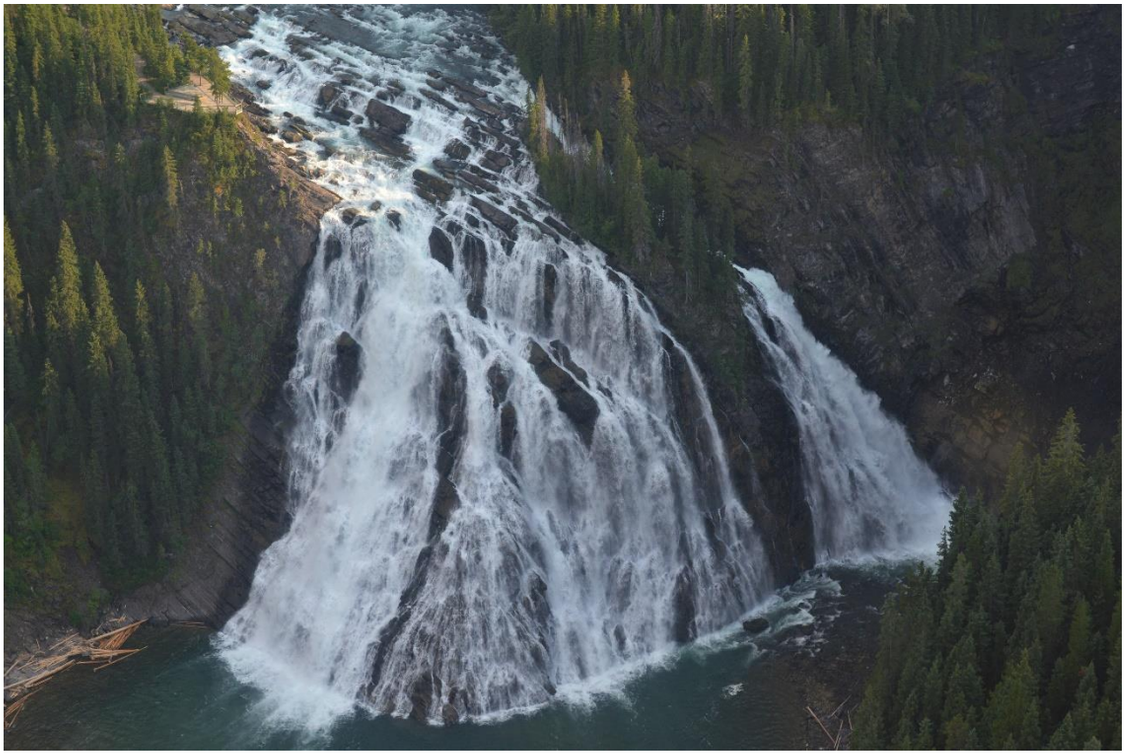
WATER QUALITY GUIDELINE SERIES

Derivation of Water Quality Guidelines for the Protection of Aquatic Life in British Columbia

Ministry of Environment and Climate Change Strategy

Water Protection & Sustainability Branch



BRITISH COLUMBIA

No. WQG-06

The Water Quality Guideline Series is a collection of British Columbia (B.C.) Ministry of environment and climate change strategy water quality guidelines. Water quality guidelines are developed for a variety of water uses: aquatic life; drinking water sources; recreation; agriculture; 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 British Columbia (B.C.) Ministry of Environment and Climate Change Strategy (ENV) develops province-wide ambient water quality guidelines (WQG) for substances or physical attributes that are important in both the fresh and marine surface waters of B.C. The ENV defines a WQG as a scientifically-derived numerical concentration or narrative statement considered to be protective of designated values in ambient conditions. The WQGs are set after considering the scientific literature, results from toxicity tests, WQGs from other jurisdictions, and background conditions in B.C.

Once approved, WQGs constitute official ENV policy and must be considered in any decision affecting water quality made within the ENV; they do not, however, have any direct legal standing. WQGs provide information and policy direction to those making decisions affecting water quality, provided they do not allow legislated effluent standards to be exceeded. WQGs can be used as the basis for authorized waste discharge limits. These limits are set out in waste management permits, approvals, plans, or operating certificates, which are legally enforceable.

B.C. WQGs are generic provincial recommendations that are based on the most current scientific information available at the time of their derivation. Both long-term chronic and short-term acute WQGs are derived if sufficient toxicological data are available. Long-term chronic WQGs are intended to be protective of all forms of aquatic life (all species, all life stages including multi-generational) from lethal and negative sub-lethal effects over indefinite exposures. Short-term guidelines are intended to protect aquatic organisms against severe effects such as lethality due to short-term intermittent and/or transient exposures to contaminants (e.g. spill events; infrequent releases of short-lived/non-persistent substances).

With this document, B.C. is adopting a statistically based approach, the species sensitivity distribution (SSD). The SSD approach is used by the Canadian Council of Ministers of the Environment (CCME), the National Standards and Guidelines Office within Environment Canada, as well as multiple international jurisdictions including the European Union, Australia and New Zealand. WQG development is a resource intense process; by adopting the SSD approach, B.C. will enhance its opportunities to collaborate and share guidelines with other jurisdictions.

B.C. is a member of the CCME and participated in the development of the CCME aquatic life WQG derivation protocol (CCME, 2007) but maintains an independent B.C.-specific protocol which is consistent with B.C. policy objectives. This is necessary given B.C.'s recognition of the uncertainty associated with WQG derivation and differences in objectives for short-term WQGs compared with CCME. Given these differences in policy, it was not possible to directly adopt the 2007 CCME protocol, however to encourage collaboration on a national basis the B.C. protocol is harmonized with the CCME protocol wherever possible.

This document includes procedures for developing three types of WQGs: Type A1, Type A2 and Type B. Type A WQGs are derived using SSD and are distinguished based on the quality and quantity of the available data. Type A1 is a novel approach unique to B.C. It has the highest data standards and is designed to minimize uncertainty. Type A2 WQG has an increased breadth of acceptable data and is congruent with the CCME Type A WQG (CCME, 2007). Type B WQG is based on the extrapolation from the lowest available and acceptable toxicity endpoint and is congruent with the CCME Type B2 guideline.

