chlorophyll production in shoots. However, this mode of action has not been elucidated in aquatic plants (Brix et al. 2017). Schlekat et al. (2010) evaluated the toxicity of Ni to duckweed (*L. minor*) in six different solutions of varying hardness, pH, and DOC. Effects varied greatly, with the 7-day EC_{10} for specific growth rate inhibition (based on root length) ranging from 3.9 µg/L (hardness = 163 mg/L $CaCO_3$, pH = 8.04, DOC = 1 mg/L) to 435.3 µg/L (hardness = 256 mg/L $CaCO_3$, pH = 6.9, DOC = 7 mg/L). The 7-day EC_{50} for *L. minor* specific growth rate inhibition ranged from 87 µg/L (hardness = 16 mg/L $CaCO_3$, pH = 7.1, DOC = 0.94 mg/L) to 1,377 µg/L (hardness = 256 mg/L $CaCO_3$, pH = 6.9, DOC = 7 mg/L; Schlekat et al. 2010). Nys et al. (2016) assessed the 7-day growth rate of *L. minor* using the number of fronds, which resulted in EC_{10}, EC_{20}, and EC_{50} values of 28, 73, and 286 µg/L, respectively (hardness = 175 mg/L $CaCO_3$, pH = 8.21, DOC = 0.045 mg/L). In comparison, Oláh et al. (2015) found that Ni inhibited the relative growth rate of giant duckweed (*Spirodela polyrhiza* (L.) Schleiden), with 7-day IC_{50} estimates of 184 µg/L and 196 µg/L for growth as total fond area and frond number, respectively (hardness = 254 mg/L $CaCO_3$, pH = 5.5, DOC = 0.3 mg/L). Therefore, it appears that *S. polyrhiza* (L.) Schleiden is more sensitive to Ni than *L. minor*.

Gopalapillai et al. (2013) studied the effect of Ni on *L. minor* in 24 different treatments with varying concentrations of different salts (either $CaSO_4$, $Ca(NO_3)_2$, $CaCl_2$, $MgSO_4$, KNO_3, or $NaNO_3$) plus a control with no salt added. The lowest 7-day IC_{25} for inhibition of root length was 7.5 µg/L, from a KNO_3 treatment (hardness = 149 mg/L $CaCO_3$, pH = 8.12, DOC = 0.5 mg/L), while the highest IC_{25} was from a $CaCl_2$ treatment (40.2 µg/L; hardness = 864 mg/L $CaCO_3$, pH = 8.16, DOC = 0.5 mg/L). The three highest $CaCl_2$ treatments resulted in IC_{25} estimates for frond count that were greater than the highest concentration measured. Not counting those results, a $MgSO_4$ treatment gave the highest IC_{25} when it was based on frond count (32 µg/L; hardness = 499 mg/L $CaCO_3$, pH = 8.15, DOC = 0.5 mg/L), while the lowest frond count IC_{25} was from a $CaSO_4$ treatment (4.46 µg/L; hardness = 300 mg/L $CaCO_3$, pH = 8.22, DOC = 0.5 mg/L).

9.3 Effects on Invertebrates

Invertebrates are generally the most sensitive aquatic species to Ni toxicity (European Chemicals Bureau 2008), with a variety of groups of invertebrates having been studied (e.g., Deleebeeck et al. 2007b, Schlekat et al. 2010, Besser et al. 2013, Nys et al. 2016, Peters et al. 2018, Soucek et al. 2020). However, the Ni sensitivity does vary among invertebrate groups. Schlekat et al. (2010) found EC_{10} values ranging from 1.1 µg/L for growth (wet weight) of the snail *L. stagnalis* (hardness = 212 mg/L $CaCO_3$, pH = 8, DOC = 0.69 mg/L) to 1,379.3 µg/L for population growth rate of the rotifer *B. calyciflorus* (hardness = 136 mg/L $CaCO_3$, pH = 7.3, DOC = 7.1 mg/L). Including their own test results, Schlekat et al. (2010) ranked *L. stagnalis* as the most sensitive species to Ni (normalized geometric mean EC_{10} = 2 µg/L) out of 15 invertebrate species in the European Ni ecotoxicity database (European Chemicals Bureau 2008), with the water flea *C. dubia* being the second most sensitive species (normalized EC_{10} = 8.5 µg/L). Niyogi et al. (2014) confirmed the high sensitivity of *L. stagnalis* to Ni. However, compared to the results found by Schlekat et al. (2010) and Niyogi et al. (2014), *L. stagnalis* was not as sensitive to Ni in tests by Crémazy et al. (2020) (see discussion in Section 9.3.29.3.2).

9.3.1 *Short-term Acute Effects*

In acute exposures, Ni is toxic to *D. magna* due to Mg^{2+} antagonism, with uptake and whole-body concentration of Mg being significantly reduced after only 24 hours of exposure to Ni (Pane et al. 2003b). Na depletion after exposure to Ni was also reported in the tissues of four aquatic invertebrates, with the gastropod *L. stagnalis* being the most affected (Leonard and Wood 2013).

Cladocerans are a group of organisms known to be most sensitive to metals (von der Ohe and Liess 2004). 48-hour LC_{50} estimates of 500 to 5,500 µg/L were reported under soft water (30 - 50 mg/L $CaCO_3$) and slightly alkaline pH conditions (7.1 to 8.1). The large differences might be due to species-specific tolerance, with *Alona affinis* and *Chydorus ovalis* being more tolerant to Ni than *Ceriodaphnia* and *Daphnia* species (Keithly et al. 2004, Deleebeeck et al. 2007b, 2008a, Kozlova et al. 2009). Using neonate *D. magna* in soft water, 48-hour LC_{50} estimates of 1,068 µg/L (hardness = 45 mg/L $CaCO_3$, pH = 7.45, DOC = 3.6 mg/L) and 1,800 µg/L (hardness = 51 mg/L $CaCO_3$, pH = 7.7, DOC = 0.7 mg/L) were reported in the studies of Pane et al. (2003b) and Chapman et al. (1980), respectively.

Experiments on mussels have been conducted using *Lampsilis cardium* juveniles propagated from *in vivo* larval stage (i.e., with the larval stage of the mussel life cycle conducted in aquaria with a fish host) and *in vitro* (i.e., the larvae are grown on cell culture). When normalized to a hardness of 50 mg/L $CaCO_3$ and at a pH of 8.4, the *in vivo* LC_{50} estimate varied from 262 to over 560 µg/L and *in vitro* from 271 to 470 µg/L (Popp et al. 2018). These results are similar to 96-hour LC_{50} estimates of 252 µg/L obtained for the mussel *Anodonta imbecilis* (now *Utterbackia imbecilis*) at a pH of 7.5 and hardness of 65 mg/L $CaCO_3$ (Keller and Zam 1991), and 234 µg/L and 269 µg/L obtained for the mussels *Amblema plicata* and *Margaritifera falcata*, respectively (hardness = ~106 mg/L $CaCO_3$, pH = ~8.3, DOC = 0.5 mg/L; Wang et al. 2017)) on toxicity tests conducted with *in vivo* juveniles. Snails are also relatively sensitive to Ni, with 96-hour LC_{50} estimates of 730 and 670 µg/L for *L. stagnalis* and *Physa gyrina*, respectively, under a pH of ~8.25, DOC of 0.5 mg/L, and hardness of ~ 105 mg/L $CaCO_3$ (Ivey et al. 2017). In comparison, the fairy shrimp (*Branchinecta lindahli*) had a 24-hour LC_{50} of 1,350 µg/L in the same conditions (Ivey et al. 2017).

Wang et al. (2020) tested two exposure times and feeding regimes on the toxicity of Ni to the amphipod *H. azteca*. A longer exposure more than halved the survival of *H. azteca*, with 48-hour and 96-hour EC_{50} estimates (based on mortality plus immobility) of 2,432 and 916 µg/L, respectively, when the organisms were not fed (hardness = 107 mg/L $CaCO_3$, pH = 7.7, DOC = 0.4 mg/L). EC_{50} estimates were increased (i.e., less toxicity) by approximately 25% and 300% for the 48-hour and 96-hour tests, respectively, when the amphipods were fed at 0 and 48 hours (48-hour EC_{50} = 3,047 µg/L; 96-hour EC_{50} = 2,732 µg/L). Wang et al. (2020) concluded that a 48-hour exposure without feeding should be conducted to remove potential interactions between DOC augmentation (i.e., via food addition) and Ni ions. In comparison, Doig and Liber (2006) found unfed juvenile *H. azteca* to be quite tolerant to Ni, with 48-hour LC_{50} estimates ranging from 9,710 to 18,490 µg/L (hardness = 137 mg/L $CaCO_3$, pH = 8.3) under variable DOC concentrations (0.6 - 30.4 mg/L), where the DOC came from different sources of DOM. However, increasing DOC concentration within a single DOM source did not decrease the acute toxicity of Ni to *H. azteca,* possibly because the concentrations of Ni required to cause acute toxicity to *H. azteca* surpass the complexing capacity of the available DOM (Doig and Liber 2006). *H. azteca* was an order of magnitude more tolerant to Ni (LC_{50} = 1,900 µg/L) than *C. dubia* (LC_{50} = 140 µg/L), but much more sensitive than the worm *L. variegatus* (LC_{50} = 75,000 µg/L) in acute tests conducted at pH 7 – 7.5 in hard water (290 mg/L $CaCO_3$; Schubauer-Berigan et al. 1993).

The midge *Chironomus* was very tolerant to Ni in acute toxicity studies (Liber et al. 2011, Leonard and Wood 2013) with 96-hour LC_{50} of 650,000 µg/L measured in a water with hardness of 141 mg/L $CaCO_3$, pHof 7.8, and DOC of 2.3 mg/L conducted with the larvae of *Chironomus riparius* (Leonard and Wood 2013). In comparison, juvenile *D. pulex* and *L. stagnalis* had LC_{50} of 1,500 and 480 µg/L, respectively, in

the same conditions. An LC_{50} of 119,261 µg/L was measured for *Chironomus dilutus* in comparable conditions, supporting the relative tolerance of this genus to Ni (Liber et al. 2011).

9.3.2 *Long-term Chronic Effects*

Similar to the results from acute toxicity tests, an exposure of *D. magna* to 131 µg/L Ni for 14 days led to a linear decrease in whole-body concentration of Mg^{2+} (Pane et al. 2003b). While impairments in respiratory function of invertebrates were not observed in acute tests, decreases in oxygen consumption rate and whole-body hemoglobin concentration by 31% and 68%, respectively, were measured during longer exposures (Pane et al. 2003b).

As under acute exposures, cladocerans were sensitive to chronic Ni exposure (e.g., De Schamphelaere et al. 2006, Deleebeeck et al. 2007b, 2008b, Nys et al. 2015, 2016, 2017, Pereira et al. 2017, 2018, Pérez and Hoang 2018, Mano et al. 2020). *C. dubia* appears to be the most sensitive water flea to Ni toxicity. It was ranked as the second-most sensitive species in the European Chemicals Bureau (2008) BLM-normalized data for six of seven water quality scenarios (behind the snail *L. stagnalis* in all cases), and was the third-most sensitive species in the seventh water quality scenario. In all water quality scenarios (based on selected eco-regions), *C. dubia* was more sensitive to Ni than the seven other cladoceran species included in the BLM (European Chemicals Bureau 2008). For example, De Schamphelaere et al. (2006) found that across six different test waters, 10-day EC_{10} estimates for *C. dubia* reproduction ranged from 1.3 µg/L (extrapolated below lowest test concentration) when hardness = 108 mg/L $CaCO_3$, pH = 7.9, and DOC = 5.02 mg/L to 44.2 µg/L when hardness = 132 mg/L $CaCO_3$, pH = 7.6, and DOC = 23.6 mg/L. A significant linear relationship between DOC and chronic Ni toxicity showed that increasing DOC reduced the toxicity of Ni to *C. dubia* (De Schamphelaere et al. 2006). In comparison, the lowest EC_{10} value compiled for *D. magna* was 7.9 µg/L (hardness = 79.5 mg/L $CaCO_3$, pH = 8.3, and DOC = 0.4 mg/L; Mano et al. 2020), while the highest *D. magna* EC_{10} estimate was 256 µg/L (hardness = 123 mg/L $CaCO_3$, pH = 6.8, DOC = 17.3 mg/L, and background dissolved Ni concentration = 4 µg/L in this field-collected water; Deleebeeck et al. 2008b). Both of these EC_{10} estimates were from 21-day tests assessing reproduction (Deleebeeck et al. 2008b, Mano et al. 2020).

As seen in the European Chemicals Bureau (2008) BLM results mentioned above, the juvenile stage of the freshwater pulmonate snail *L. stagnalis* is also generally sensitive to Ni toxicity. Nys et al. (2016) reported 28-day EC_{10} estimates for growth rate (shell length) of 4.6 µg/L when pH = 8.64 and 19.0 µg/L when pH = 8.11; both tests were conducted at a hardness of 175 mg/L $CaCO_3$ and DOC of 0.3 mg/L. In comparison, across five different exposure scenarios Schlekat et al. (2010) reported 30-day EC_{10} estimates for growth (wet weight) ranging from 1.1 µg/L when hardness = 212 mg/L $CaCO_3$, pH = 8, and DOC = 0.69 mg/L to 19.5 µg/L when hardness = 256 mg/L $CaCO_3$, pH = 7.8, and DOC = 7.1 mg/L. Similarly, Niyogi et al. (2014) found a 21-day EC_{20} for growth (wet weight) of *L. stagnalis* of <1.3 µg/L (the lowest concentration tested; hardness = 60 mg/L $CaCO_3$, pH = 7.8, DOC = 2.3 mg/L), while across five different exposure scenarios Schlekat et al. (2010) reported 30-day EC_{20} estimates for growth (wet weight) ranging from 1.6 µg/L when hardness = 212 mg/L $CaCO_3$, pH = 8, and DOC = 0.69 mg/L to 26.8 µg/L when hardness = 256 mg/L $CaCO_3$, pH = 7.8, and DOC = 7.1 mg/L. The highest EC_{20} for *L. stagnalis* from the compiled literature used in BLM development was 140 µg/L, which was from a 14-day test looking at specific growth rate where hardness = 116 mg/L $CaCO_3$, pH = 7.81, and DOC = 0.76 mg/L (Crémazy et al. 2020). A similar pattern was found with EC_{50} estimates for *L. stagnalis*, which ranged from 6.2 µg/L in a 30-day test assessing wet weight growth when hardness = 212 mg/L $CaCO_3$, pH = 8, and DOC = 0.69 mg/L (Schlekat et al. 2010) to 230 µg/L in a 14-day test assessing specific growth rate when hardness = 116 mg/L $CaCO_3$, pH = 7.81, and DOC = 0.76 mg/L (Crémazy et al. 2020). Some of the differences between these studies can likely be explained by the fact that Niyogi et al. (2014) and Schlekat et al. (2010) reported growth as wet weight biomass, while Nys et al. (2016) and Crémazy et al. (2020) used growth rate as the endpoint. Biomass has higher

variance and is more dependent on the test conditions because it does not account for the test duration in its calculation the way growth rate does (Crémazy et al. 2020). Using the calculated growth rate with the Niyogi et al. (2014) data, Crémazy et al. (2020) did not eliminate the large difference in effect concentrations.

The most tolerant invertebrate species to Ni toxicity in the compiled literature was the rotifer *B. calyciflorus,* with 48-hour EC_{10} values for population growth rate in six different test waters ranging from 103.9 µg/L when hardness = 16 mg/L $CaCO_3$, pH = 7.1, and DOC = 0.94 mg/L (Schlekat et al. 2010) to 1,576 µg/L when hardness = 175 mg/L $CaCO_3$, pH = 8.12, and DOC = 0.3 mg/L (Nys et al. 2016). Only Nys et al. (2016) calculated a 48-hour EC_{50} value for population growth rate; it was significantly higher (i.e., less toxicity; 2,459 µg/L) at pH = 8.12 than pH = 8.55 (EC_{50} = 1,576 µg/L), when hardness was held constant at 175 mg/L $CaCO_3$ and DOC was 0.3 mg/L. The midge *C. dilutus* was the second-most tolerant species tested, with 10-day EC_{10} values for growth (ash-free dry weight) ranging from 251.1 µg/L (hardness = 136 mg/L CaCO3, pH = 7.3, DOC = 7.1 mg/L) to 782.2 µg/L (hardness = 256 mg/L CaCO3, pH = 7.8, DOC = 7.1 mg/L) across five waters with varying pH, hardness, and DOC (Schlekat et al. 2010).

9.4 Effects on Fish

Nickel can be taken up by fish through the gills or olfactory epithelium, and through the gut from dietary exposure (Pyle and Couture 2011). The gastrointestinal uptake follows a concentration-dependent kinetic with saturation, suggesting that a high-affinity, low-capacity transporter is involved in the uptake of Ni in the stomach (Leonard et al. 2009). It then binds to albumin and short peptides and is transported through the blood where it can accumulate in the kidneys (Pyle and Couture 2011). Ni also inhibits the intestinal absorption of Fe^{2+} in fish, an interaction that likely occurs at the divalent metal transporter-1 of the mucosal epithelium of the intestine (Kwong and Niyogi 2009). Chemosensory and behavioral effects of Ni on fish have been sparsely reported, with rainbow trout being attracted to low Ni concentrations (6 µg/L) while avoiding concentrations over 19 µg/L (Giattina et al. 1982). In fish, damage to the respiratory epithelium is considered a major toxicity pathway (Pane et al. 2004b). However, Brix et al. (2017) concluded that effects found at the gills of fish exposed to Ni are unlikely to be due to an allergic reaction.

9.4.1 *Short-term Acute Effects*

Ni acute toxicity is caused by branchial lesions leading to decreased ventilation and oxygen extraction and overall impaired respiratory function (Pane et al. 2004b). In the kidneys of fish, Ni caused lesions in the renal tubules and impaired Mg reabsorption, leading to reduced plasma concentrations of Mg^{2+} (Pane et al. 2005). Despite these effects, Ni seems to be far less toxic to fish compared to other metals such as cadmium and copper. For instance, a juvenile rainbow trout 96-hour LC_{50} was approximately 7,800 µg/L for Ni while being 1.5 µg/L and 13.8 µg/L for Cd and Cu, respectively, at a hardness of 40 mg/L $CaCO_3$ (Buhl and Hamilton 1990, 1991). This low toxicity compared to other metals might be explained by its low binding affinity to the gills (Niyogi and Wood 2004).