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LIST OF ABBREVIATIONS

AF: Assessment Factor

ASTM: American Society for Testing and Materials

B.C.: British Columbia

BLM: Biotic ligand model

CCME: Canadian Council of Ministers of the Environment

CRAN: Comprehensive R Archive Network

DOC: Dissolved organic carbon

ECCC: Environment and Climate Change Canada

EC: European Commission

EC_x: Effect concentration affecting X% of a test population

ENV: British Columbia Ministry of Environment and Climate Change Strategy

HC_x: Hazard concentration likely to affect X% of the species

IC_x: Inhibitory concentration affecting X% of a test population

LC_x: Lethal concentration affecting X% of a test population

LOEC: Lowest observed effect concentration

MATC: Maximum allowable toxicant concentration

MDL: Method detection limit

NOEC: No observed effect concentration

PNEC: Predicted no-effect concentration

SBEB: Science-based environmental benchmark

SSD: Species sensitivity distribution

TMF: Toxicity-modifying factor

USEPA: United States Environmental Protection Agency

WQG: Water quality guideline

WQO: Water quality objective

DEFINITIONS

Ambient: refers to the surrounding environmental conditions outside the zone in which water quality may be directly affected by a waste discharge or source of contamination.

Assessment Factor: a mathematical adjustment to guideline values to account for uncertainty.

Background Concentrations: refers to the concentration of a naturally occurring substance that has not been (or very minimally) altered through anthropogenic activity.

Bioaccumulation: the process by which chemical substances are accumulated by aquatic organisms from exposure to water, sediments, or soil directly or through consumption of food containing the chemicals (CCME 1999a).

Bioconcentration: the process by which contaminants are directly taken up by organisms from the medium in which they live (CCME 1999a).

Biomagnification: the increase in tissue concentrations of accumulated chemicals from one trophic level to the next (i.e. organisms contain higher concentrations of the substance than their food sources) (CCME 1999a).

Effective Concentration (EC_X): the concentration affecting X% of the population within *T* amount of time.

Hazard Concentration (HC_x): the concentration of a contaminant estimated to effect X% of species.

Inhibiting Concentration (IC_X): the concentration causing an X% inhibition in exposed individuals within *T* amount of time.

LC_X: the concentration which causes X% mortality in the exposed population within *T* amount of time.

Lowest Observed Effect Concentration (LOEC): the lowest tested concentration of a substance that has been reported to have an effect on organisms tested.

Maximum Acceptable Toxicant Concentration (MATC): the MATC is calculated as the geometric mean of the NOEC and LOEC for a chronic level exposure.

No Observed Effect Concentration (NOEC): the highest tested concentration of a substance that has been reported to have no effect on organisms tested.

Science-Based Environmental Benchmark (SBEB): a quantifiable receiving environment parameter or attribute developed by a qualified professional through a rigorous scientific process, with the intent to guide management decisions and mitigative actions for a regulated activity at a specific location. A SBEB is developed to support the impact assessment for a specific effluent discharge decision (ENV 2016).

Toxicity-Modifying Factor: an environmental factor (physical, chemical, or biological) that modifies the exposure and/or behavior of chemical substances and associated toxicity to aquatic animals and plants.

Water Quality Guideline (WQG): a scientifically-derived numerical concentration or narrative statement considered to be protective of the water values and uses in generic ambient conditions. Long-term WQGs are protective against chronic effects resulting from long-term exposures, while short-term WQGs are protective against acute effects resulting from short-term intermittent or transient exposures.

Water Quality Objective (WQO): a scientifically-derived numerical concentration or narrative statement considered to be protective of the water values and uses in ambient conditions for a specific waterbody.

1. GENERAL OVERVIEW

1.1 Introduction

The British Columbia (B.C.) Ministry of Environment and Climate Change Strategy (ENV) develops province-wide ambient water quality guidelines (WQG) for substances or physical attributes that are important in both the fresh and marine surface waters of B.C. The ENV defines a WQG as a scientifically-derived numerical concentration or narrative statement considered to be protective of designated values in ambient conditions. WQGs apply generically province-wide and are used to:

- protect aquatic life, wildlife, and their habitats;
- protect water uses, including: drinking water, public supply, and food processing; agriculture (livestock watering and irrigation); and recreation;
- provide the basis for the evaluation of ambient water quality and environmental impact assessments to inform resource management decisions;
- provide the basis for the establishment of water quality objectives (WQO);
- provide the basis for wastewater discharge authorization limits; and
- report to the public on the state of water quality and promote water stewardship.

The WQGs are set after considering the scientific literature, results from toxicity tests, WQGs from other jurisdictions, and background conditions in B.C. The scientific literature gives information on toxic effects but this information is typically based exclusively on laboratory tests conducted on partial life cycles of a limited number of species. Applying this information directly to field conditions, where organisms experience complex ecosystem interactions as well as a myriad of anthropogenic stressors, comes with an unknown level of uncertainty. ENV WQGs incorporate assessment factors to account for the uncertainties of extrapolating laboratory derived toxicity values to field conditions.

Once approved, WQGs constitute official ENV policy and must be considered in any decision affecting water quality made within the ENV; they do not, however, have any direct legal standing. WQGs provide information and policy direction to those making decisions affecting water quality, provided they do not allow legislated effluent standards to be exceeded. WQGs can be used as the basis for authorized waste discharge limits. These limits are set out in waste management permits, approvals, plans, or operating certificates, which are legally enforceable.

This document provides the derivation protocol for WQGs for the protection of aquatic life. Agriculture and wildlife WQGs are derived following the guidance provided by the Canadian Council of Ministers of the Environment (CCME) in the following documents:

- *Protocols for Deriving Water Quality Guidelines for the Protection of Agricultural Water Uses (Irrigation and Livestock Water)* (CCME, 1999b).
- *Protocol for the Derivation of Canadian Tissue Residue Guidelines for the Protection of Wildlife that Consume Aquatic Biota* (CCME, 1999c); and

When a recreational or drinking water WQG is not available from Health Canada, or circumstances dictate a B.C.-specific WQG is required, ENV works collaboratively with the B.C. Ministry of Health to develop or adopt an appropriate WQG, after considering the following documents:

- *Guidelines for Canadian Recreational Water Quality* (Health Canada, 2012); and
- *Guidelines for Canadian Drinking Water Quality* (Health Canada, 2017).

B.C. has developed WQGs for many substances of concern and continues to develop and update WQGs for priority substances. For substances without approved WQGs, ENV adopts WQGs from other jurisdictions as "working" WQGs to provide guidance for water quality assessments and resource management decisions on an interim basis. The working WQGs are largely based on the CCME WQGs, but WQGs from other jurisdictions and information from the published scientific literature are used as well.