Hoang et al. (2004) also reported that toxicity of Ni to fish is a function of fish age, alkalinity, hardness, and natural organic matter (NOM). In general, older fathead minnows (28-days-old) were less sensitive than <1-day-old fish, with 96-hour LC_{50} estimates of 6,380 µg/L and 1,680 µg/L, respectively, at a hardness of 47 – 52 mg/L $CaCO_3$, a pH of 7.0 – 7.2, and a DOC of 0.5 mg/L. In contrast, no difference in sensitivity was observed between alevins and juveniles of Arctic grayling (*Thymallus arcticus*) and coho salmon (*Oncorhynchus kisutch*), while juveniles of rainbow trout were more sensitive (96-hour LC_{50} of 7,800 µg/L) than alevins (96-hour LC_{50} of 25,100 µg/L; Buhl and Hamilton 1991). The fathead minnow seems to be the most sensitive species to Ni in several studies. For instance, in a study conducted in northern Canadian lakes, larval fathead minnows were more sensitive than larval northern pike (*Esox lucius*), white sucker (*Catostomus commersonii*), or rainbow trout (Pyle 2000). The acute LC_{50} for fathead minnow was 500 µg/L

at a hardness of 50 mg/L $CaCO_3$ and circumneutral pH, and 2,300 µg/L at a hardness of 160 mg/L $CaCO_3$ and pH of 7.6 (Pyle et al. 2002). In contrast, under a hardness of 91 mg/L $CaCO_3$, a pH of 8, and a DOC of 0.8 mg/L, 96-hour LC_{50} of rainbow trout was 20,800 µg/L (Brix et al. 2004), while being 15,300 µg/L under similar pH but a DOC of 3 mg/L and hardness of 140 mg/L $CaCO_3$ (Pane et al. 2003a).

Rainbow trout was also more tolerant to Ni (96-hour EC_{50} > 17,000 µg/L; hardness = ~110 mg/L $CaCO_3$, pH = 8, DOC = 0.4 mg/L) than the warm-water fishes, such as the mottled sculpin (*Cottus bairdi*; 96-hour EC_{50} of 2,400 µg/L) and lake sturgeon (*Acipenser fulvescens*; 96-hour EC_{50} of 14,000 µg/L; Besser et al. 2020).

9.4.2 *Long-term Chronic Effects*

Few studies have been conducted on the long-term toxicity of Ni to fish. Toxic effects of a longer exposure to Ni are related to histopathological damage of several tissues, including the gill structure, liver, intestine, and kidney. After 30 days of exposure to 5,700 µg/L Ni, the silver carp (*Hypophthalmichthys molitrix*) showed severe erosion and degeneration of gill epithelium, aggregation of blood corpuscle, and fusion of the boundary of the secondary lamellae (Athikesavan et al. 2006). Inflammation, degeneration and necrosis of the epithelium of the liver, kidneys, and intestines were also observed, typical of cell injury caused by reactive oxygen stress observed with other metals (Athikesavan et al. 2006). The sublethal effects of Ni on the gill's ultrastructure of rainbow trout exposed to 394 µg/L Ni for 34 days (pH 7.9, DOC 3 mg/L, and hardness 140 mg/L $CaCO_3$) could explain the lower maximum oxygen consumption and aerobic scope for activity measured in Pane et al. (2004a). In addition, egg spawning and hatchability appear as sensitive indicators of long-term exposure to Ni. An exposure of more than 30 days to a concentration of 730 µg/L Ni caused a reduction of the number of eggs per spawning and hatchability of these eggs in fathead minnows (pH 7.9 and hardness 140 mg/L $CaCO_3$) (Pickering 1974). Egg hatchability was also reduced by 50% in common carp (*Cyprinus carpio*) exposed to 6,000 µg/L (Blaylock and Frank 1979), while hatching was delayed in zebrafish (*Danio rerio*) exposed to 40 µg/L (Dave and Xiu 1991).

LC_{50} and IC_{50} for chinook salmon (*Oncorhynchus tshawytscha*) were 640 µg/L and 960 µg/L for 28-day survival and growth endpoints, respectively, at a pH of 7.6, hardness of ~ 95 mg/L $CaCO_3$, and DOC of 1.5 mg/L (ECCC 2018). For rainbow trout, a NOEC of 466 µg/L was reported because no toxicity endpoint (hatchability, swim-up, survival, length, and weight) showed any response at the highest concentration tested over a period of 28 to 55 days (Brix et al. 2004). A survival-based 28-day EC_{20} of 592 µg/L was estimated for lake sturgeon, while the lowest observed effect concentration (LOECs) for biomass of 420 µg/L and 1100 µg/L were estimated for lake sturgeon and mottled sculpin, respectively, at pH of 8.2, hardness of 100 mg/L $CaCO_3$ and DOC of 0.5 mg/L (Besser et al. 2020). Similar to acute toxicity experiments, the fathead minnow was more sensitive to Ni than other fish species in chronic toxicity tests, with a reported LOEC of 120 µg/L (hardness = 102.6 mg/L $CaCO_3$, pH = 7.4, DOC = 0.3 mg/L; Birge et al. 1984).

9.5 Effects on Amphibians

Very few studies have been conducted on the toxicity of Ni to amphibians, especially on species that live in Canada. Klemish et al. (2018) reported an 8-day EC_{10} and EC_{20} for body condition (weight/length) of wood frog (*Lithobates sylvaticus*) tadpoles of 1,070 µg/L and 2,440 µg/L, respectively (median hardness = 164 mg/L $CaCO_3$, pH = 8.28, DOC = 1.47 mg/L). However, no effects were found on survival, activity, food consumption, or chemosensory function, at Ni concentrations up to 5,500 µg/L (Klemish et al. 2018). Klemish et al. (2018) reported that most studies that investigated Ni toxicity to amphibians did not report the water hardness used. Klemish et al. (2018) converted the LC_x and EC_x estimates from studies that did report hardness to a hardness of 164 mg/L $CaCO_3$, using the U.S. EPA (1996) Ni water quality criteria equations, and found that 96-hour LC_{50} estimates ranged from 1,700 µg/L (eastern narrow-mouthed toad, *Gastrophryne carolinensis*) to 22,900 µg/L (Asian common toad, *Duttaphrynus melanostrictus*).

Fort et al. (2006) assessed the toxicity of Ni to the embryo-larval stages of three amphibian species: eastern narrow-mouthed toad (*G. carolinensis*), South African clawed frog (*Xenopus laevis*), and southern toad (*Bufo terrestris*). Although these species do not live in Canada, given the lack of data on the toxicity of Ni to amphibians, the Fort et al. (2006) results are included here for some additional context. Using a standard 96-hour developmental toxicity test method (hardness = 100 mg/L $CaCO_3$, pH = 7.7, DOC < 1 mg/L), Fort et al. (2006) found that based on mortality, eastern narrow-mouthed toad was most sensitive to Ni, followed by southern toad and South African clawed frog (mean LC_{50} estimates from two tests of 1,150 µg/L, 2,990 µg/L, and 7,950 µg/L, respectively). However, the malformation and growth endpoints were more sensitive, and by these measurements southern toad was the most tolerant species to Ni. For example, the mean EC_{10} estimates for malformation were 200 µg/L, 250 µg/L, and 1,200 µg/L for eastern narrow-mouthed toad, South African clawed frog, and southern toad, respectively (Fort et al. 2006).

The European Chemicals Bureau (2008) normalized the toxicity data for 31 species used in its BLM to seven different water quality scenarios (based on selected eco-regions). In six of the seven scenarios, the southern toad was the most tolerant species to Ni; it was the second-most tolerant species in the seventh scenario. The normalized NOEC for growth of the southern toad in these seven scenarios ranged from 433 to 3,218 µg/L (European Chemicals Bureau 2008). The other two amphibians in the dataset (eastern narrow-mouthed toad and South African clawed frog) were always among the top eight most tolerant species in the seven BLM scenarios (European Chemicals Bureau 2008). Based on the above information, amphibians appear to generally be less sensitive to Ni than algae, aquatic plants, invertebrates, and fish.

9.6 Bioaccumulation and Bioconcentration of Nickel in the Aquatic Environment

Bioconcentration (BCF; Arnot and Gobas 2006) reflects the uptake of a chemical into an organism from the surrounding water (e.g., through gills), after accounting for excretion of the chemical (e.g., through fecal egestion, metabolic biotransformation, elimination through the gills). In comparison, bioaccumulation, measured using a bioaccumulation factor (BAF), takes into account dietary (in addition to aqueous uptake) sources of the chemical (Arnot and Gobas 2006). Tissue residue concentrations are more often used as toxicity thresholds for organic pollutants than metals because homeostasis processes in plants and animals typically maintain metals within a limited concentration range between deficiency and toxicity (Gopalapillai et al. 2013).

Ni bioaccumulation was found to follow a Michaelis-Menten kinetic, with a clear hyperbolic and saturable relationship in four organisms (*D. pulex*, *L. stagnalis*, *L. variegatus,* and *C. riparius*; Leonard and Wood 2013). Niyogi et al. (2014) also found that accumulation of Ni in the soft tissue of the snail *L. stagnalis* over 24 days of exposure was concentration-dependent, with snails exposed to the two highest test concentrations (8.2 and 16.9 µg/L) accumulating significantly more Ni in their soft tissue than the control snails. The kinetics were tested over a larger concentration range, which showed there were two phases to the Ni uptake, with the Ni^{2+} intake rate following a saturable curve at lower concentrations (36 – 189 nM; similar to the concentration range used in the chronic toxicity test) and increasing in a linear manner at higher concentrations (189 – 1897 nM; Niyogi et al. 2014). In the streaked prochilod fish (*Prochilodus lineatus*), an acute exposure for periods of 24 and 96 hours to 25 - 2500 µg/L Ni led to the accumulation of this metal differently in organs, with highest to lowest concentrations measured in the kidney > liver > gills > muscles (Palermo et al. 2015). Despite these findings, MC Geer et al. (2003) demonstrated that the Ni BCFs are inversely correlated with Ni exposure concentrations across species. Consequently, it is reasonable to assert that bioconcentration is not an intrinsic attribute of nickel (McGeer et al. 2003). In addition, there is currently no clear evidence that Ni is biomagnified in the aquatic food chain.

10. THE BIOTIC LIGAND MODEL

The toxicity of Ni varies depending on water chemistry and therefore the sensitivity of organisms to Ni can depend on the chemistry of the water that they are exposed in. These toxicity modifying factors can be accounted for using a multiple linear regression (MLR) or a biotic ligand model (BLM). The BLM was selected over the MLR for several reasons including: 1) the limited studies that varied DOC necessary to derive the MLR; 2) the flexibility of the BLM for accepting new toxicity data as it becomes available; and the consideration of additional toxicity modifying factors within the BLM that have been mechanistically shown to influence toxicity (Santore et al. 2021).

The Biotic Ligand Model (BLM) expands the Gill Surface Interaction Model (GSIM) and predicts metal speciation, the binding capacity of available organic and inorganic compounds with various metal species, and the protective effects of competing cations to model the accumulation of the metal at a biologically sensitive receptor (i.e., biotic ligand) (Di Toro et al. 2001; Santore et al. 2001; Paquin et al. 2002; Santore et al. 2002). Therefore, the BLM can be used to predict the amount of metal accumulation at the biotic ligand (e.g., gill), which represents the site of action for metal toxicity, for a variety of water chemistry conditions and metal concentrations (Figure 10.1) (Di Toro et al. 2001; Paquin et al. 2002).

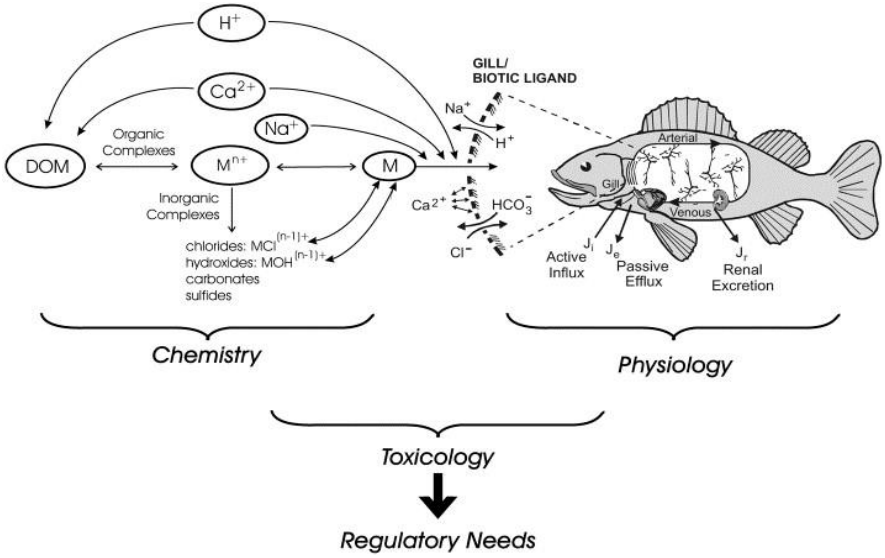

Figure 10.1. Schematic representation of the biotic ligand model. Letter "M" represents metals (reprinted with permission from Paquin et al. 2002).

According to the conceptual framework of the BLM, the accumulation of a metal at the biotic ligand at or above a critical threshold concentration leads to toxicity. Since the BLM includes inorganic and organic metal speciation and competitive complexation with the biotic ligand, it can estimate the amount of dissolved metal required to reach this threshold, depending on water chemistry.

Using published information on the accumulation of Ni on a biotic ligand (i.e., gill) as a function of water chemistry, the Ni BLM was developed to establish the Ni accumulation-toxicity relationship in different water chemistry scenarios. Nickel BLM was originally derived following Cu BLM (Wu et al. 2003). Recently, Santore et al. (2021) compiled a toxicity database on 66 freshwater species to assess and recalibrate the Ni BLM (Santore et al. 2021). After the calibration the Ni BLM showed to be a powerful tool in predicting toxicity for several aquatic plants, invertebrates, fish, and amphibian for both chronic and acute exposures (Santore et al., 2021). The only exception to the performance of the model was for some invertebrate species (e.g., *C. dubia*) where at pH values above 8, an extreme increase in toxicity was observed. The

authors argued that the increased pH does not act as a TMF for Ni. Instead, the increased effect is the toxicity of elevated bicarbonate at high alkalinity which acts as an additive effect to toxic effect of Ni (Santore et al. 2021). Nonetheless, by applying some adjustments to the model for normalization of bicarbonate-sensitive organisms (e.g., *C. dubia, Neocloeon triangulifer,* and *L. siliquoidea*) it is ensured that those organisms are protected when exposed to high Ni at high pH values.

Using a special software, BLM first calculates a critical accumulation value and then normalizes this value to the water chemistry conditions specified in the input data (Figure 10.2). The normalized acute or chronic dataset of critical accumulations values is then used to sort the normalized toxicity data that are used to derive WQGs.

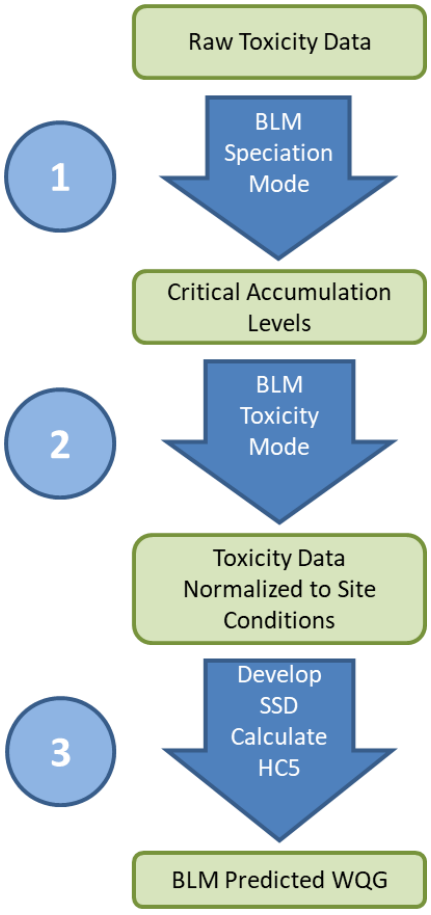

Figure 10.2 Steps and calculations involved in the BLM calculation of a WQG from the toxicity database.

Considering the ability of BLM to predict toxicity in various water chemistries and calculation of site-specific WQGs, several water quality guidelines/criteria have been derived using BLM including the B.C. Cu WGS (ENV, 2019b), the Canadian Cu Environmental Quality guidelines (ECCC 2021), The U.S. Environmental Protection Agency (USEPA) Cu Water Quality Criteria (2007), and the European Environmental Quality Standard (EQS) for Ni (European Commission, 2011).

11. Water Quality Guidelines from Other Jurisdictions

Nickel WQGs/criteria from six provincial and national jurisdictions are summarized in Table 11.1. Three types of guidelines are used: a static number, an equation considering hardness, and the BLM. In general, most of the older WQGs are hardness-based, while more recent WQGs are calculated using BLM software

that considers a wider range of TMFs. In addition, while older guidelines are for total Ni, recent guidelines use the dissolved fraction of Ni.

11.1 Canadian Council of Ministers of the Environment (CCME)

The CCME develops national WQGs for the protection of aquatic life. The CCME WQG for long-term exposure to total Ni is presented as a hardness-based equation (Table 11.1). When water hardness is below 60 mg/L $CaCO_3$, the WQG is 25 µg/L (CCREM, 1987). The CCME long-term hardness equation is valid for water hardness values between 60 and 180 mg/L $CaCO_3$. At hardness concentrations greater than 180 mg/L $CaCO_3$, the long-term WQG is 150 µg/L. The CCME does not have an acute WQG for Ni.

11.2 Provincial Water Quality Guidelines

The provinces of Canada typically develop province-specific WQGs or adopt the WQG from another jurisdiction (e.g., CCME).

The Ontario Ministry of Environment and Energy sets policies to manage Ontario's water resources, including providing Provincial Water Quality Objectives (PWQOs) for surface water to protect aquatic life (OMOEE, 1994). The OMEE has set a chronic PWQO for total Ni as a static value of 25 µg/L (OMOEE, 1994).

Saskatchewan adopted the chronic CCME WQG as an interim surface water quality objective with some minor modifications (Water Security Agency, 2015). The Saskatchewan Water Security Agency used the CCME equation as the basis for developing static guideline values of 25, 65, 110 and 150 µg/L for hardness ranges of 0-60, 60-120, 120-180 and >180 mg/L $CaCO_3$, respectively.

11.3 U.S. Environmental Protection Agency Water Quality Criteria

The USEPA develops acute (i.e., short-term) and chronic (i.e., long-term) national WQC for the protection of aquatic life. The USEPA currently recommends a hardness-based approach to calculate WQCs for total Ni (U.S. EPA 1996). The Criterion Maximum Concentrations (CMC) and Criterion Continuous Concentrations (CCC) are calculated using the equations provided in Table 11.1 (U.S. EPA 1996).

11.4 Australia and New Zealand

Australia and New Zealand have joint WQGs described as trigger values (TGVs) that trigger a response if exceeded (ANZECC 2000a; 2000b). Although four TGVs have been calculated to provide various levels of protection (i.e., 80-99% of species), ANZECC (2000a) recommends application of the 80%, 95% and 99% protection levels for protection of highly disturbed ecosystems, slightly-moderately disturbed ecosystems, and high conservation/ecological value ecosystems, respectively (ANZECC, 2000a). As an example, to protect 95% of aquatic life, ANZECC (2000a; 2000b) has developed a trigger value for Ni of 11 µg/L for waters with a hardness of 30 mg/L $CaCO_3$. The Ni trigger value can be adjusted using site-specific water hardness and then be used in the equation presented in Table 11.1.

11.5 European Union

The European Union considers Ni as a "priority substance" and therefore derived a European EQS for Ni (European Commission, 2011). The EU derived a generic EQS of 2 µg/L bioavailable Ni which is applicable and protective of all water bodies. If the measured concentration of a site exceeds the generic EQS, then a screening tool is used to calculate site-specific EQSs based on the chemistry of the site. The screening tools mimics BLM in a precautionary way and uses Ca, pH, and DOC to calculate the guidelines.

Table 11.1. Summary of water quality guidelines from other jurisdictions.

Jurisdiction	Chronic guideline (µg/L)	Acute guideline (µg/L)	Type of guideline	Total/ Dissolved	Publication Year
CCME	25 when hardness<60 mg/L; when 60<hardness<180 mg/L; use the equation: WQG = $e^{\{0.76[\ln(\text{hardness})]+1.06\}}$ 150 when hardness>180 mg/L	NA	Hardness-based	Total	1987
USEPA	CCC= $e^{\{0.846[\ln(\text{hardness})]+0.0584\}}$	CMC= $e^{\{0.846[\ln(\text{hardness})]+2.255\}}$	Hardness-based	Total	1996
Australia/ New Zealand	NA	TV*$(\text{hardness}/30)^{0.85}$	Based on hardness and intended level of protection	Dissolved	2000
European Union	Generic EQS and BLM based screening tool	NA	Bioavailability-based	Dissolved	2011
Ontario	25	NA	Static	Total	1994

TV*: trigger value varies depending on intended level of protection.

12. DERIVATION OF NICKEL WATER QUALITY GUIDELINES

12.1 Acquisition, Evaluation and Classification of Toxicological Data

WQGs for the protection of freshwater aquatic life were derived using the guidance in the *Derivation of Water Quality Guidelines for the Protection of Aquatic Life in British Columbia* (ENV 2019a). The current scientific literature on Ni toxicity to freshwater aquatic organisms was searched using Google Scholar and USEPA ECOTOX producing a total of 117 studies. Only studies on Canadian species (indigenous and non-invasive exotic species) were selected for classification which resulted in a total of 82 studies. Selected studies were evaluated to determine if they were scientifically sound and of high-quality (ENV 2019a). Information on the test species, test conditions, experimental design, chemical and physical properties of the test water, statistical analyses, and negative control performance were reviewed. Studies were then classified as primary, secondary, or unacceptable based on the criteria given in ENV (2019a). A summary of all short-term and long-term primary and secondary data, and the studies classified as unacceptable, is provided in Appendix 1.

The classification of data points as chronic long-term or acute short-term was conducted in accordance with published protocols (ENV 2019a; CCME 2007). Toxicity tests of 96 hours or less using acute endpoints (e.g., mortality) were considered to be acute short-term studies. The exception to this was studies using organisms with short life cycles, such as algae, where exposure periods as short as 24 hours may be considered long-term chronic (CCME 2007). To be considered for the chronic data set, experiments had to meet the criteria specified for fish and invertebrates in the CCME protocol (CCME 2007). Although, sub-lethal endpoints (e.g., growth and inhibition) were preferred in the chronic data set, survival data were used when chronic endpoints were not available for a given species.

From the total of 82 studies that presented data on Canadian studies a total 1824 data points were selected including 815 long-term chronic (721 primary and 94 secondary) and 297 short-term acute (215 primary and 82 secondary) (Table 12.1). A total of 712 datapoints were classified as unacceptable (Appendix 1).

The final database includes 243 primary long-term growth inhibition data points from six algae species, *Ankistrodesmus falcatus, Chlorella vulgaris, Desmodesmus spinosus, Navicula pelliculosa, Pseudokirchneriella subcapitata, Scenedesmus accuminatus* (Appendix 1).

Two duckweed species (*Lemna minor* and *Spirodela polyrhiza*) are the only macrophytes with toxicity data in the nickel toxicity dataset providing 15 primary and 86 secondary chronic data points for growth inhibition.

The dataset includes 449 chronic datapoints on 18 invertebrate species with 441 primary and 8 secondary long-term data points. The acute dataset included 177 primary and 63 secondary data points for 23 aquatic invertebrate species. Invertebrate data from 28 studies were classified as unacceptable (Appendix 1).

From the six fish species with toxicity data on Ni, there exist 20 primary long-term data points, 38 primary acute datapoints and 19 secondary acute datapoints. Data from 11 fish studies were classified as unacceptable (Appendix 1).

Toxicity data were found only on one amphibian species, Wood Frog (*Lithobates sylvaticus*) which included two primary long-term datapoint on growth.

Most unacceptable studies were missing water chemistry data, insufficient data analysis information, or a lack of mortality data for control group.

Table 12.1. Distribution of chronic and acute data points between different taxonomic groups.

Chronic Long-term							
Taxonomic group	**Total**	**Primary**			**Secondary**		
		Growth	**Reproduction**	**Survival**	**Growth**	**Reproduction**	**Survival**
Algae	243	243	0	0	0	0	0
Macrophytes	101	15	0	0	86	0	0
Aquatic invertebrates	449	147	283	11	8	0	0
Fish	20	13	0	7	0	0	0
Amphibians	2	2	0	0	0	0	0
Total	815	420	283	18	94	0	0

Acute Short-term			
Taxonomic group	**Total**	**Primary**	**Secondary**
Algae	NA*	NA*	NA*
Macrophytes	NA*	NA*	NA*
Aquatic invertebrates	240	177	63
Fish	57	38	19
Amphibians	0	0	0
Total	297	215	82

*: no data point was considered as acute for algae and macrophytes.

12.1.1 *Long-Term Chronic Water Quality Guidelines*

The chronic studies provided a total of 815 data points on several species and included multiple endpoints and effect levels for different life-stages and test durations. Using the BC Ni BLM software the whole chronic data set was normalized to a standard water chemistry (i.e., a temperature of 15°C, pH of 7.75, DOC of 3 mg/L, and hardness of 50 mg/L as $CaCO_3$[1]) and all endpoints were sorted in order of their sensitivity to Ni. Next, only the endpoint-effect level combinations that captured the lowest effects concentrations (i.e., most sensitive) for each species were selected for further use in the guideline derivation. If there was more than one comparable record (i.e., same species, same life stage, same endpoint, same exposure duration), the geometric mean of the effect concentrations were used. From this process, no-effect/low-effect estimates on 34 Canadian resident species were selected including seven algal species, two macrophyte species, 19 invertebrate species, five resident fish species (including two salmonid species) and one amphibian species (Table 12.2).

The chronic dataset does not qualify for type A1 because it does not have EC_{10} data on three fish species (EC_{10} data are available for only two of the five fish species). However, the chronic data set fulfills the minimum number of species required for the next preferred type of guideline (i.e., type A2) (ENV 2019a).

[1] The selected water quality factors are the median of the samples with available data collected from B.C. (Section 13.2).

Table 12.2. Data points used to develop the nickel long-term chronic water quality guideline.

Receptor Group / Species	Selected toxicity test endpoint	Duration	Classification	Reference
Algae/Macrophytes				
Ankistrodesmus falcatus	IC_{10}; Growth	96 h	P	Martinez-Ruiz & Marinez-Jeronimo 2015
Chlorella sp. *	EC_{10}; Growth	72 h	P	Peters et al. 2018; Macoustra et al. 2020
Chlorella vulgaris	MATC, Growth	7 d	P	Santos et al., 2019
Desmodesmus spinosus *	EC_{10}; Growth	72 h	P	Deleebeeck et al. 2006; 2009a
Lemna minor *	EC_{10}; Growth	7 d	P & S	Schlekat et al. 2010; Nys et al. 2016
Navicula pelliculosa	EC_{10}; Growth	72 h	P	Nagai & De Schamphelaere 2016
Pseudokirchneriella subcapitata *	EC_{10}; Growth	72 h	P	Deleebeeck et al. 2009
Scenedesmus accuminatus *	EC_{10}; Growth	72 h	P	Deleebeeck et al. 2006; 2009a
Spirodela polyrhiza *	EC_{50}; Growth	7 d	S	Olah et al. 2015
Invertebrates				
Brachionus calyciflorus *	EC_{10}; Growth	48 h	P & S	Schlekat et al. 2010; Nys et al. 2016
Ceriodaphnia dubia *	EC_{10}; Reproduction	10 d	P	De Schamphelaere et al. 2006
Ceriodaphnia pulchella *	EC_{10}; Reproduction	17 d	P	Deleebeeck et al. 2007a
Ceriodaphnia quadrangula *	EC_{10}; Reproduction	17 d	P	Deleebeeck et al. 2007a
Chironomus dilutus *	EC_{10}; Growth	10 d	P	Schlekat et al. 2010
Daphnia ambigua	EC_{10}; Reproduction	21 d	P	Mano et al. 2020
Daphnia galeata	EC_{10}; Reproduction	21 d	P	Mano et al. 2020
Daphnia magna *	EC_{10}; Reproduction	5th brood	P	Pereira et al. 2018
Daphnia pulex	EC_{10}; Reproduction	21 d	P	Mano et al. 2020
Daphnia similis	EC_{10}; Reproduction	21 d	P	Mano et al. 2020
Hexagenia sp. *	EC_{10}; Growth	28 d	P	Besser et al. 2011; Besser et al. 2013
Hyalella azteca	EC_{10}; Reproduction	6 w	P	Wang et al. 2020a
Hydra viridissima *	EC_{10}; Growth	96 h	P	Peters et al. 2018
Lampsilis siliquoidea *	EC_{10}; Growth	28 d	P	Besser et al. 2011; Wang et al. 2020a
Lymnaea stagnalis *	EC_{10}; Growth	30 d	P	Schlekat et al. 2010
Neocloeon triangulifer	LC_{20}; Survival	25-30 d	P	Soucek et al. 2020
Peracantha truncate *	EC_{10}; Reproduction	17 d	P	Deleebeeck et al. 2007a
Simocephalus serrulatus *	EC_{10}; Reproduction	17 d	P	Deleebeeck et al. 2007a
Simocephalus vetulus *	EC_{10}; Reproduction	17 d	P	Deleebeeck et al. 2007a
Fish – non-salmonid species				
Acipenser fulvescens	EC_{10}; Growth	28 d	P	Besser et al. 2020
Cottus bairdi	MATC, Growth	28 d	P	Besser et al. 2020
Pimephales promelas	MATC; Survival	32 d	P	Birge et al. 1984
Fish – salmonid species				
Oncorhynchus mykiss *	NOEC; Survival	26 d	S	Deleebeeck et al. 2007b
Oncorhynchus tshawytscha	IC_{10}; Growth	21 d	P	ECCC, 2018
Amphibian				
Lithobates sylvaticus	EC_{10}; Growth	8 d	P	Klemish et al. 2018

EC = effective concentration; d=days; and h=hours. IC = inhibitory concentration; LC = lethal concentration; * The reported effect concentrations are geometric means of similar data points (i.e., same species, same life stage, same endpoint and same exposure duration).

12.1.2 *Short-Term Acute Water Quality Guidelines*

The acute studies provided a total of 297 data points on several species and included multiple endpoints for different life-stages and test durations. Similar to chronic dataset (Section 12.1.1) BC Ni BLM software was used to normalized the whole acute data set to a standard water chemistry and all endpoints were sorted in order of their sensitivity to Ni. Next, only the endpoint-effect level combinations that captured the lowest effects concentrations (i.e., most sensitive) for each species were selected for further use in the guideline derivation. If there was more than one comparable record (i.e., same species, same life stage, and same exposure duration), the geometric mean of the effect concentrations were used. From this process, acute effect estimates on 28 Canadian resident species were selected including 23 invertebrate species and five resident fish species (including one salmonid species) (Table 12.3).

The acute dataset does not qualify for type A1 because of lack of data on resident amphibian species. The acute data set fulfills the minimum number of species required for the next preferred type of guideline (i.e., type A2) (ENV, 2019a).

Table 12.3. Data points used to develop the nickel short-term acute water quality guideline.

Receptor Group / Species	Selected toxicity test endpoint	Duration	Classification	Reference
Invertebrates				
Alona affinis *	LC_{50}	48 h	S	Deleebeeck et al. 2007a
Amblema plicata	LC_{50}	96 h	P	Wang et al. 2017
Anodonta imbecillis *	LC_{50}	96 h	S	Keller & Zam 1991
Bosmina coregoni *	LC_{50}	48 h	S	Deleebeeck et al. 2007a
Branchinecta lindahli	LC_{50}	24 h	S	Ivey et al. 2017
Ceriodaphnia dubia *	LC_{50}	48 h	P	De Schamphelaere et al. 2006
Ceriodaphnia pulchella *	LC_{50}	48 h	S	Deleebeeck et al. 2007a
Ceriodaphnia quadrangular *	LC_{50}	48 h	S	Deleebeeck et al. 2007a
Chironomus dilutus	LC_{50}	96 h	P	Liber et al. 2011
Chydorus ovalis *	LC_{50}	48 h	S	Deleebeeck et al. 2007a
Daphnia magna *	LC_{50}	48 h	P & S	Chapman et al. 1980; Pane et al. 2003b; Xie et al. 2007; Deleebeeck et al. 2008a; Ferreira et al. 2010; Ivey et al. 2017; Kim et al. 2017; Lari et al. 2017; Traudt et al. 2017; Mano & Shinohara 2020
Daphnia pulex *	LC_{50}	48 h	P	Leonard & Wood 2013
Hyalella Azteca *	LC_{50}	96 h	P & S	Keithly et al. 2004; Liber et al. 2011; Wang et al. 2020a
Lampsilis cardium *	LC_{50}	96 h	P	Popp et al. 2018
Lampsilis siliquoidea *	LC_{50}	96 h	P	Wang et al. 2017
Lumbriculus variegatus *	LC_{50}	96 h	P	Leonard & Wood 2013
Lymnaea stagnalis *	LC_{50}	96 h	P & S	Leonard & Wood 2013; Ivey et al.
Margaritifera falcata	LC_{50}	96 h	P	Wang et al. 2017
Peracantha truncate *	LC_{50}	48 h	S	Deleebeeck et al. 2007a
Physa gyrina *	LC_{50}	96 h	S	Ivey et al. 2017
Simocephalus serrulatus *	LC_{50}	48 h	S	Deleebeeck et al. 2007a
Simocephalus vetulus *	LC_{50}	48 h	S	Deleebeeck et al. 2007a
Utterbackia imbecillis *	LC_{50}	96 h	P & S	Keller 2000; Popp et al. 2018; Wang et al. 2017
Fish – non-salmonid species				
Acipenser fulvescens	LC_{50}	96 h	P	Besser et al. 2020
Cottus bairdi	LC_{50}	96 h	P	Besser et al. 2020
Lepomis macrochirus	LC_{50}	96 h	P	Cairns et al. 1981
Pimephales promelas *	LC_{50}	96 h	P & S	Schubauer-Berigan et al. 1993; Pyle et al. 2002; Hoang et al. 2004
Fish – salmonid species				
Oncorhynchus mykiss *	LC_{50}	96 h	P & S	Pane et al. 2003a; Brix et al. 2004

EC = effective concentration; h=hours. IC = inhibitory concentration; LC = lethal concentration; * The reported effect concentrations are geometric means of similar data points (i.e., same species, same life stage, and same exposure duration).

12.2 BC Ni BLM

Software is necessary to run the BLM model and calculate a WQG for a specific set of water chemistry conditions. The BC Ni BLM software is available on the B.C. approved WQG website. A User's Manual accompanies the software. The BLM software allows users to enter water chemistry conditions and provides a guideline value specific to the input water chemistry. The software that accompanies this guideline was developed by Windward Environmental Ltd. from published information on Ni toxicity and accumulation of Ni on biotic ligands as a function of water chemistry and follows the ENV derivation protocol (ENV, 2019a) to calculate WQGs specific to BC waterbodies.

12.2.1 *Estimating Water Quality Conditions from Laboratory Studies*

A minimum set of water chemistry data is needed to normalize toxicity values using the BC Ni BLM. In some cases, toxicity studies lacked information on one or more water chemistry factors. For example, there were cases in which a primary or secondary study reported hardness but not the concentrations of individual ions. To include as much toxicity data as possible in the database, missing water chemistry data were estimated following the method described in Santore et al. (2021).