1.2 Protocol Update Overview

With this document, B.C. is adopting a statistically based approach, the species sensitivity distribution (SSD). The SSD approach is used by the CCME, the National Standards and Guidelines Office within Environment Canada, as well as multiple international jurisdictions including the European Union, Australia and New Zealand. WQG development is a resource intense process; by adopting the SSD approach, B.C. will enhance its opportunities to collaborate and share guidelines with other jurisdictions.

B.C. is a member of the CCME and participated in the development of the CCME aquatic life WQG derivation protocol (CCME 2007) but maintains an independent B.C.-specific protocol which is consistent with B.C. policy objectives. This is necessary given B.C.'s recognition of the uncertainty associated with WQG derivation and differences in objectives for short-term WQGs compared with the CCME[1]. Given these differences in policy, it was not possible to directly adopt the 2007 CCME protocol, however to encourage collaboration on a national basis the B.C. protocol is harmonized with the CCME (2007) protocol wherever possible.

This document includes procedures for developing three types of guidelines: Type A1, Type A2 and Type B. Type A guidelines are SSD-based and classified based on the quality and quantity of the available data used to derive the WQG. Type A1 is a novel WQG unique to B.C. It has the highest data standards and is designed to minimize uncertainty. Type A2 WQGs have an increased breadth of acceptable data and are congruent with the CCME Type A WQGs (CCME, 2007). Type B WQGs are based on the extrapolation from the lowest available and acceptable toxicity endpoint and are congruent with the CCME Type B2 WQG. Each approach has defined minimum toxicological data set requirements, which are discussed further in Sections 4.5 and 5. The preference of WQG types is as follows: A1>A2>B. Where possible, WQGs are developed for both short-term and long-term exposures.

This protocol update relies heavily on the CCME aquatic life WQG derivation protocol (CCME, 2007) and sections of the CCME protocol are replicated here verbatim. Large sections of replicated text are included as block quotations (indented 1 inch) and are highlighted light blue. Square brackets are used to indicate inserted text and page and section references are given in the foot notes.

1.3 Guiding Principles

The following fundamental principles are used in developing WQGs in B.C.:

- B.C. WQGs are generic provincial recommendations that are based on the most current scientific information available at the time of their derivation. While they may consider some

[1] B.C.'s acute WQGs are for the protection of all individuals against severe effects such as lethality and CCME's acute WQGs are for protection of a specified fraction of individuals. See discussions in Sections 1.3 and 5.1.6.

toxicity-modifying factors (TMF), they do not directly consider technological, socio-economic, or management factors that may influence their implementation.

- Both long-term chronic and short-term acute WQGs are derived if sufficient toxicological data are available.
    - Long-term chronic WQGs are intended to be protective of all forms of aquatic life (all species, all life stages including multi-generational) from lethal and negative sub-lethal effects over indefinite exposures.
    - Short-term WQGs are intended to protect aquatic organisms against severe effects such as lethality due to short-term intermittent and/or transient exposures to contaminants (e.g. spill events or infrequent releases of short-lived/non-persistent substances).
- The approach to develop WQGs reflects the philosophy that protection of aquatic life is characterized by protection of individuals, which in turn also protects populations[2]. It is noted, however, that this approach may not protect individuals weakened to some degree through age, illness, or injury.
- Inclusion of all higher components of aquatic ecosystems (e.g. algae, macrophytes, invertebrates, amphibians and fish) in WQG derivation is preferred and the absence of one or more components increases the overall uncertainty of the WQG.
- Only a long-term WQG may be derived for substances which may not be acutely toxic but pose a risk to aquatic life due to their low water solubility (e.g. PCBs and dioxins), ability to bioaccumulate, or mode of action (e.g. endocrine disrupters).
- Toxicity modifying factors (TMFs) affect the bioavailability of some substances (e.g. metals) through different mechanisms (e.g. chemical speciation). The effects of TMFs will be incorporated fully in the derivation of WQGs, provided that the scientific information to do so is available.
- B.C. WQGs provide the basis for the derivation of Water Quality Objectives (WQO), which take local circumstances (e.g. waterbody) into account.
- Scientific uncertainty and data limitations in the derivation of WQGs will be documented in a transparent manner and be counterbalanced using an assessment factor.
- The derivation of B.C. WQGs will be done in a clear and transparent manner. Whenever possible it will follow the process outlined in this document. However, if scientifically warranted, it is acceptable to deviate from this process (e.g. a WQG for general parameters, such as pH or temperature, may require a different approach than for chemical substances).

Generally, guidelines are set separately for freshwater and marine environments because of the fundamental differences in the chemistry of these two types of water bodies, which often result in different toxic effects elicited by a substance. However, for substances for which no significant influence on chemical behavior can be shown or reasonably anticipated, and where no differences in toxicity toward freshwater and marine organisms

[2] Toxicological tests are conducted at the organismal level and the extrapolation of these results to ecological populations is problematic given that ecological populations are subject to multiple factors not considered in laboratory tests (Johnston and Sumpter 2016).

(by comparison of similar taxonomic groups) can be seen, toxicity data from freshwater organisms may be used to broaden the marine database.[3]

1.4 Selection of Substances for Guideline Development

The selection of substances for the development of new WQGs or the update of existing WQGs is done by the Water Protection and Sustainability Branch (Environmental Sustainability and Strategic Policy Division) in consultation with the Regional Operations Branch of ENV's Environmental Protection Division. Input from external sources (e.g. public, academia, industry, other levels of government) may also be considered. Substances are prioritized annually according to operational needs and available resources. Deriving or updating WQGs is not initiated until the appropriate resources have been identified and allocated to the project.

1.5 The Guideline Development Process

Generally, the development of a B.C. WQG consists of 5 distinct steps (Figure 1). In Step 1, the background information is compiled and the general aspects of the substance in question are described. These include the physical and chemical properties, environmental fate, ambient concentrations in B.C., and WQGs from other jurisdictions.

In Step 2, the toxicity literature is reviewed to summarize the toxicity of substances to aquatic life, the mode of toxicity, as well as to identify and quantify TMFs (e.g. pH, temperature, hardness [Ca, Mg, and carbonate ion], organic matter, oxygen, and counter-ions [i.e. the anion or cation linked to the ion of interest]).