12.2.2 *Water Chemistry Input Necessary for BC Ni BLM*

There are 11 essential water chemistry parameters needed to calculate WQGs using the BC Ni BLM software: temperature, pH, DOC, the humic acid ratio of DOC, alkalinity, Ca, Mg, Na, K, SO_4, and Cl. Some of the water chemistry parameters (i.e., pH, DOC, Ca, Mg, Na and alkalinity) have an important effect on BC Ni BLM predictions, while others have only minor effects.

To overcome the fact that not all of these parameters are routinely measured, a simplified version of the BC Ni BLM was included in the software that requires only four water chemistry parameters: temperature, DOC, pH, and hardness.

Estimates of the remaining water chemistry parameters are based on the following median molar ion ratios calculated from the minimally impacted water bodies in B.C. (described in Section 6):

- Ca:Mg - 3.33
- Ca:Na - 3.30
- Ca:K - 23.51
- SO_4:Cl - 3.86

The BC Ni BLM simplified version estimates the humic acid content and the alkalinity based on the methods described in (ENV, 2019b) Section 12.2.1.

12.2.3 *Long-term water quality guidelines*

The selected chronic datapoints on the 34 Canadian species were used for derivation of site-specific Ni WQGs using the BC Ni BLM. For each site, the selected toxicity datapoints are first normalized to that site's water chemistry. The BC Ni BLM then fits the species sensitivity distributions to the normalized data following the approach provided in *ssdtools* R package (Thorley and Schwarz, 2018). Similar to *ssdtools*, the BC Ni BLM estimates an HC_5 value using maximum likelihood estimation (MLE) and model averaging of three distributions (i.e., log logistic, log normal, and gamma).Figure 12.1 is an example of model averaged SSD produced by BC Ni BLM in a water with the chemistry specified in (Table 12.4).

Table 12.4. The "standard" water chemistry used to normalize the toxicity data.

Parameter	Value
Temperature (°C)	15
pH	7.5
DOC (mg/L)	5
Humic acid content (%)	10
Calcium (mg/L)	15
Magnesium (mg/L)	3
Sodium (mg/L)	3
Potassium (mg/L)	1
Sulfate (mg/L)	36
Chloride (mg/L)	3.5
Alkalinity (mg/L)	15

The five most sensitive species on the SSD are invertebrates and the most sensitive chronic endpoint to Ni is the growth of the juvenile snail *L. stagnalis* (EC_{10} of 2.1 µg/L) (Table 12.5, Figure 12.1). The sensitivities of plants were spread across the SSD graph with the most sensitive species being the microalgae *A. falcatus*, with a normalized 96-hour IC_{10} of 14.7 µg/L (Figure 12.1; Table 12.5). Fish are generally amongst the least sensitive species to Ni. The lowest chronic effect concentration reported for fish is for the fathead minnow (*P. promelas*), with a 32-day survival maximum acceptable toxicant concentration (MATC) of 103.3 µg/L. Only one study on amphibians living in B.C. was available, and it is ranked as one of the least sensitive endpoints (the growth-based 8-day EC_{10} of the wood frog tadpole (*Lithobates sylvaticus*) of 581.9 µg/L.

To account for the sources of uncertainty associated with WQG derivation, an AF must be applied to the calculated HC_5 (ENV, 2019a). For a type A2 WQG, the AF begins with a default value of five that may be reduced or increased depending upon the residual uncertainty of the WQG (ENV, 2019a). The Ni toxicity dataset is a relativity rich dataset and has data for algae, macrophytes, planktonic crustaceans, mollusks, EPT, salmonid fish, non-salmonid fish and amphibians resident to B.C. In addition, the average model has a good fit based on visual inspection. Considering the above points, the lowest AF of 2 was selected and applied to the calculated HC_5 to derive the WQG specific to each water chemistry.

Protection clause

To ensure that the chronic WQG aligns with the protection principles outlined in the derivation protocol (ENV 2019a), an automatic verification is conducted to determine whether the protection clause should be invoked if necessary. This process is automated in the BLM software. If a lethal endpoint is below the HC_5, the software will automatically replace the HC_5 with the lethal endpoint, as per the Protection Clause in the BC WQG derivation protocol (2019a). Following this adjustment, the assessment factor will be applied to the newly established lethal endpoint.

12.2.4 *Short-term water quality guidelines*

To derive the acute WQG an approach similar to the one used for chronic WQG is used (Section 12.2.3). For each water chemistry, the selected acute datapoints of the 28 Canadian species are normalized and model averaging of the three SSD distributions are used to calculate the HC_5 for each site.

After normalization to the standard water chemistry (Section 12.2.3)30, the most sensitive invertebrate species to acute Ni exposure was juveniles (> 2 days old) of *C. dubia* (normalized LC_{50} of 75.1 µg/L which is a geometric mean of 6 data points) (Figure 12.2;Table 12.6). The least sensitive species to acute Ni exposure was also an invertebrate *C. dilutus*, with a 96-hour LC_{50} of 60,210 µg/L. Fish are not generally

sensitive to acute exposure to Ni. Nonetheless, the most sensitive fish species is fathead minnow (*P. promelas*) with a normalized 96-hour LC_{50} of 1,530 µg/L.

The Ni acute toxicity dataset is relativity a rich dataset and has data for the following B.C. resident species: planktonic crustaceans, mollusks, sensitive macroinvertebrates (i.e., Ephemeroptera, Plecoptera, and Trichoptera; EPT), salmonid fish, and non-salmonid fish. The average model for acute datapoints has a good fit based on visual inspection. However, the toxicity dataset does not include data on any amphibian species. Considering the above points, the AF of 3 was selected and applied to the calculated HC_5 to derive the acute WQG specific to each water chemistry.

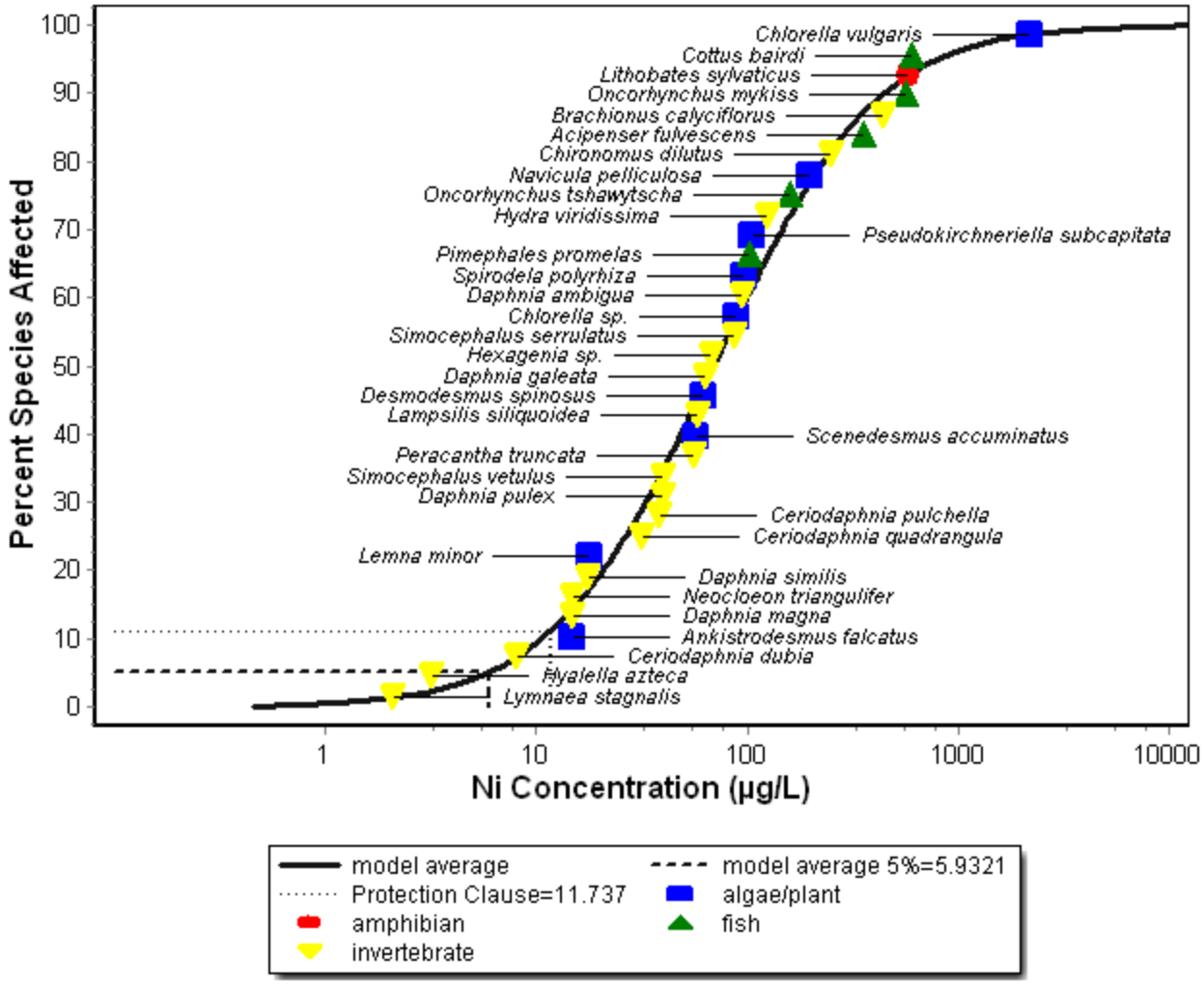

Figure 12.1. The distribution of normalized long-term effect used to derive the nickel type A2 long-term water quality guideline for the protection of freshwater aquatic life.

Table 12.5. Toxicity data used to develop the Ni long-term WQG normalized to standard water chemistry.

Species	Group	Selected toxicity test endpoint	Exposure duration	Effect value (µg/L)
Lymnaea stagnalis		EC_{10}; Growth	30 days	2.1*
Hyalella azteca		EC_{10}; Reproduction	6 weeks	3.2
Ceriodaphnia dubia		EC_{10}; Reproduction	10 days	8.1*
Ankistrodesmus falcatus		IC_{10}; Growth	96 hours	14.7
Daphnia magna		EC_{10}; Reproduction	5th brood	14.7*
Neocloeon triangulifer		EC_{20}; Growth	25-30 days	14.9
Daphnia similis		EC_{10}; Reproduction	21 days	17.7
Lemna minor		EC_{10}; Growth	7 days	17.9*
Ceriodaphnia quadrangula		EC_{10}; Reproduction	17 days	31.9*
Ceriodaphnia pulchella		EC_{10}; Reproduction	17 days	38.4*
Daphnia pulex		EC_{10}; Reproduction	21 days	39.7
Simocephalus vetulus		EC_{10}; Reproduction	17 days	40.0*
Peracantha truncata		EC_{10}; Reproduction	17 days	55.6*
Scenedesmus accuminatus		EC_{10}; Growth	72 hours	57.1*
Lampsilis siliquoidea		EC_{10}; Growth	28 days	58.8*
Desmodesmus spinosus		EC_{10}; Growth	72 hours	62.6*
Daphnia galeata		EC_{10}; Reproduction	21 days	63.5
Hexagenia sp.		EC_{10}; Growth	28 days	68.1*
Simocephalus serrulatus		EC_{10}; Reproduction	17 days	86.4*
Chlorella sp.		EC_{10}; Growth	72 hours	88.9*
Daphnia ambigua		EC_{10}; Reproduction	21 days	95.4
Spirodela polyrhiza		IC_{50}; Growth	7 days	95.8*
Pimephales promelas		MATC; Survival	32 days	103.3
Pseudokirchneriella subcapitata		EC_{10}; Growth	72 hours	106.3*
Hydra viridissima		EC_{10}; Growth	96 hours	124.5*
Oncorhynchus tshawytscha		IC_{10}; Growth	28 days	161.9
Navicula pelliculosa		EC_{10}; Growth	72 hours	198.9
Chironomus dilutus		EC_{10}; Growth	10 days	247.6*
Acipenser fulvescens		EC_{10}; Growth	28 days	358.5
Brachionus calyciflorus		EC_{10}; Growth	48 hours	438.1*
Oncorhynchus mykiss		NOEC; Survival	17-26 days	568.4*
Lithobates sylvaticus		EC_{10}; Growth	8 days	581.9
Cottus bairdi		MATC; Growth	28 days	600.2
Chlorella vulgaris		LOEC; Growth	7 days	2201

* The reported effect concentrations are geometric means of similar data points (i.e., same species, same life stage, same endpoint and same exposure duration).

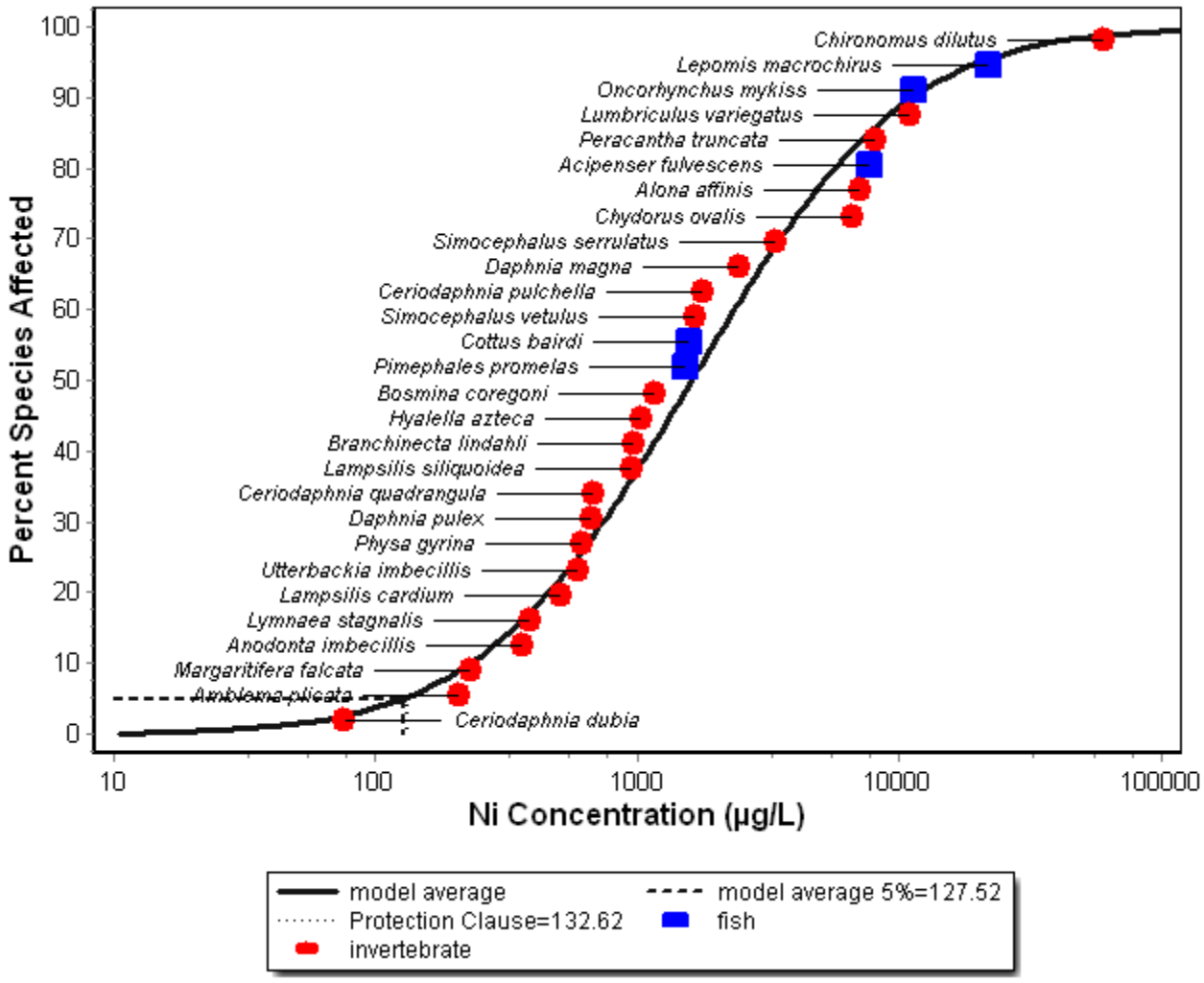

Figure 12.2. The distribution of normalized acute estimates used to derive the nickel type A2 short-term water quality guideline for the protection of freshwater aquatic life.

Table 12.6. Toxicity data used to develop the Ni short-term WQG normalized to standard water chemistry.

Group/Species	Exposure duration	LC_{50} (µg/L)
Ceriodaphnia dubia	48 hours	75.1*
Amblema plicata	96 hours	206.9
Margaritifera falcata	96 hours	230.5
Anodonta imbecillis	96 hours	360.8*
Lymnaea stagnalis	96 hours	388.6*
Lampsilis cardium	96 hours	512.8*
Utterbackia imbecillis	96 hours	591.5*
Physa gyrina	96 hours	612.7*
Daphnia pulex	48 hours	668.2*
Ceriodaphnia quadrangula	48 hours	683.6*
Lampsilis siliquoidea	96 hours	944.3*
Branchinecta lindahli	24 hours	965.3
Hyalella azteca	96 hours	1026*
Bosmina coregoni	48 hours	1164*
Pimephales promelas	96 hours	1530*
Cottus bairdi	96 hours	1578*
Simocephalus vetulus	48 hours	1659*
Ceriodaphnia pulchella	48 hours	1773*
Daphnia magna	48 hours	2441
Simocephalus serrulatus	48 hours	3353
Chydorus ovalis	48 hours	6621
Alona affinis	48 hours	7122
Acipenser fulvescens	96 hours	7657
Peracantha truncata	48 hours	8142
Lumbriculus variegatus	96 hours	10980
Oncorhynchus mykiss	96 hours	11320
Lepomis macrochirus	96 hours	22070
Chironomus dilutus	96 hours	60210

* The reported effect concentrations are geometric means of similar data points (i.e., same species, same life stage, same endpoint and same exposure duration).

12.2.5 *Model bounds*

The BC Ni BLM software was developed and calibrated for a range of values for each essential water chemistry parameter (e.g., pH, DOC, or hardness cations). Since it is only within those bounds that the protectiveness of the WQGs calculated by BC BLM has been shown, if a water chemistry value is outside of this range, the BC Ni BLM substitutes the minimum or maximum value before normalizing the data. For each parameter, the minimum and maximum reported values in the chronic and acute toxicity studies used to validate the BLM models were compiled (Table 12.7) with the following two exceptions discussed below.

The BC Ni BLM predicts Ni to be less toxic in waters with a combination of high DOC (higher than 25) and low hardness (lower than 20), based on the estimation that the DOC binding sites are free from cations and available to Ni ions to bind. Therefore, the predicted effect concentration is increased significantly which in turn would result in high WQGs. However, the toxicity data that are collected in high DOC and low hardness are rare and thus the effect cannot be verified. To prevent this phenomenon the higher bound for DOC is set to 25 mg/L and the lower bound for hardness is set to 20 mg/L. For full BLM the lower bounds of 6.2 and 1.1 mg/L for Ca and Mg were set which reflects the BC median Ca:Mg molar ratio of 3.33.

Table 12.7. Model bounds for the BC Ni BLM.

Parameter	Limits from Toxicity Database		Input Chemistry Bounds Used in BLM Software		Parameter Importance
	Lower	Upper	Lower	Upper	
Temperature (°C)	4	27	4	27	Low
pH	3.5	8.9	3.5	8.9	High
DOC (mg/L)	0.032	41	0.032	25	High
Humic acid content (%)	0.01	99	0.01	99	Low
Calcium (mg/L)	0.05	392	6.2	392	High
Magnesium (mg/L)	0.2	115	1.112	115	High
Sodium (mg/L)	0.00751	322	0.00751	322	High
Potassium (mg/L)	3.91e-6	166.86	3.91e-6	166.86	Low
Sulfate (mg/L)	0.51	890.29	0.51	890	Low
Chloride (mg/L)	0.0414	550	0.0414	550	Low
Alkalinity (mg/L)	0.01	728.48	0.01	728	High
DIC (mmol/L)	0.00162	14.729	0.00162	14.729	High
Hardness (mg/L)	1.1	1098.35	20	1100	High

12.2.6 *Protectiveness of BC BLM Acute Guidelines Against Short-term Effects on Survival*

The WQG derivation protocol characterizes the protection of aquatic life by the protection of individual organisms, which results in the overall protection of populations (ENV, 2019a). However, the most abundant effect level in short-term toxicity studies is the LC_{50}. Although the addition of an uncertainty factor will offer further protection, acute WQGs that are derived based on LC_{50}s may not protect the whole population of the sensitive species against lethality. To test this, the acute WQG was compared against no effect concentrations for the sensitive species.

To determine if the acute WQG calculated by BC Ni BLM is protective of the most sensitive organism at the most sensitive life stage, normalized LC_{10} values for the five lowest effect concentrations were compared against the acute WQG (Table 12.8). The LC_{10} or the raw data to calculate the LC_{10} were not available for *C. dubia* and the paper pondshell mussel. For the remaining species, ratios of the acute

WQG/LC_{10} concentrations ranged from 0.19 to 0.26, demonstrating that the acute WQG would be protective of sensitive species against short-term effects on survival.

Table 12.8. Normalized effect concentrations of the most sensitive endpoints and their ratio to acute WQG. Data were normalized to the standard water chemistry (see Section above12.2.3)

Species	LC_{50} (µg/L)	Normalized LC_{50} (µg/L)	LC_{10} (µg/L)	Normalized LC_{10} (µg/L)	Acute WQG	Acute WQG/ LC_{10}	Reference
Ceriodaphnia dubia	183 35.2 50.8 34.6 88.7 161	114.7 100.3 160.6 76.33 121.1 210.8	NA*	NA	42.5	NA	De Schamphelaere et al. 2006
Three ridge mussel	234	206.9	182**	160.9	42.5	0.23	Wang et al., 2017
Western pearlshell	269	209	212.1	181.7	42.5	0.26	Wang et al., 2017
Paper pondshell mussel	189.6 251.5	286.6 376.6	NA	NA	42.5	NA	Keller & Zam 1991
Great pond snail	480 736	258.1 584.9	228 530	227.2***	42.5	0.19	Leonard & Wood 2013 Ivey et al. 2017

* LC_{10} were not available nor the raw the data to calculate the LC_{10}.
** LC_{10} calculated from the data presented in the study.
*** Geometric mean of two or more datapoints.

12.2.7 *Simplified vs Full BLM*

The BC BLM is based on 13 water quality parameters but for some sites these parameters may not be available. The simplified BLM is a screening tool that estimates missing water quality data based on median background conditions and it can be used to estimate the Ni WQG when the full suite of water chemistry data are not available. However, using the simplified BLM introduces a level of uncertainty. Consequently, the guideline value estimated under these conditions may vary from the one calculated with full chemistry. In specific situations, the guideline derived from simple chemistry might be notably higher than the guideline determined with full chemistry.

To assess the results of estimating the WQG using the simplified BC BLM, data from background sites in BC were run using both simple and full chemistry over the range of conditions the guideline may be applied. The estimated simplified WQG was greater than the full WQG for 87% of the data points for the long-term chronic guideline and 76% of the data points for the short-term acute guideline (Figure 12.3 panel A). As uncertainty was observed to rise with increased hardness, a correction factor that varies with hardness was developed for long-term guidelines to ensure the estimated simple WQG did not exceed the full WQG (Figure 12.3 panel B).

The long-term chronic correction factor was derived by creating equally spaced log-hardness bins and determining the minimum correction factor that would be needed to result in an estimate of the WQG with simple chemistry that does not exceed the WQG calculated using full chemistry. A best-fit line was fit through these points (Figure 12.3, panel C), with a floor for the correction factor set at the minimum hardness concentrations in the background CABIN data, and the ceiling set below the model bounds for hardness. This relationship is represented in the following equation (written as a function for MS Excel)

Equation 1:

$$CF_{long-term} = \min\left(0.8371, \max(0.5446, -0.1067 * log_{10}(Hardness, mg/L) + 0.7846)\right)$$

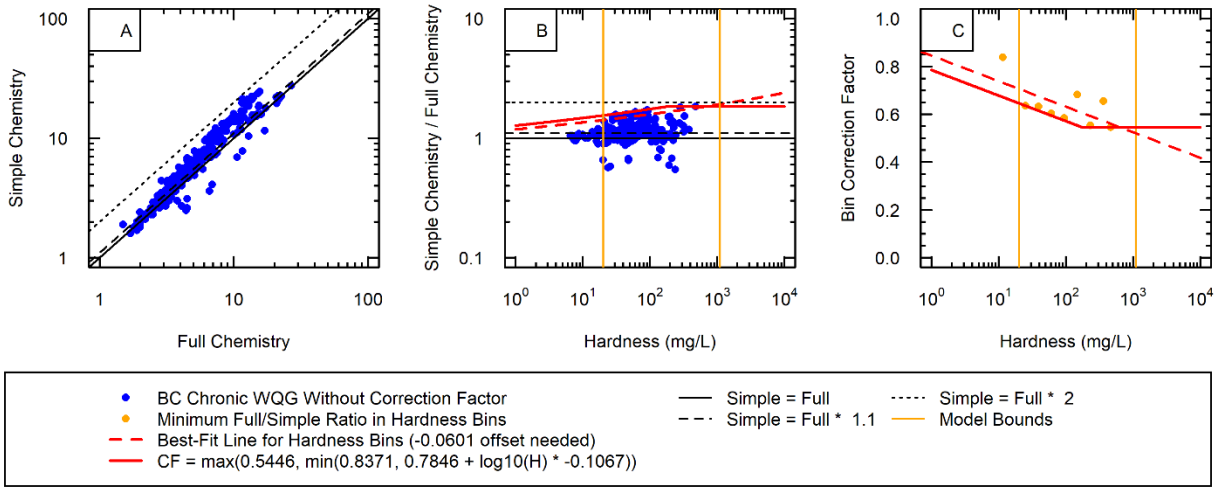

Figure 12.3. Derivation of the long-term simple-to-full correction factor: (A) comparison of simple and full chemistry guideline values, (B) ratio of simple to full chemistry compared to hardness, and (C) correction factors needed to be conservative in hardness bins.

The short-term acute correction factor was examined in a similar way to the long-term methods (Figure 12.4). Although there was a hardness trend seen in the short-term estimates, it was not statistically significant and for simplicity, a single value was derived that would ensure the short-term benchmark estimated by the simplified BLM would not exceed the full chemistry benchmark for all the BC background and CABIN data. The short-term correction factor was therefore set to 0.671 for all input hardness values.

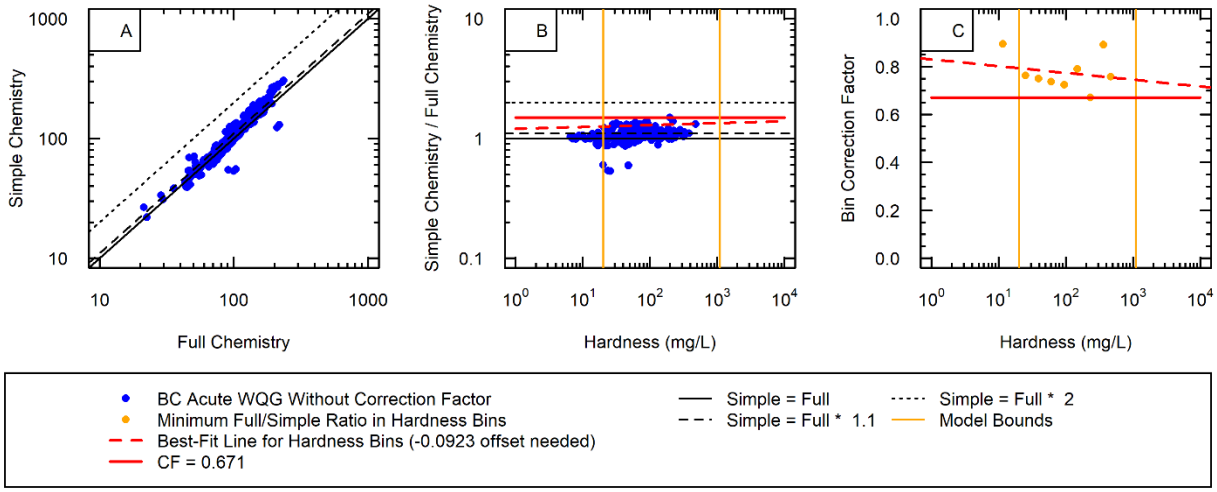

Figure 12.4. Derivation of the short-term simple-to-full correction factor: (A) comparison of simple and full chemistry HC5 values, (B) ratio of simple to full chemistry compared to hardness, and (C) correction factors needed to be conservative in hardness bins.

The correction factor is applied within the software. However, if the user is missing calcium, magnesium, or alkalinity from the full chemistry, but is using the full chemistry tab in the software, the correction factor must be manually calculated using Equation 1 for calculating the estimated simple chronic WQG. For simple acute WQGs, the resultant estimated WQG should be manually multiplied by the acute CF of 0.671.

12.2.8 *Examples of Nickel Water Quality Guidelines in Various Water Chemistry*

As the Ni WQGs are determined in relation to water chemistry conditions, values under various water chemistry scenarios are presented to provide a sense of the range in Ni WQGs. Since the effect of temperature on determining Ni WQGs is minor, a constant temperature (15° C) was considered for all scenarios. Using the BC Ni BLM, both chronic and acute WQGs were calculated for the 60 water chemistry scenarios (Table 12.9).

Table 12.9. Examples of chronic and acute Ni WQGs calculated using simplified BC Ni BLM for B.C. waters.

Water Chemistry			Chronic WQG (µg/L)	Acute WQG (µg/L)
Hardness (mg/L)	DOC (mg/L)	pH		
20	1.0	5.5	0.6	9.4
20	1.0	7	0.6	9.6
20	1.0	8.5	0.6	8.6
20	5	5.5	0.8	12.5
20	5	7	1	15.5
20	5	8.5	1.8	20.1
20	10	5.5	1	16.7
20	10	7	1.5	23.9
20	10	8.5	3.7	36.8
20	20	5.5	1.6	26.8
20	20	7	2.6	43.8
20	20	8.5	7.9	76.4
50	1.0	5.5	1	18.2
50	1.0	7	1	18.1
50	1.0	8.5	0.8	14.6
50	5	5.5	1.2	20.9
50	5	7	1.4	24
50	5	8.5	2.3	27.0
50	10	5.5	1.4	24.4
50	10	7	1.9	31.8
50	10	8.5	4.4	43.6
50	20	5.5	1.8	31.9
50	20	7	2.9	48.5
50	20	8.5	9.2	80.6
150	1.0	5.5	2.5	47.3
150	1.0	7	2.4	46.2

Table. 12.5. Continued.

Water Chemistry			Chronic WQG (µg/L)	Acute WQG (µg/L)
Hardness (mg/L)	DOC (mg/L)	pH		
150	1.0	8.5	1.6	34.2
150	5	5.5	2.6	50
150	5	7	2.9	52.9
150	5	8.5	3.4	48.5
150	10	5.5	2.8	53.3
150	10	7	3.4	61.4
150	10	8.5	5.9	66.7
150	20	5.5	3.1	60.1
150	20	7	4.5	78.9
150	20	8.5	11.4	105.0
400	1.0	5.5	6.1	118.9
400	1.0	7	6	115.2
400	1.0	8.5	3.4	79.5
400	5	5.5	6.3	121.7
400	5	7	6.5	123
400	5	8.5	5.4	95.5
400	10	5.5	6.4	125.2
400	10	7	7.3	132.8
400	10	8.5	8.1	115.4
400	20	5.5	6.8	132.1
400	20	7	8.7	152.7
400	20	8.5	14.1	155.4
1100	1.0	5.5	16.4	317
1100	1.0	7	15.8	305.8
1100	1.0	8.5	8.1	201.6
1100	5	5.5	16.5	320
1100	5	7	16.6	315.3
1100	5	8.5	10.5	220.0
1100	10	5.5	16.7	323.9
1100	10	7	17.6	327.1
1100	10	8.5	13.6	242.7
1100	20	5.5	17.1	331.5
1100	20	7	19.6	350.7
1100	20	8.5	20.3	287.4

13. APPLYING THE NICKEL WATER QUALITY GUIDELINES

An accompanying document, the BC Ni BLM User's Manual, provides instructions for installation and use of the BC BLM software (ENV, 2022). In general, water chemistry data are entered into the software, which then calculates a specific acute or chronic WQG for the input water chemistry. The output files include: a text file with the water chemistry inputs and the calculated guideline; an Excel spreadsheet with the normalized toxicity endpoints for each species; and a graph showing the SSD for normalized toxicity data and the final guideline value.

The long-term chronic and short-term acute Ni WQGs for the protection of aquatic life are calculated using the BC Ni BLM software and are based on the dissolved fraction of Ni in water. The dissolved fraction provides a better estimate of bioavailable Ni since Ni associated with suspended particulates is generally not available for biological uptake. Total metal concentrations still provide valuable information, especially in systems with high total Ni to dissolved Ni ratios, as changes in water chemistry (e.g., pH) can change the dynamics of particulate and dissolved Ni.

13.1 Application of Long-term Chronic and Short-Term Acute Water Quality Guidelines

The long-term chronic and short-term acute Ni WQGs for the protection of aquatic life are based on dissolved Ni and therefore must be compared against dissolved concentration of Ni in the aquatic ecosystems. However, if only the total concentration is available, the total concentration can be compared against the dissolved WQG. If an exceedance is observed, a resampling for dissolved Ni is required to determine if the dissolved concentration has exceeded the WQG. Given that a correction factor is incorporated into the simplified BLM, if the simplified WQG is surpassed it is recommended to gather the requisite water chemistry data for computing the complete WQG.

Like any other water quality factor, Ni concentrations are variable in natural waters. Therefore, an averaging period approach is recommended, which allows Ni concentrations to fluctuate above and below the chronic WQG over the specified averaging period (i.e., 5 samples in 30 days). For each sample, the water chemistry parameters needed for the "full" or "simplified" BC Ni BLM are required to calculate the WQG. To meet the chronic WQGs, two criteria must be met:

1. Only 20 percent of the samples (e.g., 1 in 5 samples) can exceed the chronic WQG calculated for the associated water chemistry, provided that the short-term acute WQG is never exceeded.
2. The average Ni concentration should not exceed the average chronic WQG.

In cases where less than five samples are available, each Ni concentration should be compared against the chronic long-term WQG.

Nickel short-term acute WQGs is a concentration that should not be exceeded at any time to meet the intended protection of the most sensitive species and life stage against severe effects. Short-term maximum WQGs are intended to assess risks associated with infrequent and transient exposure events such as spills.

13.2 Comparison of ambient concentrations from B.C. with proposed guidelines

Water quality guidelines are commonly used to determine the potential risk of toxicity to aquatic life caused by a given substance in ambient conditions. In general, if ambient concentrations are below WQG concentrations, the risk is assumed to be low. Water quality guidelines are periodically updated to reflect the most current information available. These updates may result in different numerical WQG values for several reasons, including:

- toxicity studies using more sensitive species;
- lower MDLs;
- the inclusion of additional TMFs;
- WQGs based on the dissolved form, rather than the total form of a substance; and,
- changes to the applied uncertainty factor(s).

It is important to understand the applicability of any proposed WQG given the distribution of nickel in natural (i.e., unimpacted) waters of B.C. To answer this question, water quality data were extracted from the B.C. EMS database for unimpacted (i.e., minimally disturbed) lakes, ponds, creeks, rivers, and streams. Only records with results for total Ni, dissolved Ni, dissolved hardness, pH, and DOC were used to calculate both the previous B.C. working WQG (i.e., CCREM 1987) and revised WQG for each sample. Non-detect data were not included if the MDL for dissolved Ni, total Ni, and DOC was greater than 0.5 µg/L, 25 µg/L, and 0.5 mg/L, respectively. Results reported as "<MDL", were given the value of the MDL. Using these selection criteria, a total of 1,471 records, from 2001 to 2021, were retrieved.