In Step 3, the available toxicity data are compiled, evaluated, and classified based on criteria for data quality. This step includes standardizing (i.e. normalizing) data to account for any TMFs quantified in the previous step. This standardization allows a more accurate comparison and evaluation of the available toxicity data and results in more appropriate WQGs.

WQGs are derived in Step 4. Once the data have been collated, classified and normalized, it is possible to determine what type of WQG can be developed, i.e. whether adequate data are available for short- and/or long-term WQGs and for marine and freshwater environments. This step includes the analysis and/or graphical representation of the toxicity data and the application of an assessment factor. The draft WQG is further evaluated for adequate protection of species and compared with ambient concentrations.

Step 5 includes the completion of the draft technical report that contains all the information collected and the rationale for decisions. This draft report is then circulated for internal and external reviews (see Section 5.5 for further discussion of the review process). Once comments from these reviews have been incorporated, the WQG is presented to ENV Executive for final approval and then posted on the ENV website.

[3] From CCME (2007) Part I, page 2.

Overview of B.C. Water Quality Derivation Process

Step 1 **Compilation of Background Information**

- Physical and chemical properties
- Environmental fate and transportation
- Bioaccumulation and bioconcentration
- Analytical methods for substance quantification in environmental samples
- Ambient and background concentrations
- Water quality guidelines from other jurisdictions

Step 2 **Review of Toxicological Literature**

- Toxicity of substance to aquatic life
- Exposure and route of uptake
- Essentiality
- Mode of toxic action
- Identification and quantification of TMFs

Step 3 **Compilation of Toxicity Data**

- Evaluation of toxicological studies
- Data quality classification
- Check for minimum data requirements
- Standardize data to account for TMF

Step 4 **Water Quality Guideline Derivation**

- Decision on WQG type given data quality and quantity (preference: A1>A2>B)
- Data analysis and graphical representation
- Estimation of HC_5 (Type A) or critical data point (Type B)
- Application of assessment factor
- Evaluation against protection clause
- Comparison with ambient concentrations

Step 5 **Water Quality Guideline Review and Approval**

- Completion of draft technical report
- Internal government expert and technical review
- External expert and public review
- Technical report finalized
- Executive approval
- Posting on ENV website

Figure 1.1. Steps for developing British Columbia water quality guidelines for the protection of aquatic life.

1.6 Definitions of Freshwater and Marine Systems

Guidelines are set separately for freshwater and marine systems. Freshwater is defined as water with a total dissolved salt content equal to or less than 1,000 ppm (1 g/L, 1 ‰ [parts per thousand]).

Marine water is defined as water with a total dissolved salt concentration greater than 5,000 ppm (5 g/L, 5 ‰). Marine water (open ocean) generally has a dissolved salt concentration of approximately 34 – 35 g/L (34-35 ‰), but near shore marine water can have considerably lower concentrations (often approximately 28 g/L).

When total dissolved salt concentrations are 1 – 5 g/L (e.g. in some brackish waters), the WQG protecting the most sensitive condition, be it for freshwater or marine, should be applied, unless sufficient data are available on resident aquatic species and environmental conditions to justify a different choice.

The same definitions also apply in the categorization of toxicity data as applicable for the derivation of the freshwater and marine guidelines. Toxicity tests conducted in low-salinity brackish water (e.g. when the total dissolved salt concentrations are 1 – 5 ‰) are categorized according to best scientific judgement.

In this protocol, marine species include those species found in estuarine, coastal, and open ocean habitats, any of which may be used to derive a guideline.[4]

1.7 Uncertainty in Developing Water Quality Guidelines

Water quality guidelines are derived from acute or chronic laboratory-based toxicity tests of single contaminants on a limited number of aquatic species, which may or may not be representative of sensitive species found in B.C. The extrapolation of these data to aquatic ecosystems exposed indefinitely to mixtures of substances comes with several sources of uncertainty, including: inter and intra-species variation; laboratory to field extrapolation; and statistical modelling.

A major source of uncertainty is the differences in sensitivity to contaminants between species. Toxicological testing is only done on a small fraction of the freshwater species which may or may not adequately describe the sensitivity of all species in an ecosystem. For example, B.C. has 67[5] species of native freshwater fish, but WQGs are derived using data from as few as three fish species). Furthermore, some studies have demonstrated ecologically-relevant aquatic species to be significantly more sensitive to several contaminants than routinely tested lab species (e.g. Besser et al. 2005; Dwyer et al. 2005; Wang et al. 2014; Besser et al. 2016). Within species, laboratory strains may have different sensitivities compared to field populations creating further uncertainty regarding the effect of contaminants across the range of genetic variability within a species (Sun et al. 2015; Sun et al. 2018).

Toxicological testing typically involves exposing individual organisms to a single contaminant for short time periods under laboratory conditions. These results are then used to determine potential risks to species populations and ecosystems. There are a number of uncertainties associated with extrapolating laboratory studies to field situations. These include:

[4] From CCME (2007) Part I, page 6.

[5] http://ibis.geog.ubc.ca/biodiversity/efauna/freshwaterfishes.html

- whole life-cycle exposures vs. partial life-cycle exposures, which are typical of many toxicity tests;
- delayed effects such as delayed mortality, impaired reproduction, and multigenerational effects;
- shifts in exposure from a single contaminant to multiple contaminants (additive, synergistic, antagonistic effects);
- the toxicity of associated metabolites;
- indirect effects (e.g. food web dynamics); and
- the cumulative effects of other environmental stressors (e.g. habitat loss and impacts of climate change) and exposure to the contaminant in question.

Assessment factors (AF) are used to account for the uncertainty associated with extrapolating from laboratory based toxicological data to different species and field conditions. The AF provides a mechanism to account for the uncertainty that is not considered in the design of toxicological experiments (European Commission 2011). Its use allows the transparent identification and consideration of uncertainty to derive a WQG that is a best estimate of low risk conditions; meeting the policy requirement of a WQG that can act as a protective environmental management benchmark.

The value of the AF is determined in the derivation of each WQG after considering the quality and quantity of available data, the representation of the taxonomic groups covered, knowledge of the presumed mode of action, toxic effects of the substance, statistical uncertainties, and comparisons with field and mesocosm studies (European Commission 2011). AFs generally range from 2 to 10 but may be as high as 100 in some cases. The choice of AFs must be clearly justified in the technical report for the WQG. An AF greater than 10 would imply high uncertainty with the WQG. In these cases, the WQG will be approved on an interim basis with an adjoining list of research priorities that once resolved would reduce the uncertainty of the WQG and also reduce the associated AF.

1.8 Use of Water Quality Guidelines in Resource Management

In B.C., WQGs provide basic scientific information about the effects of water quality parameters on aquatic life and have three basic applications: the assessment of ambient conditions (e.g. state of environment reporting); environmental impact assessments to support management decisions (e.g. waste management authorizations); and the development of WQOs. The assessment of ambient conditions considers water quality outside the initial dilution zone of a waste discharge and, depending on the location of sampling sites, can be used to assess the cumulative effects of several point and non-point source discharges. Environmental impact assessments involve the evaluation of predicted water quality conditions resulting from a proposed land development. WQGs are often used to determine if the predicted water quality is acceptable with respect to the identified values for a specific waterbody. To do this, the desired level of protection must be clearly articulated. This information can be used to identify water quality issues and, where needed, establish water quality benchmarks, such as WQOs or science-based environmental benchmarks (SBEB) (see ENV 2016).

The term “assimilative capacity” refers to the ability of a natural waterbody to receive wastewaters or toxic substances without deleterious effects and without damage to aquatic life or humans who consume the water. The simplest way to estimate assimilative capacity is to calculate the difference between the ambient level of a substance and the corresponding WQG level. Once this is known, it is important to specify the amount of assimilative capacity to be allocated for waste management purposes, based on the desired level of protection for the waterbody in question. Sensitive aquatic habitats or waterbodies of significant value may dictate that there should be no substantial change from background, allowing minimal allocation of the assimilative capacity to protect existing conditions or set remediation goals where conditions have been degraded. In other situations, some allocation of the

assimilative capacity may be acceptable to facilitate resource development while protecting the identified values. Finally, full allocation of the assimilative capacity may be acceptable in situations where existing operations or legacy impacts have changed the water quality from background conditions but are still protective of designated uses. Whichever level of protection is identified, the WQGs provide the starting point for water quality assessments to interpret ambient conditions or the acceptable level of risk associated with a development proposal. Allocation of the total assimilative capacity should be avoided to provide a margin of safety against fluctuations in waste discharges and unknown effects.

1.9 Limitations of Water Quality Guidelines in Resource Management

WQGs are the best estimate of low risk conditions that can be derived given the available scientific information. However, it is important to note that a WQG is an estimate that has not been validated under field conditions. There are major uncertainties associated with deriving a WQG (see section 1.7) and although there is an attempt to account for these in the derivation method, a WQG is only an estimate of low risk conditions. Ongoing ecological monitoring is necessary to ensure the WQG is affording the expected level of protection (Johnson and Sumpter 2016). Programs such as the Canadian Biotic Monitoring Network (CABIN) offer long term standardized data sets that can be used to test the efficacy of WQGs and other benchmarks (e.g. WQOs, SBEBs) for protecting aquatic life.

B.C. WQGs are provincial in scope, which means they are based on generic, rather than site-specific, environmental fate and behaviour data. In most instances, B.C. WQGs are sufficient to assess water quality issues in the environment. However, as sites and ecosystems vary in aquatic species composition and environmental characteristics (such as pH, water hardness, temperature regimes, chemical composition, etc.), the actual site-specific toxicity and environmental impact exhibited by the parameter of concern varies as well. This can result in a situation where the B.C. WQG is potentially under- or even over-protective at a site. It is important to understand that B.C. WQGs are not enforceable standards, but rather provide information to inform water resource management decisions. They can be applied directly in those decisions or modified as required with sufficient supporting information.

> The concentration of a substance in the ambient environment is influenced by natural factors, human actions, or a combination of both. Ambient concentrations are variable in space and time. For substances that occur naturally, it can become important to distinguish between the concentration that is due to natural causes (i.e. the natural background concentration) and the concentration that is due, at least in part, to anthropogenic activities. The natural background concentration of naturally occurring substances is a very site-specific matter and cannot be incorporated into a [provincially] applicable guideline value. [A provincial WQG] is derived considering all acceptable and applicable toxicological data from a variety of toxicological studies. These location-independent toxicological studies will have been performed with different species, with different histories and under different exposure conditions, so it is possible that the recommended [provincial WQG] will fall below (or outside) the natural background concentration (or natural condition) of a particular site of interest. While this fact does not invalidate the derivation process of a [provincial WQG], it points out the need for the

user to understand the derivation process and to know how to properly apply [provincial WQG values].[6]

2. COMPILATION OF BACKGROUND INFORMATION

The first step in deriving a WQG is to gather background information to better understand the substance and its chemical, physical and biological properties. A review of the literature should provide the information described in the following sections to support the derivation of the WQG. This information is then provided in the technical document to ensure users of the WQG have all the necessary information to inform decisions. Data gaps must be clearly identified and described.

Comprehensive information for each substance selected is desirable, but not necessary for the development of a guideline. [A literature search is conducted to obtain available information on the following]:

- physical and chemical [properties];
- [anthropogenic production, releases, and uses];
- ambient environmental concentrations (and where applicable and possible, information on whether elevated levels are due to natural or anthropogenic causes);
- environmental fate processes, persistence, and behavior of the substance in water, soil, sediment, air, and aquatic biota;
- routes of exposure and uptake by aquatic organisms;
- mode of toxic action and related toxicokinetics;
- toxicity to aquatic biota after short- and long-term exposures;
- bioavailability, including the conditions under which the [substance] is bioavailable;
- bioaccumulation potential;
- toxic interactions, behavior of mixtures, and interactions with other variables (i.e. parameters affecting exposure and toxicity);
- essentiality (if applicable);
- analytical and toxicological testing methods (including current detection limits);
- breakdown of products and by-products; and
- additional information (e.g. guidelines, objectives, criteria, standards, etc. from other jurisdictions).[7]

[6] From CCME (2007) Part I, page 2-3.
[7] CCME (2007), Part II, Section 1, page 1.