For each record, the 1987 CCME total Ni chronic WQG (Section 11.1) was calculated and compared against the measured total Ni concentration, and the revised dissolved Ni chronic WQG was calculated using the simplified B.C. BLM (Section 12.2) and compared against the measured dissolved Ni concentration. There were no total Ni concentrations that exceeded the 1987 CCME WQGs (Figure 13.1), while only 33 dissolved Ni concentrations (2.2 %) exceeded the revised WQGs (Figure 13.2; Table 13.1). These results are due to differences between the two WQGs (e.g., inclusion of more TMFs in the revised WQG and the fact that the revised WQG is based on the dissolved fraction instead of total Ni).

The slightly higher frequency of exceedances in the revised Ni WQGs is due, in part, to the fact that the minimum value in the 1987 CCME WQGs is 25 µg/L (Figure 13.1), whereas the revised WQGs, based on several TMFs and the inclusion of recent toxicity data from more Ni-sensitive species, can be as low as 0.5 µg/L (Figure 13.2). Users should bear in mind that WQGs represent low-risk conditions and ambient concentrations exceeding WQG concentrations do not necessarily imply increased risk. Further work would be required to assess site-specific conditions and the actual level of risk to aquatic life.

Table 13.1. Comparison of total exceedances between the 1987 total Ni chronic WQG and the revised dissolved Ni chronic WQGs for 1,471 B.C. water samples collected between 2001 and 2021.

	1987 CCME Total Ni WQG (µg/L)	**2022 B.C. BLM Dissolved Ni WQG (µg/L)**
Average of WQGs	50.1	2.79
Range of WQGs	25 – 150	0.5 – 17.6
Number of records with ambient Ni concentrations exceeding WQGs	0 (0%)	33 (2.2%)

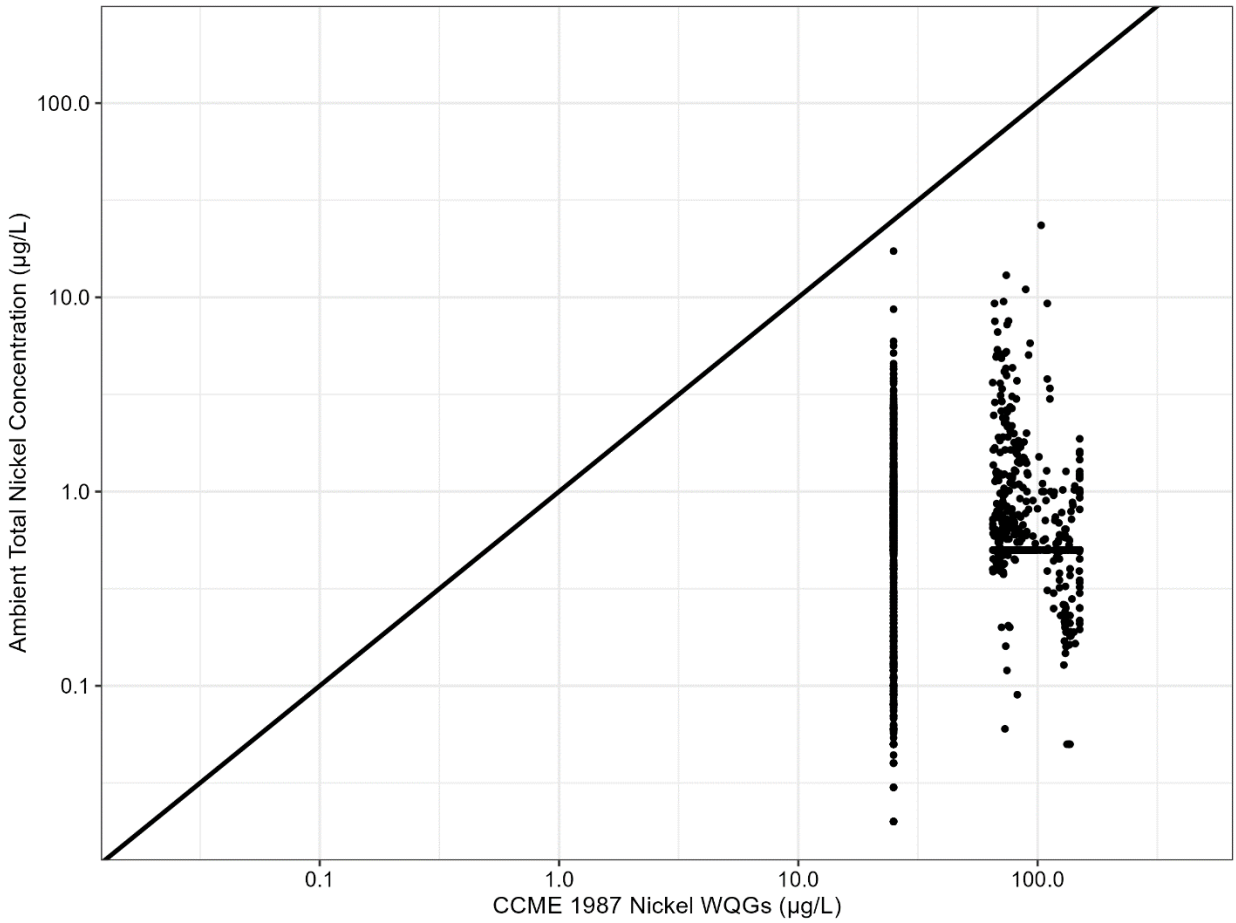

Figure 13.1. Ambient total Ni concentrations compared to the 1987 CCME chronic total Ni WQGs.

Points above the solid 1:1 line represent exceedances. The minimum value for the 1987 CCME chronic WQG is 25 µg/L total Ni. The maximum value for the 1987 CCME chronic WQG is 150 µg/L total Ni, based on a water hardness of 180 mg/L. Both axes are on a log scale.

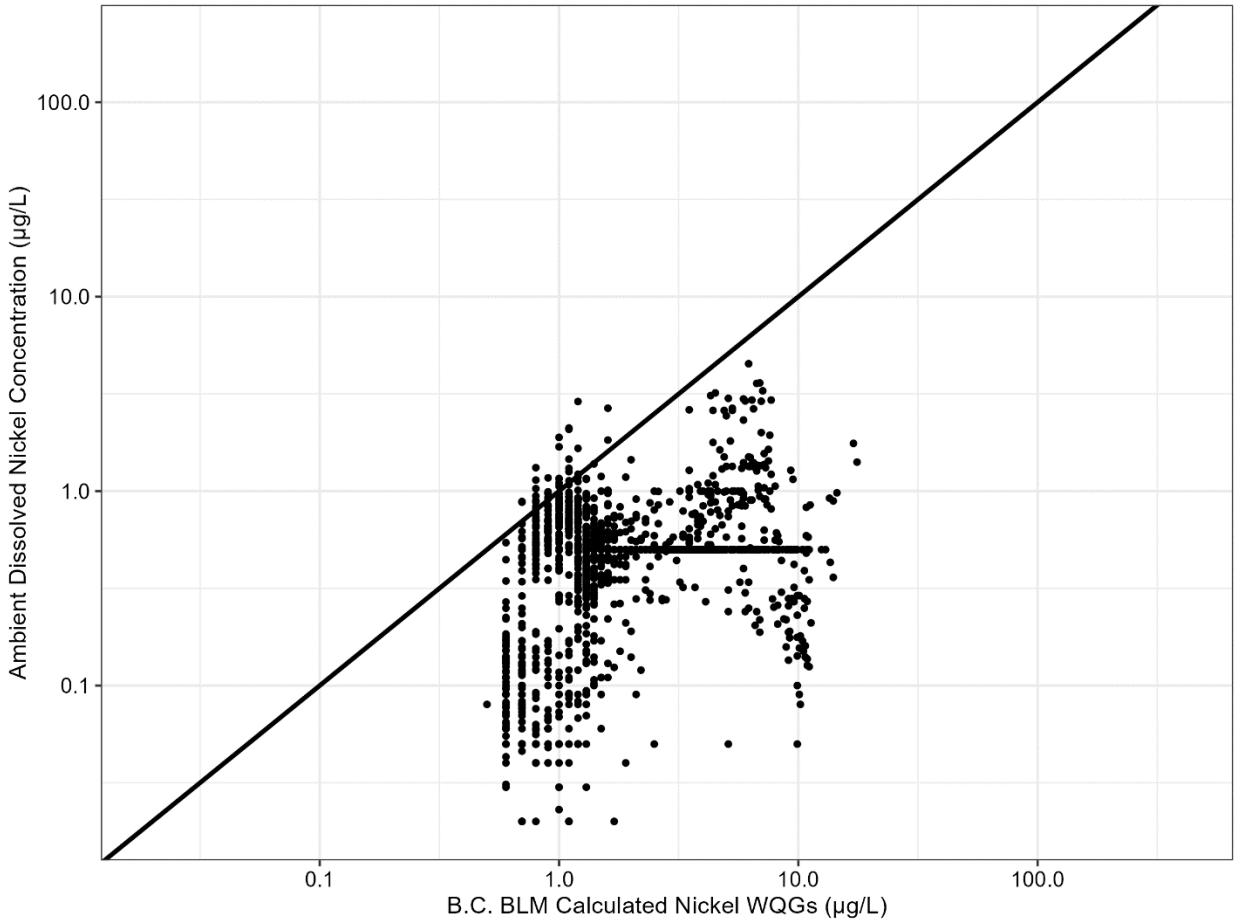

Figure 13.2. Ambient dissolved Ni concentrations compared to the 2022 chronic dissolved Ni WQGs.

Points above the solid 1:1 line represent exceedances. The minimum value for the 2022 chronic WQG is 0.5 µg/L dissolved Ni and the maximum value is 17.6 µg/L dissolved Ni based on the water chemistry data contained in the test Ni dataset. Both axes are on a log scale.

A comparison of background exceedances was repeated for each region (Table 13.1; Figures 13.3-13.7). Of the 33 records where dissolved Ni was higher than the revised WQG, 32 samples were in the Cariboo Region (3.1%) and one sample in the West Coast Region (2.4%). Note that there were no data from the Kootenay-Boundary, Northeast, and South Coast regions that met the criteria for inclusion in this analysis.

Table 13.2. Regional comparison of ambient Ni concentrations with the 1987 CCME total Ni WQG and the 2022 dissolved Ni WQG.

Region	Total Samples	Number of Exceedances	
		Previous 1987 CCME WQGs	B.C. BLM WQGs
Cariboo	1025	0	32 (3.1%)
Kootenay-Boundary	0	NA	NA
Northeast	0	NA	NA
Omineca	7	0	0
Skeena	198	0	0
South Coast	0	NA	NA
Thompson-Okanagan	200	0	0
West Coast	41	0	1 (2.4%)

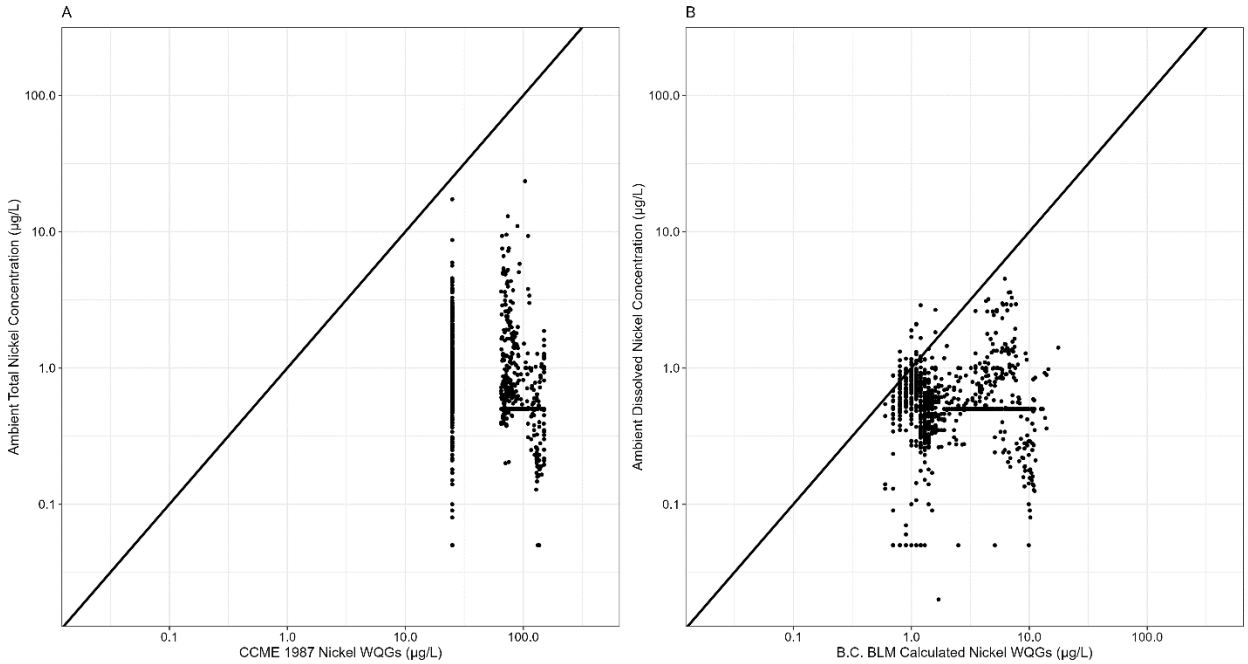

Figure 13.3. Comparison of previous (A) and new (B) WQGs vs ambient Ni concentrations in Cariboo Region.

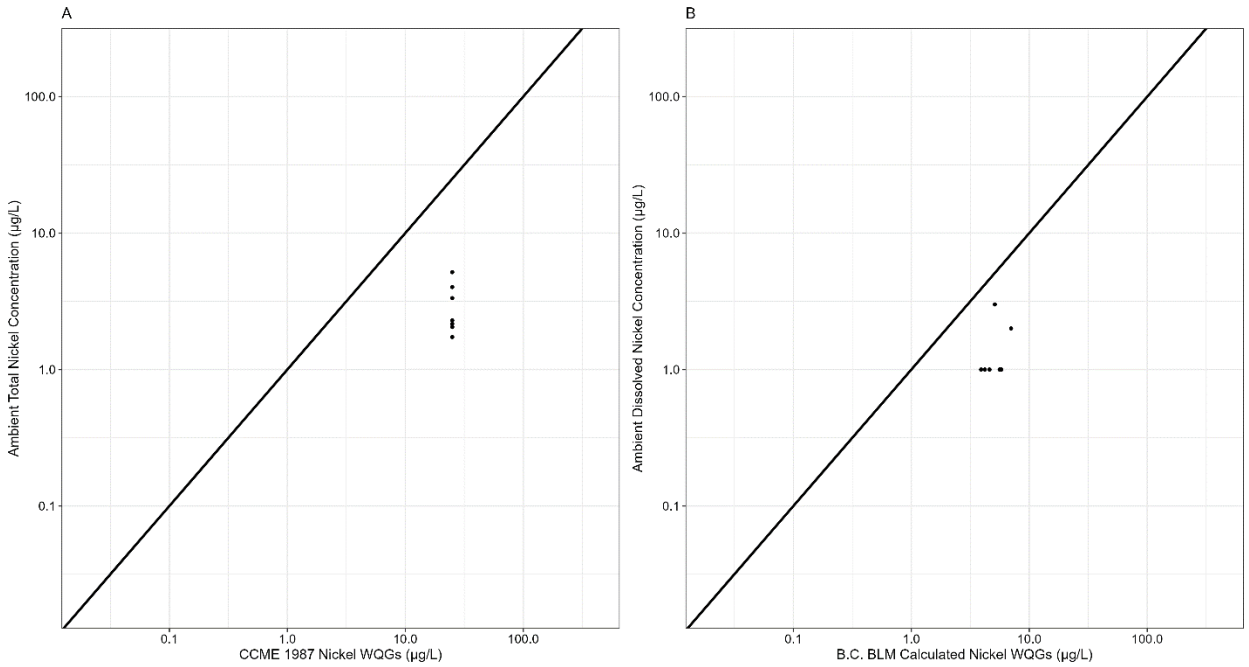

Figure 13.4. Comparison of previous (A) and new (B) WQGs vs ambient Ni concentrations in Omenica Region.

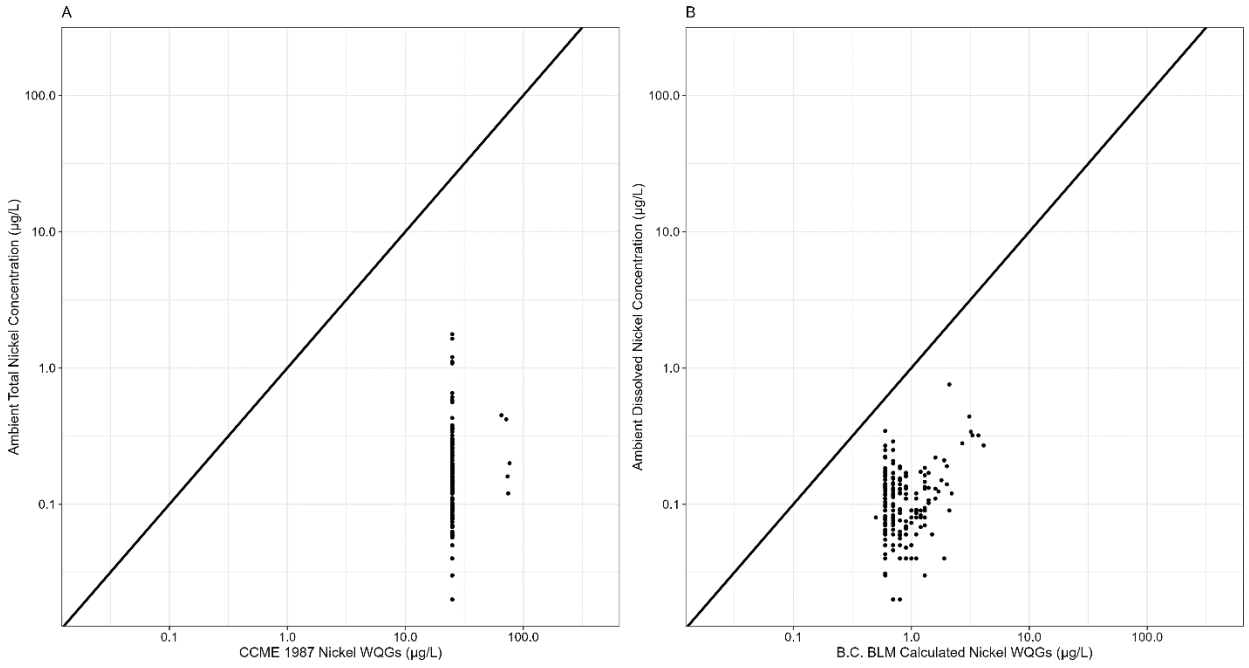

Figure 13.5. Comparison of previous (A) and new (B) WQGs vs ambient Ni concentrations in Skeena Region.