2.1 Environmental Behaviour, Fate, and Persistence

It is not required to have complete information on each of the following points. However, the intent is to be able to identify the major environmental pathways and fate of a substance in the aquatic environment. Specifically, the following should be determined:

- solubility of the substance in the various aquatic environments (freshwater and marine, hard versus soft water, pH and temperature influence, etc.);
- mobility of the substance and the compartments of the aquatic environment in which it is most likely to be present;
- kinds of chemical and biological reactions that take place during transport and after deposition;
- eventual chemical form under various environmental conditions;
- persistence of the substance in water, sediment and biota; and
- toxic interactions with other substance (i.e. parameters affecting exposure and toxicity).[8]

It is important to understand the basic physical and chemical behaviour of the substance in the aquatic environment, such as water solubility and precipitation, chemical speciation, and chemical reactivity. The mobility of the substance and the compartments (i.e. water, sediment, biota, soil, and air) in which the substance is most likely to be present should be identified. Potential fate processes include volatilization, hydrolysis, oxidation, photolysis, speciation, aerobic and anaerobic biotransformation (e.g. methylation/demethylation), long-range transport, soil and sediment sorption/desorption, bioaccumulation, and, for a few organic substances, biomagnification. The chemical speciation and the factors influencing changes in speciation are especially important for metals. These variables should be described in detail.

When possible and applicable, the residence time of the substance should be expressed in terms of its residence half-life in water, sediment, biota, soil, and air, while considering potential degradation and speciation. Understanding the actual residence time (i.e. persistence) can be especially important for the potentially bioavailable metal fractions in water, sediment, and biota.[9]

2.1.1 *Speciation of the Substance*

Speciation is the determination of various physical and chemical forms of a substance. Physical speciation represents the different states of a substance in the environment (hydrated [i.e. filtered/dissolved], labile particulate, refractory particulate, organically complexed, labile dissolved, colloidal, or total). Chemical speciation refers to the identity of the chemical species in solution (e.g. Cr^{3+}, $Cr_2O_7^{2-}$, or CrO_4^{2-}). Speciation is an important concept in the aquatic environment because of the continual interactions between

[8] CCME (2007), Part II, Section 1, page 1.
[9] CCME (2007), Part II, Section 1, page 2.

substances and various biogeochemical factors (such as DOM, pH, temperature, ligands, etc.) that modify the chemical species present in solution and, therefore, can have an impact on the bioavailability and toxicity of the substance in the aquatic media. The solubility and the persistence of the fraction should be assessed in order to predict the deleterious effects encountered over the short- and long-terms in the aquatic environment.[10]

2.2 Bioaccumulation

Mechanisms of bioaccumulation of naturally occurring inorganic substances (i.e. accumulation via water and food) are different than for organic substances. While the bioaccumulation of organic substances (including organo-metals) depends mainly on hydrophobicity, molecular size, lipid content of the exposed organism (allowing the use of predictive models that employ the octanol-water partition coefficient [K_{ow}] approach to estimate bioaccumulation within an individual), and persistence, the bioaccumulation of naturally occurring inorganic substances in aquatic systems depends largely on speciation, on the properties of the surrounding medium, and the specific physiological mechanisms of uptake organisms.

While the notion of bioaccumulation is important in aquatic toxicology, it is not considered to be part of the protocol for the derivation of [B.C. WQGs], as this protocol deals with the concentration of the substance in the water column and the toxic effects resulting from direct exposure. However, the ability and likelihood of a substance to bioaccumulate should be discussed in the guideline document, with routes of exposure limited to water. It must be noted that bioaccumulation does not necessarily result in toxicity; this depends on bioreactivity within organisms. Because the bioaccumulation potential of a substance depends on many factors and is situation-specific, no defining criteria are provided to categorize a substance as bioaccumulative. If necessary, the substance will be assessed on a case-by-case basis. Similarly, while biomagnification is important in many anthropogenically created organic compounds, it does not seem to be prevalent in most naturally occurring substances (exceptions are methylmercury, organo-selenium compounds, and, potentially, some other organo-metals). These issues of bioaccumulation and biomagnification are not addressed formally in the derivation sections, but should be considered in a case-by-case approach during the WQG derivation of substances, if appropriate and/or required. Bioaccumulation and biomagnification are more appropriately considered in the derivation of other types of guidelines (e.g. tissue residue guidelines).[11]

2.3 Analytical Methods for Substance Quantification in Environmental Samples

A description of the analytical methods for substance quantification in environmental samples must be included in the technical report. This provides information to potential guidelines users as well as

[10] CCME 2007, Part II, Section 2, Page 3
[11] CCME 2007, Part II, Section 1, Page 4

provides context when evaluating draft WQGs. The B.C. Environmental Laboratory Manual (ENV 2015) is used as the primary source for this information. If necessary, additional sources may be considered.

In particular, it is necessary to consider the method detection limit (MDL) of the recommended analytical method. The MDL is defined as "the minimum concentration of a substance that can be measured and reported with 99% confidence that the analyte concentration is greater than zero and is determined from analysis of a sample in a given matrix containing the analyte" (CFR 2011). 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" (USEPA 1985d), is in practice about 5-10 times higher than the MDL (USEPA 1985d). Therefore, it is recommended that the analytical methods to measure ambient concentrations should have MDLs, 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.

2.4 Ambient and Background Concentrations

Where possible, information on ambient and background concentrations in the B.C. environment should be provided, including areas where concentrations are naturally or anthropogenically elevated. Water quality data are available from the ENV Environmental Monitoring System (EMS), the Canadian Aquatic Biomonitoring Network (CABIN), and Environment and Climate Change Canada's National Long-term Water Quality Monitoring Database. Providing such information helps in assessing the relevance of the final WQG value and its application in B.C.

The natural background concentration of a substance varies according to the geological setting and the natural processes occurring in the surrounding environment. It is important to consider the variability associated with a substance across the province, as certain areas can have naturally elevated concentrations. These concentrations may be higher than what the sensitive aquatic organisms are able to tolerate. Sensitive species or sensitive members of a species that cannot acclimate or adapt will not exist in such areas and testing with naïve (i.e. non-tolerant) laboratory species will not be environmentally relevant for such situations. This issue must be considered at the time of WQG derivation and its application at each location.

Data should be summarized on a regional basis and separately for lentic and lotic environments. Both total and dissolved fractions should be characterized, where appropriate. It is important to recognize the influence of MDLs when characterizing ambient concentrations. Outdated MDLs can be much higher than current MDLs and results reported as less than the MDL may lead to misinterpretation of the data. The data set must be examined for outliers due to data entry or analytical errors. Clear documentation must be provided to describe how outliers are treated.

2.5 Water Quality Guidelines from Other Jurisdictions

It is helpful to include WQGs from other jurisdictions in the technical report and to address any discrepancies with the new B.C. WQG. In most cases, WQGs from other jurisdictions are dated and do not consider new scientific evidence. It may also be possible that different approaches are used by other jurisdictions in accordance with the jurisdiction's internal policy and guiding principles. These need to be identified in the technical report to avoid potential confusion over the discrepancies.

3. REVIEW OF TOXICOLOGICAL LITERATURE

3.1 Toxicity of a Substance to Aquatic Life

The toxicological information must be relevant for the derivation of WQGs, i.e. it must relate to a negative effect on an aquatic organism or population and it must be ecologically significant. In cases where the organisms are semi-aquatic or have a partial aquatic life stage, the negative effect must result from exposure in the aquatic environment. For invertebrates, acceptable data are for fully aquatic life forms such as Crustacea (plankton, benthic), insect larvae (e.g. Ephemeroptera, Plecoptera, Trichoptera, Chironimidae), and Mollusca, or for the aquatic life stage of semi-aquatic insects (e.g. beetles [Coleoptera] and some Hemiptera) that can leave the aquatic environment. Data on the non-aquatic life stages of these insects are not considered.

As some elements (e.g. copper and zinc) are essential for the physiological and metabolic processes of organisms, care must be taken in the analysis and evaluation of toxicological studies of these elements. Observed negative effects associated with such a substance may be due to over-exposure, as well as under-exposure (i.e. deficiency of an essential element). It is therefore important to understand the range of concentrations of a substance that is harmful, as well as essential to an organism.

Information on toxic interactions and the behaviour of a substance in mixtures is important, but unfortunately often still too incomplete to be incorporated into WQG development. However, where possible (i.e. [where sufficient data and appropriate methods are] available), information on toxic interactions and the behaviour of a substance in mixtures will be incorporated.[12]

3.1.1 *Use of Traditional and Non-traditional Endpoints*

Effects endpoints used in the derivation of WQGs include the traditional endpoints (i.e. growth, reproduction, and survival), as well as non-traditional endpoints (e.g. behaviour [predator avoidance, swimming ability, swimming speed] and physiological changes), but only if the ecological relevance of these non-traditional endpoints can be demonstrated. Non-traditional endpoints are evaluated on a case-by-case basis, using as criteria whether the measured impact has the potential to have a strong negative influence on ecological competitiveness at a population level, as well as the overall reliability and reproducibility of the laboratory test.

Ecological relevance pertains to whether physical abilities (e.g. swimming speed, orientation ability, and migratory fitness), physical traits (e.g. fin size/shape), physiological abilities (e.g. egg laying), physiological traits (e.g. production of a certain enzyme), and/or behavioural tendencies (e.g. swimming in groups) of organisms are important enough to influence a species' ecological competitiveness. Characteristics that are of high ecological relevance are those that have a strong positive or negative influence on survival, reproductive ability, and growth (e.g. stunting, high fertility, and organ failure). Non-

[12] CCME 2007, Part II, section 1, page 2-3

> traditional toxicological endpoints are often difficult to link to ecological relevance because the adverse effects they test do not have a primary impact on survival, reproductive ability, or growth.[13]

3.2 Exposure and Route of Uptake

> Aquatic organisms are exposed to substances via uptake directly from the water and diet. For many substances, water exposure is likely the dominant uptake route. However, for some substance, exposure from bedded or suspended solids, as well as other dietary sources, may be equally or more important. For example, the organo-forms of selenium and mercury, as well as chlorinated pesticides such as DDT, are accumulated primarily via the diet, resulting in toxic responses. For these substances, the sediment quality guidelines and the tissue residue guidelines are important.[14]

When deriving WQGs for a substance with the potential to bioaccumulate, caution must be exercised to clearly identify the route of exposure (i.e. water, diet, or both). For substances in which the dietary intake does not significantly contribute to uptake and toxicity, the derivation of WQGs should focus primarily on studies with a water exposure route.

3.3 Essentiality

It is important to consider the essentiality of naturally occurring substances (i.e. elements) during WQG derivation to ensure the WQG is not below a concentration essential to aquatic life.

> The essentiality of an element means that the absence or deficiency of the element results in the impairment of life functions, and that the impairment can be prevented or corrected only by supplementation of physiological levels of this element and not by others (Chowdhury and Chandra 1987). Therefore, essential elements differ from nonessential elements and other non-nutritive chemicals, as negative effects on organisms are observed when insufficient levels (i.e. levels below the compensation limit of the accumulation/assimilation of the organism) of the essential element are present in the environment. This deficiency varies between organisms, between aquatic species, and within aquatic species based on their respective locale (adaptation). As organisms have adapted to their natural habitat, it can be assumed that the natural background concentrations of essential elements at a given locale fulfill the requirement of essentiality to organisms there. Organisms requiring levels of essential elements in greater quantities than those naturally present in an environment are not expected to be present in this environment to begin with or, if present, would suffer from deficiency not caused by anthropogenic influences.
>
> In order to prevent anthropogenically created adverse health effects to organisms caused by a deficiency of essential elements, recommended threshold levels for these elements should not fall below the level required by the organisms at a particular site needing the highest concentration to remain healthy (i.e. the organism with the highest deficiency

[13] CCME 2007, Part II, section 1, page 2-3
[14] CCME 2007, Part II, section 1, page 2

threshold). This necessitates the caveat that if a toxicity-derived guideline value is below the natural background concentration at a certain locale, this background number would be taken as the guideline value. This will prevent recommending a guideline value that could lead to potential deficiency effects.