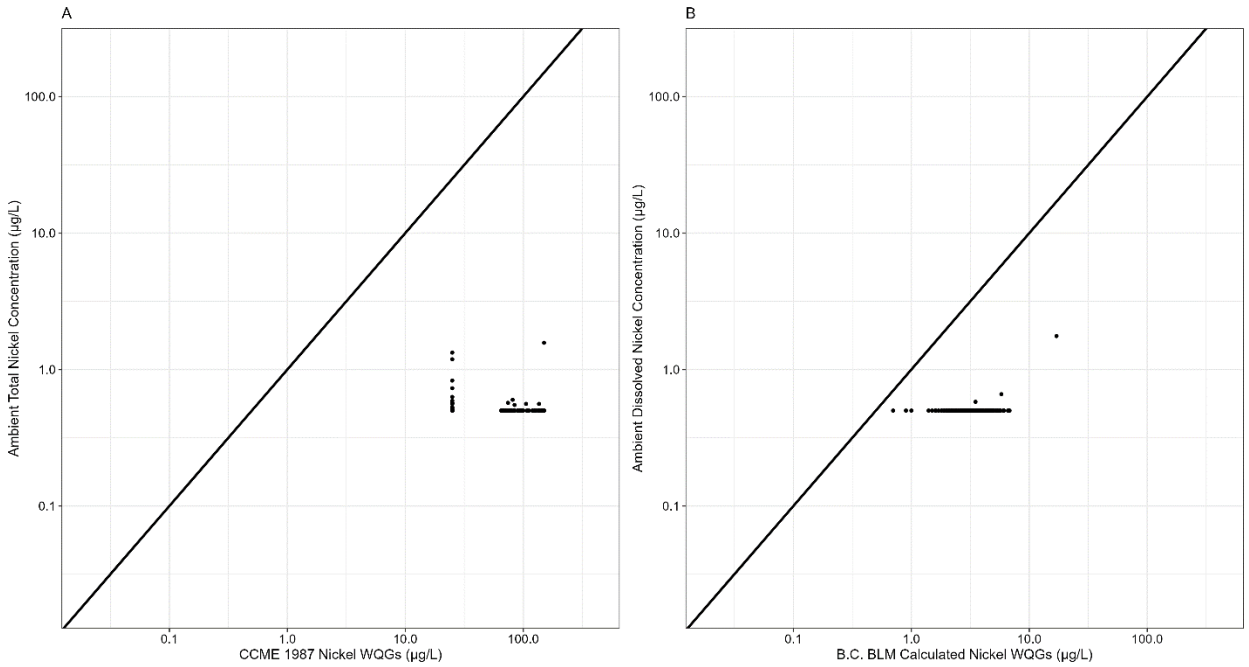

Figure 13.6. Comparison of previous (A) and new (B) WQGs vs ambient Ni concentrations in Thompson-Okanagan Region.

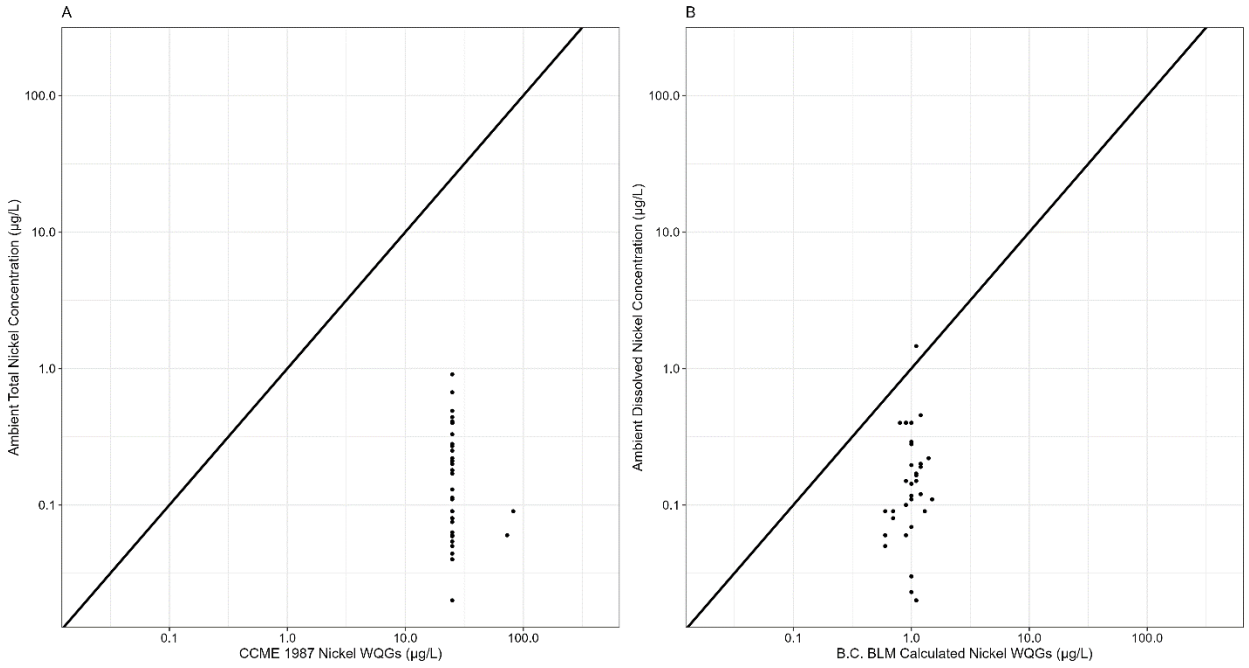

Figure 13.7. Comparison of previous (A) and new (B) WQGs vs ambient Ni concentrations in West Coast Region.

14. DATA GAPS AND RESEARCH NEEDS

Although Ni is considered a relatively well studied contaminant, there are still many unanswered questions. Major research and data needs include:

- Additional toxicity data with water chemistry information for:
    - freshwater algae, macrophytes and amphibians species;
    - sensitive yet ecologically relevant endpoints (e.g., olfactory toxicity and swim performance) for sensitive fish species;
- Further information on:
    - the effect of metal mixtures;
- Additional water quality data needed for running BC BLM in "full" mode, especially from lentic systems, to compare WQGs calculated using full and simplified BC BLM; and
- Environmental effects monitoring to determine if the guideline values calculated using BC BLM are protective of aquatic life in various water quality conditions.

REFERENCES

Adams, W., R. Blust, R. Dwyer, D. Mount, E. Nordheim, P. H. Rodriguez, and D. Spry. 2020. Bioavailability Assessment of Metals in Freshwater Environments: A Historical Review. Environmental Toxicology and Chemistry 39:48–59.

AECOM Technical Services, Inc. 2011. Acute toxicity of selected mixtures of nickel, cobalt, copper, and iron to freshwater algae (*Pseudokirchneriella subcapitata*) exposed in water under static test conditions: Series 5 (nickel only). Page 21. Fort Collins, CO.

ANZECC (Australia and New Zealand Environment and Conservation Council), 2000a. National Water Quality Management Strategy. Paper No. 4. Australian and New Zealand Guidelines for Fresh and Marine Water Quality. Volume 1.

ANZECC (Australia and New Zealand Environment and Conservation Council), 2000b. National Water Quality Management Strategy. Paper No. 4. Australian and New Zealand Guidelines for Fresh and Marine Water Quality. Volume 2.

Arnot, J. A., and F. A. P. C. Gobas. 2006. A review of bioconcentration factor (BCF) and bioaccumulation factor (BAF) assessments for organic chemicals in aquatic organisms. Environmental Reviews 14:257–297.

Athikesavan, S., S. Vincent, T. Ambrose, and B. Velmurugan. 2006. Nickel induced histopathological changes in the different tissues of freshwater fish, *Hypophthalmichthys molitrix* (Valenciennes). Journal of Environmental Biology 27:391–395.

ATSDR (Agency for Toxic Substances and Disease Registry). 2005. Toxicological profile for nickel: Update. Page 351 pp + Appendices. United States Department of Health and Human Services. Public Health Service, Atlanta, Georgia.

Besser, J. M., W. G. Brumbaugh, C. G. Ingersoll, C. D. Ivey, J. L. Kunz, N. E. Kemble, C. E. Schlekat, and E. R. Garman. 2013. Chronic toxicity of nickel-spiked freshwater sediments: Variation in toxicity among eight invertebrate taxa and eight sediments: Toxicity of Ni-spiked sediments. Environmental Toxicology and Chemistry 32:2495–2506.

Besser, J. M., R. Dorman, C. D. Ivey, D. Cleveland, and J. A. Steevens. 2020. Sensitivity of warm-water fishes and rainbow trout to selected contaminants. Bulletin of Environmental Contamination and Toxicology 104:321–326.

Birge, W. J., and J. A. Black. 1980. Aquatic toxicology of nickel. Pages 349–366 in J. O. Nriagu, editor. Nickel in the Environment. John Wiley & Sons, New York.

Birge, W. J., J. A. Black, J. F. Hobson, A. G. Westerman, and T. M. Short. 1984. Toxicological studies on aquatic contaminants originating from coal production and utilization: the induction of tolerance to silver in laboratory populations of fish and the chronic toxicity of nickel to fish early-life stages. Kentucky Water Resources Research Inst., Lexington (USA).

Blaylock, B. G., and M. L. Frank. 1979. A comparison of the toxicity of nickel to the developing eggs and larvae of carp (*Cyprinus carpio*). Bulletin of Environmental Contamination and Toxicology 21:604–611.

Brix, K. V., J. Keithly, D. K. DeForest, and J. Laughlin. 2004. Acute and chronic toxicity of nickel to rainbow trout (*Oncorhynchus mykiss*). Environmental Toxicology and Chemistry 23:2221–2228.

Brix, K. V., C. E. Schlekat, and E. R. Garman. 2017. The mechanisms of nickel toxicity in aquatic environments: An adverse outcome pathway analysis. Environmental Toxicology and Chemistry 36:1128–1137.

Buhl, K. J., and S. J. Hamilton. 1990. Comparative toxicity of inorganic contaminants released by placer mining to early life stages of salmonids. Ecotoxicology and Environmental Safety 20:325–342.

Buhl, K. J., and S. J. Hamilton. 1991. Relative sensitivity of early life stages of arctic grayling, coho salmon, and rainbow trout to nine inorganics. Ecotoxicology and Environmental Safety 22:184–197.

CCME (Canadian Council of Ministers of Environment). 2007. Protocol for the Derivation of Water Quality Guidelines for the Protection of Aquatic Life. Winnipeg, Manitoba: Canadian Council of Ministers of the Environment.

CCREM (Canadian Council of Resource and Environment Ministers). 1987. Canadian Water Quality Guidelines.

Chapman, G. A., S. Ota, and F. Recht. 1980. Effect of water hardness on the toxicity of metals to *Daphnia magna* (Status Report - Sept 1980). Corvallis Environmental Research Laboratory, Office of Research and Development, U.S. Environmental Protection Agency, Corvallis, OR.

Chau, Y. K., and O. T. R. Kulikovsky-Cordeiro. 1995. Occurrence of nickel in the Canadian environment. Environmental Reviews 3:95–120.

Chowdhury, M. J., C. Bucking, and C. M. Wood. 2008. Is nickel an essential metal for aquatic animals? Integrated Environmental Assessment and Management 4:266–267.

Cornwall, H. R. 1966. Nickel deposits of North America, Geological Survey Bulleting 1223. United States Government Printing Office, Washington, USA.

Crémazy, A., K. V. Brix, D. S. Smith, W. Chen, M. Grosell, C. E. Schlekat, E. R. Garman, E. T. Middleton, and C. M. Wood. 2020. A mystery tale: Nickel is fickle when snails fail—Investigating the variability in Ni toxicity to the great pond snail. Integrated Environmental Assessment and Management 16:983–997.

DalCorso, G. 2012. Heavy Metal Toxicity in Plants. Pages 1–25 in A. Furini, editor. Plants and Heavy Metals. Springer Netherlands, Dordrecht.

Dave, G., and R. Xiu. 1991. Toxicity of mercury, copper, nickel, lead, and cobalt to embryos and larvae of zebrafish, *Brachydanio rerio*. Archives of Environmental Contamination and Toxicology 21:126–134.

De Schamphelaere, K., L. Van Laer, N. Deleebeeck, B. Muyssen, F. Degryse, E. Smolders, and C. Janssen. 2006. Nickel speciation and ecotoxicity in European natural surface waters: development, refinement, and validation of bioavailability models. Ghent University, Gent, Belgium.

Deleebeeck, N. M. E., K. A. C. De Schamphelaere, and C. R. Janssen. 2007a. A bioavailability model predicting the toxicity of nickel to rainbow trout (*Oncorhynchus mykiss*) and fathead minnow (*Pimephales promelas*) in synthetic and natural waters. Ecotoxicology and Environmental Safety 67:1–13.

Deleebeeck, N. M. E., B. T. A. Muyssen, F. De Laender, C. R. Janssen, and K. A. C. De Schamphelaere. 2007b. Comparison of nickel toxicity to cladocerans in soft versus hard surface waters. Aquatic Toxicology 84:223–235.

Deleebeeck, N. M. E., K. A. C. De Schamphelaere, D. G. Heijerick, B. T. A. Bossuyt, and C. R. Janssen. 2008a. The acute toxicity of nickel to *Daphnia magna*: Predictive capacity of bioavailability models in artificial and natural waters. Ecotoxicology and Environmental Safety 70:67–78.

Deleebeeck, N. M. E., K. A. C. De Schamphelaere, and C. R. Janssen. 2008b. A novel method for predicting chronic nickel bioavailability and toxicity to *Daphnia magna* in artificial and natural waters. Environmental Toxicology and Chemistry 27:2097–2107.

Deleebeeck, N. M. E., F. De Laender, V. A. Chepurnov, W. Vyverman, C. R. Janssen, and K. A. C. De Schamphelaere. 2009a. A single bioavailability model can accurately predict Ni toxicity to green microalgae in soft and hard surface waters. Water Research 43:1935–1947.

Deleebeeck, N. M. E., K. A. C. De Schamphelaere, and C. R. Janssen. 2009b. Effects of Mg2+ and H+ on the toxicity of Ni2+ to the unicellular green alga *Pseudokirchneriella subcapitata*: Model development and validation with surface waters. Science of The Total Environment 407:1901–1914.

Di Toro, D. M., H.E. Allen, H.L. Bergman, J.S. Meyer, P.R. Paquin, and R.C. Santore. 2001. Biotic ligand model of the acute toxicity of metals. 1. Technical basis. Environmental Toxicology and Chemistry, 20: 2383-2396.

Doig, L. E., and K. Liber. 2006. Influence of dissolved organic matter on nickel bioavailability and toxicity to *Hyalella azteca* in water-only exposures. Aquatic Toxicology 76:203–216.

ECCC (Environment & Climate Change Canada). 2018. Nickel toxicology project report: Chinook 28-day chronic toxicity test.

ENV (Ministry of Environment and Climate Change Strategy) 2019a. Derivation of water quality guidelines for the protection of aquatic life in British Columbia. Water Quality Guideline Series, WQG-06. Prov. B.C, Victoria, B.C.

ENV (British Columbia Ministry of Environment and Climate Change Strategy), 2019b. Copper Water Quality Guideline for the Protection of Freshwater Aquatic Life-Technical Report. Water Quality Guideline Series, WQG-03-1. Prov. B.C., Victoria B.C.

ENV (Ministry of Environment and Climate Change Strategy). 2020. Section C: Metals. Page British Columbia Environmental Laboratory Manual (J. Austin, ed.). Victoria, B.C.

European Chemicals Bureau. 2008. European Union risk assessment report: Nickel. Ispra, Italy: European Commission, Joint Research Centre, European Chemicals Bureau.

EC (European Commission). 2011. Nickel and its compounds. Nickel EQS dossier.

Fort, D. J., R. L. Rogers, J. H. Thomas, W. A. Hopkins, and C. Schlekat. 2006. Comparative developmental toxicity of nickel to *Gastrophryne carolinensis, Bufo terrestris*, and *Xenopus laevis*. Archives of Environmental Contamination and Toxicology 51:703–710.

Giattina, J. D., R. R. Garton, and D. G. Stevens. 1982. Avoidance of copper and nickel by rainbow trout as monitored by a computer-based data acquisition system. Transactions of the American Fisheries Society 111:491–504.

Gopalapillai, Y., B. Hale, and B. Vigneault. 2013. Effect of major cations (Ca_2^+, Mg_2^+ , Na^+ , K^+) and anions (SO_4^{2-}, Cl^-, NO_3^-) on Ni accumulation and toxicity in aquatic plant (*Lemna minor L.*): Implications for Ni risk assessment. Environmental Toxicology and Chemistry 32:810–821.

Green-Pedersen, H., B. T. Jensen, and N. Pind. 1997. Nickel adsorption on MnO_2, $Fe(OH)_3$, montmorillonite, humic acid and calcite: A comparative study. Environmental Technology 18:807–815.

Hoang, T. C., J. R. Tomasso, and S. J. Klaine. 2004. Influence of water quality and age on nickel toxicity to fathead minnows (*Pimephales promelas*). Environmental Toxicology and Chemistry 23:86–92.

Huston, C., and E. Juarez-Colunga. 2009. Guidelines for computing summary statistics for data-sets containing non-detects. Written for the Bulkley Valley Research Center with assistance from the B.C. Ministry of Environment.

Ivey, C. D., J. M. Besser, C. G. Ingersoll, N. Wang, D. C. Rogers, S. Raimondo, C. R. Bauer, and E. J. Hammer. 2017. Acute sensitivity of the vernal pool fairy shrimp, *Branchinecta lynchi* (Anostraca; Branchinectidae), and surrogate species to 10 chemicals: Acute toxicity of 10 chemicals to fairy shrimp. Environmental Toxicology and Chemistry 36:797–806.

Keithly, J., J. A. Brooker, D. K. Deforest, B. K. Wu, and K. V. Brix. 2004. Acute and chronic toxicity of nickel to a cladoceran (*Ceriodaphnia dubia*) and an amphipod (*Hyalella azteca*). Environmental Toxicology and Chemistry 23:691–696.

Keller, A. E., and S. G. Zam. 1991. The acute toxicity of selected metals to the freshwater mussel, *Anodonta imbecilis*. Environmental Toxicology and Chemistry 10:539–546.

Klemish, J. L., S. J. Bogart, A. Luek, M. J. Lannoo, and G. G. Pyle. 2018. Nickel toxicity in wood frog tadpoles: Bioaccumulation and sublethal effects on body condition, food consumption, activity, and chemosensory function. Environmental Toxicology and Chemistry 37:2458–2466.

Kozlova, T., C. M. Wood, and J. C. McGeer. 2009. The effect of water chemistry on the acute toxicity of nickel to the cladoceran *Daphnia pulex* and the development of a biotic ligand model. Aquatic Toxicology 91:221–228.

Kwong, R. W. M., and S. Niyogi. 2009. The interactions of iron with other divalent metals in the intestinal tract of a freshwater teleost, rainbow trout (*Oncorhynchus mykiss*). Comparative Biochemistry and Physiology Part C: Toxicology & Pharmacology 150:442–449.

Lapointe, D., and P. Couture. 2010. Accumulation and effects of nickel and thallium in early-life stages of fathead minnows (*Pimephales promelas*). Ecotoxicology and Environmental Safety 73:572–578.

Lee, L. 2017. NADA: Nondetects and data analysis for environmental data. https://cran.r-project.org/web/packages/NADA/index.html.

Leonard, E. M., S. R. Nadella, C. B. and, and C. M. Wood. 2009. Characterization of dietary Ni uptake in the rainbow trout, Oncorhynchus mykiss. Aquatic Toxicology 93:205–216.

Leonard, E. M., and C. M. Wood. 2013. Acute toxicity, critical body residues, Michaelis–Menten analysis of bioaccumulation, and ionoregulatory disturbance in response to waterborne nickel in four invertebrates: *Chironomus riparius, Lymnaea stagnalis, Lumbriculus variegatus* and *Daphnia pulex.* Comparative Biochemistry and Physiology Part C: Toxicology & Pharmacology 158:10–21.

Liber, K., L. E. Doig, and S. L. White-Sobey. 2011. Toxicity of uranium, molybdenum, nickel, and arsenic to *Hyalella azteca* and *Chironomus dilutus* in water-only and spiked-sediment toxicity tests. Ecotoxicology and Environmental Safety 74:1171–1179.

Macoustra, G. K., D. F. Jolley, J. L. Stauber, D. J. Koppel, and A. Holland. 2021. Speciation of nickel and its toxicity to Chlorella sp. in the presence of three distinct dissolved organic matter (DOM). Chemosphere 273.

Mandal, R., N. M. Hassan, J. Murimboh, C. L. Chakrabarti, M. H. Back, U. Rahayu, and D. R. S. Lean. 2002. Chemical speciation and toxicity of nickel species in natural waters from the Sudbury area (Canada). Environmental Science & Technology 36:1477–1484.

Mano, H., N. Shinohara, and W. Naito. 2020. Reproduction sensitivity of five Daphnia species to nickel. Journal of Water and Environment Technology 18:372–382.

Martínez-Ruiz, E. B., and F. Martínez-Jerónimo. 2015. Nickel has biochemical, physiological, and structural effects on the green microalga *Ankistrodesmus falcatus*: An integrative study. Aquatic Toxicology 169:27–36.

Mathis, B. J., and T. F. Cummings. 1973. Selected metals in sediments, water, and biota in the Illinois River. Journal Water Pollution Control Federation 45:1573–1583.

McGeer, J. C., K. V. Brix, J. M. Skeaff, D. K. DeForest, S. I. Brigham, W. J. Adams, and A. Green. 2003. Inverse relationship between bioconcentration factor and exposure concentration for metals: Implications for hazard assessment of metals in the aquatic environment. Environmental Toxicology and Chemistry 22:1017–1037.

Meyer, J. S., R. C. Santore, J. P. Bobbitt, L. D. Debrey, C. J. Boese, P. R. Paquin, H. E. Allen, H. L. Bergman, and D. M. DiToro. 1999. Binding of nickel and copper to fish gills predicts toxicity when water hardness varies, but free-ion activity does not. Environmental Science & Technology 33:913–916.

Muyssen, B. T. A., K. V. Brix, D. K. DeForest, and C. R. Janssen. 2004. Nickel essentiality and homeostasis in aquatic organisms. Environmental Reviews 12:113–131.

Nagai, T., and K. A. C. De Schamphelaere. 2016. The effect of binary mixtures of zinc, copper, cadmium, and nickel on the growth of the freshwater diatom *Navicula pelliculosa* and comparison with mixture toxicity model predictions: Metal mixture effects on a diatom. Environmental Toxicology and Chemistry 35:2765–2773.

Natural Resources Canada. 2021. Nickel facts. Web Page, Natural Resources Canada. https://www.nrcan.gc.ca/our-natural-resources/minerals-mining/minerals-metals-facts/nickel-facts/20519.

Niyogi, S., and C. M. Wood. 2004. Biotic ligand model, a flexible tool for developing site-specific water quality guidelines for metals. Environmental Science & Technology 38:6177–6192.

Niyogi, S., K. V. Brix, and M. Grosell. 2014. Effects of chronic waterborne nickel exposure on growth, ion homeostasis, acid-base balance, and nickel uptake in the freshwater pulmonate snail, *Lymnaea stagnalis*. Aquatic Toxicology 150:36–44.

Nys, C., J. Asselman, J. D. Hochmuth, C. R. Janssen, R. Blust, E. Smolders, and K. A. C. De Schamphelaere. 2015. Mixture toxicity of nickel and zinc to *Daphnia magna* is noninteractive at low effect sizes but becomes synergistic at high effect sizes. Environmental Toxicology and Chemistry 34:1091–1102.

Nys, C., C. R. Janssen, P. Van Sprang, and K. A. C. De Schamphelaere. 2016. The effect of pH on chronic aquatic nickel toxicity is dependent on the pH itself: Extending the chronic nickel bioavailability models. Environmental Toxicology and Chemistry 35:1097–1106.

Nys, C., T. Van Regenmortel, C. R. Janssen, R. Blust, E. Smolders, and K. A. C. De Schamphelaere. 2017. Comparison of chronic mixture toxicity of nickel-zinc-copper and nickel-zinc-copper-cadmium mixtures between *Ceriodaphnia dubia* and *Pseudokirchneriella subcapitata*. Environmental Toxicology and Chemistry 36:1056–1066.

von der Ohe, P. C., and M. Liess. 2004. Relative sensitivity distribution of aquatic invertebrates to organic and metal compounds. Environmental Toxicology and Chemistry 23:150.

Oláh, V., A. Hepp, and I. Mészáros. 2015. Comparative study on the sensitivity of turions and active fronds of giant duckweed (*Spirodela polyrhiza* (L.) Schleiden) to heavy metal treatments. Chemosphere 132:40–46.

OMOEE (Ontario Ministry of Environment and Energy), 1994. Water Management: Policies: Guidelines: Provincial Water Quality Objectives of the Ministry of Environment and Energy.

Palermo, F. F., W. E. Risso, J. D. Simonato, and C. B. R. Martinez. 2015. Bioaccumulation of nickel and its biochemical and genotoxic effects on juveniles of the neotropical fish *Prochilodus lineatus*. Ecotoxicology and Environmental Safety 116:19–28.

Pane, E. F., J. G. Richards, and C. M. Wood. 2003a. Acute waterborne nickel toxicity in the rainbow trout (*Oncorhynchus mykiss*) occurs by a respiratory rather than ion regulatory mechanism. Aquatic Toxicology 63:65–82.

Pane, E. F., C. Smith, J. C. McGeer, and C. M. Wood. 2003b. Mechanisms of acute and chronic waterborne nickel toxicity in the freshwater cladoceran, *Daphnia magna*. Environmental Science & Technology 37:4382–4389.

Pane, E. F., A. Haque, G. G. Goss, and C. M. Wood. 2004a. The physiological consequences of exposure to chronic, sublethal waterborne nickel in rainbow trout (*Oncorhynchus mykiss*): Exercise vs resting physiology. Journal of Experimental Biology 207:1249–1261.

Pane, E. F., A. Haque, and C. M. Wood. 2004b. Mechanistic analysis of acute, Ni-induced respiratory toxicity in the rainbow trout (*Oncorhynchus mykiss*): an exclusively branchial phenomenon. Aquatic Toxicology 69:11–24.

Pane, E. F., C. Bucking, M. Patel, and C. M. Wood. 2005. Renal function in the freshwater rainbow trout (*Oncorhynchus mykiss*) following acute and prolonged exposure to waterborne nickel. Mechanisms in Metal Toxicology 72:119–133.

Pane, E. F., C. N. Glover, M. Patel, and C. M. Wood. 2006. Characterization of Ni transport into brush border membrane vesicles (BBMVs) isolated from the kidney of the freshwater rainbow trout (*Oncorhynchus mykiss*). Biochimica et Biophysica Acta (BBA) - Biomembranes 1758:74–84.

Paquin, P.R., J.W. Gorsuch, S. Apte, G.E. Batley, K.C. Bowles, P.G.C. Campbell, C. G. Delos, D. M. Di Toro, R.L. Dwyer, F. Galvez, R.W. Gensemer, G.G. Goss, C. Hogstrand, C.R. Janssen, J.C. McGeer, R.B. Naddy, R.C. Playle, R.C. Santore, U. Schneider, W. A. Stubblefield, C.M. Wood, and K.B. Wu. 2002. The biotic ligand model: a historical overview. Comparative Biochemistry and Physiology C: Toxicology & Pharmacology, 133: 3-35.

Pereira, C. M. S., D. Deruytter, R. Blust, and K. A. C. De Schamphelaere. 2017. Effect of temperature on chronic toxicity of copper, zinc, and nickel to *Daphnia magna*. Environmental Toxicology and Chemistry 36:1909–1916.

Pereira, C. M. S., G. Everaert, R. Blust, and K. A. C. De Schamphelaere. 2018. Multigenerational effects of nickel on *Daphnia magna* depend on temperature and the magnitude of the effect in the first generation. Environmental Toxicology and Chemistry 37:1877–1888.

Pérez, E., and T. C. Hoang. 2018. Responses of *Daphnia magna* to chronic exposure of cadmium and nickel mixtures. Chemosphere 208:991–1001.

Peters, A., G. Merrington, C. Schlekat, K. De Schamphelaere, J. Stauber, G. Batley, A. Harford, R. van Dam, C. Pease, T. Mooney, M. Warne, C. Hickey, P. Glazebrook, J. Chapman, R. Smith, and R. Krassoi. 2018. Validation of the nickel biotic ligand model for locally relevant species in Australian freshwaters: Nickel BLM in Australia. Environmental Toxicology and Chemistry 37:2566–2574.

Phipps, T., S. L. Tank, J. Wirtz, L. Brewer, A. Coyner, L. S. Ortego, and A. Fairbrother. 2002. Essentiality of nickel and homeostatic mechanisms for its regulation in terrestrial organisms. Environmental Reviews 10:209–261.

Pickering, Q. H. 1974. Chronic toxicity of nickel to the fathead minnow. Journal Water Pollution Control Federation 46:760–765.

Popp, A., W. G. Cope, M. A. McGregor, T. J. Kwak, T. Augspurger, J. F. Levine, and L. Koch. 2018. A Comparison of the chemical sensitivities between in vitro and in vivo propagated juvenile freshwater mussels: Implications for standard toxicity testing. Environmental Toxicology and Chemistry 37:3077–3085.

Puttaswamy, N., and K. Liber. 2012. Influence of inorganic anions on metals release from oil sands coke and on toxicity of nickel and vanadium to *Ceriodaphnia dubia*. Chemosphere 86:521–529.

Pyle, G., and P. Couture. 2011. 5 - Nickel. Pages 253–289 in C. M. Wood, A. P. Farrell, and C. J. Brauner, editors. Fish Physiology. Academic Press.

Pyle, G. G. 2000. The toxicity and bioavailability of nickel and molybdenum to standard toxicity-test fish species and fish species found in northern Canadian lakes. Department of Biology, University of Saskatchewan, Saskatoon.

Pyle, G. G., S. M. Swanson, and D. M. Lehmkuhl. 2002. The influence of water hardness, pH, and suspended solids on nickel toxicity to larval fathead minnows (*Pimephales promelas*). Water, Air, and Soil Pollution 133:215–226.

R Core Team. 2022. The R Project for Statistical Computing. https://www.r-project.org/.

Ragsdale, S. W. 1998. Nickel biochemistry. Current Opinion in Chemical Biology 2:208–215.

Ragsdale, S. W. 2011. Nickel enzymes & cofactors. Pages 3378–3393 in R. B. King, editor. Encyclopedia of inorganic chemistry. John Wiley, New York.

Richter, R. O., and T. L. Theis. 1980. Nickel speciation in a soil/water system. Pages 189–202 in J. O. Nriagu, editor. Nickel in the Environment. John Wiley & Sons, New York.

Santore, R. C., D. M. Di Toro, P. R. Paquin, H. E. Allen, and J. S. Meyer. 2001. Biotic ligand model of the acute toxicity of metals. 2. Application to acute copper toxicity in freshwater fish and Daphnia. Environmental Toxicology and Chemistry, 20: 2397-2402.

Santore, R. C., R. Mathew, P. R. Paquin, and D. Di Toro, 2002. Application of the biotic ligand model to predicting zinc toxicity to rainbow trout, fathead minnow, and *Daphnia magna*. Comparative Biochemistry and Physiology - C: Toxicology & Pharmacology, 133: 271-285.

Santore, R. C., K. Croteau, A. C. Ryan, C. Schlekat, E. Middleton, E. Garman, and T. Hoang. 2021. A review of water quality factors that affect nickel bioavailability to aquatic organisms: Refinement of the biotic ligand model for nickel in acute and chronic exposures. Environmental Toxicology and Chemistry 40:2121–2134.

Santos, F. M., L. P. Mazur, D. A. Mayer, V. J. P. Vilar, and J. C. M. Pires. 2019. Inhibition effect of zinc, cadmium, and nickel ions in microalgal growth and nutrient uptake from water: An experimental approach. Chemical Engineering Journal 366:358–367.

Schlekat, C. E., E. Van Genderen, K. A. C. De Schamphelaere, P. M. C. Antunes, E. C. Rogevich, and W. A. Stubblefield. 2010. Cross-species extrapolation of chronic nickel biotic ligand models. Science of The Total Environment 408:6148–6157.

Schubauer-Berigan, M. K., J. R. Dierkes, P. D. Monson, and G. T. Ankley. 1993. pH-Dependent toxicity of Cd, Cu, Ni, Pb and Zn to *Ceriodaphnia dubia, Pimephales promelas, Hyalella azteca* and *Lumbriculus variegatus*. Environmental Toxicology and Chemistry 12:1261–1266.

Soucek, D. J., A. Dickinson, C. Schlekat, E. Van Genderen, and E. J. Hammer. 2020. Acute and chronic toxicity of nickel and zinc to a laboratory cultured Mayfly (*Neocloeon triangulifer*) in aqueous but fed exposures. Environmental Toxicology and Chemistry 39:1196–1206.

U.S. EPA (Environmental Protection Agency). 1985. National Primary Drinking Water Regulations; Volatile Synthetic Organic Chemicals; Final Rule and Proposed Rule. Federal Register. Vol. 50, No 219, page 46902. November 13, 1985.

U.S. EPA (Environmental Protection Agency). 1994. Method 200.8 Determination of trace elements in waters and wastes by inductively coupled plasma - mass spectrometry. Revision 5.4.

U.S. EPA (Environmental Protection Agency). 1996. 1995 updates: Water quality criteria documents for the protection of aquatic life in ambient water. Office of Water, Washington, DC.

Van Regenmortel, T., and K. A. C. De Schamphelaere. 2018. Mixtures of Cu, Ni, and Zn act mostly noninteractively on *Pseudokirchneriella subcapitata* growth in natural waters: Toxicity of Cu-Ni-Zn mixtures to *P. subcapitata*. Environmental Toxicology and Chemistry 37:587–598.

Wang, N., C. D. Ivey, C. G. Ingersoll, W. G. Brumbaugh, D. Alvarez, E. J. Hammer, C. R. Bauer, T. Augspurger, S. Raimondo, and M. C. Barnhart. 2017. Acute sensitivity of a broad range of freshwater mussels to chemicals with different modes of toxic action: Freshwater mussel sensitivity to different chemicals. Environmental Toxicology and Chemistry 36:786–796.

Wang, N., J. L. Kunz, D. M. Cleveland, J. A. Steevens, E. J. Hammer, E. Van Genderen, A. C. Ryan, and C. E. Schlekat. 2020. Evaluation of acute and chronic toxicity of nickel and zinc to 2 sensitive freshwater benthic invertebrates using refined testing methods. Environmental Toxicology and Chemistry 39:2256–2268.

Water Security Agency, 2015. Surface Water Quality Objectives. Water Security Agency EPB356.

Wu, K. B., P.R. Paquin, V. Navab, R. Mathew, R.C. Santore, and D.M. DiToro. 2003. Development of a biotic ligand model for nickel: Phase I. Water Environment Research Foundation, Report, (01-ECO).

Xue, H. B., S. Jansen, A. Prasch, and L. Sigg. 2001. Nickel speciation and complexation kinetics in freshwater by ligand exchange and DPCSV. Environmental Science & Technology 35:539–546.

Zheng, G.-H., C.-M. Liu, J.-M. Sun, Z.-J. Feng, and C. Cheng. 2014. Nickel-induced oxidative stress and apoptosis in Carassius auratus liver by JNK pathway. Aquatic Toxicology 147:105–111.