Consequently, guideline derivation procedures for essential and nonessential elements are the same.[15]

3.4 Mode of Toxic Action

Toxicity can occur because of direct physical damage to an external biological surface (e.g. eyes, scales, and respiratory surface) or alterations to physiological processes within an organism. Guidelines are derived for waterborne concentrations, so an understanding of the relevance of waterborne concentrations to the overall mode of toxic action and resulting toxicity (especially toxicokinetic aspects) is necessary.[16]

Some substances (e.g. metals) exhibit a complex environmental chemistry and toxicology, and therefore create unique challenges in their WQG derivation and environmental management.[17]

Substances are not toxic unless they are available to organisms at a sufficient dose in a bioavailable form. Bioavailability is defined as the portion of a substance that is immediately available for uptake by organisms. Bioavailability of different substances can change over time. Bioaccessibility refers to the fraction of a substance, such as a chemical present in the environment that may be available for uptake by organisms over the long term. This fraction includes the portion of the chemical that is currently bioavailable as well as the portion that may become bioavailable over time (e.g. as and if conditions change). Actual uptake of a substance by an organism is termed bioabsorption. Bioreactivity refers to the portion of a chemical within an organism that causes toxicity; it comprises the bioabsorbed fraction minus the fraction that is depurated, internally sequestered, or used by the organism for its own needs.[18]

3.5 Identification and Quantification of Toxicity Modifying Factors

To provide the best guidance, WQGs which factor in bioavailability (i.e. the fractions toxic to aquatic organisms), based on the relevant physical and chemical speciation, will be developed where possible. These WQGs should focus on the bioavailable and potentially toxic forms of substances related to:

- the form of the substance as it enters the environment, as well as the forms it acquires while circulating through the environment;
- the abiotic environmental conditions affecting the substance (i.e. water and sediment chemistry, climatological conditions, etc.); and

[15] CCME 2007, Part II, Section 2, Page 4
[16] CCME 2007, Part II, section 1, page 3
[17] CCME 2007, Part I, page 3
[18] CCME 2007, Part I, page 3

- the biotic environment (i.e. selective uptake and excretion by organisms, aquatic species sensitivity, exposure routes, etc.).

A variety of environmental factors and physical, chemical, and biological interactions modify the exposure and behaviour of chemical substances and thus toxicity to aquatic plants and animals. Toxicity-modifying factors (TMFs) may be grouped as follows:

- substance-ion interactions (e.g. hardness/alkalinity, pH, salinity, and other anions or cations);
- substance-organic matter interactions (e.g. humic substances, organic carbon, and turbidity impact);
- substance-substance interactions; and
- temperature and other physical influences (e.g. light intensity, water turbulence, turbidity impacts, etc.).

Where possible, TMFs are important to identify, document, and account for in the derivation of WQGs. To expand the applicability of the WQGs, guidance on how these TMFs alter the toxicity and the WQG value must be given. The WQG derivation of substances such as cadmium, copper, and ammonia has, in the past, considered the hardness, pH, and/or temperature of the water to predict an impact to aquatic biota. Dissolved organic matter, alkalinity, and a variety of other factors have also been identified as important modifiers to aquatic toxicity.

The incorporation of TMFs results in a range of situation-specific WQGs. The user can then select the WQGs that are most appropriate to use for the site characteristics or situation in question.

Development of WQG equations and matrices are the most often used derivation route, as these can be derived independently from other parameters and, if necessary, with only a limited data set. This approach can be tailored to the specific needs and data availability of the substance and can range from a simple, single parameter equation to complex, multi-factor equations and matrices. These in turn can then be combined to derive models. While providing provincial guidance on the substance, the incorporation of the functional relationship between toxicity and TMFs readily allows the application of the WQG on a site-specific level.[19]

3.5.1 *Approach for Evaluating TMFs*

The first step in evaluating TMFs consists primarily of a detailed search of the toxicology literature for the substance in question with the goal of identifying studies that examined the toxicity-modifying factors, grouping of these studies, and analyzing them thoroughly.

After evaluation of all factors and the information available on them, the most pertinent ones (i.e. the factors that influence the expressed toxicity of the substance in question the most) are identified.

[19] CCME 2007, Part II, Section 2, Page 1

The next step entails quantifying, where possible, the influence of the most pertinent TMFs identified. This can be done through either the use of simple equations and/or matrices or the use of complex equations or models (e.g. Biotic Ligand Model), where appropriate. The extent and magnitude of influence that the selected parameters will have on the final WQG values depends on the amount and depth of data available and the level of understanding of the interaction between these factors and the substance.

The Biotic Ligand Model (BLM) (Paquin et al. 2002) quantitatively evaluates the way several water chemistry parameters affect the speciation and bioavailability of metals in aquatic systems. To date, it is validated for a small but growing number of metals. While initially developed only for freshwater systems and acute toxicity, it can now be applied to chronic toxicity and marine waters. The BLM can be used in the standardization of the data before a guideline is derived and in the expansion and application of the guideline to specific environmental conditions.

Once the impact of the most important TMFs is quantified, the toxicity data set can be standardized (normalized) as much as possible to the most appropriate conditions. This fully or partially standardized toxicity data set is used to derive the appropriate guidelines as described in [Section 5].

The reverse of the applied standardization method can be applied to the resulting guidelines to develop an equation and obtain guideline values that are more appropriate and pertinent to situations or sites with specific environmental characteristics. The issue of introducing bias in the guideline when back-transforming data is recognized and should be examined on a case-by-case basis.[20]

4. COMPILATION OF TOXICITY DATA

4.1 Evaluation of Toxicological Data

Each relevant toxicological study … is evaluated to ensure acceptable laboratory practices were used in the design and execution of the experiment. Each study is then classified as primary, secondary, or unacceptable, based on criteria [described in Section 4.2].[21]

While the evaluation of toxicological data should follow a basic format with certain requirements, scientific judgement is often required for the classification of studies. It is not mandatory for toxicity studies to follow standard design protocols; however, the data must be appropriate with respect to the substance in question. Nonstandard testing procedures can yield usable results and should be evaluated on a case-by-case basis for inclusion in the data set.

Since standard protocols for toxicity testing may become outdated or are not always available or followed, a great deal of variability exists in the quality of published data. To ensure a consistent scientific evaluation for each substance, the studies included in the data set should include the following information:

[20] CCME 2007, Part II, Section 2, Page 2
[21] CCME 2007, Part II, Section 1, Page 4

- test conditions/design (e.g. flow-through, renewal, static, single species study, community study, mesocosm, etc.);
- test concentrations;
- test containers;
- temperature, hardness (Ca^{2+} and Mg^{2+}), alkalinity, pH, dissolved oxygen, salinity, organic matter, adjuvants (chelators), and carrier solvents;
- solubility limits of substances in relation to tested concentrations;
- experimental design (i.e. analytical methodology, quality control/quality assurance, controls, and number of replicates); and
- description of statistics used to evaluate the data.

Where necessary, the influence of environmental factors on the expression of toxicity should be evaluated (i.e. TMFs – [see Section 3.5]).[22]

A variety of standardized toxicity test protocols have been developed for fish, invertebrates, and plants. The primary source of these is the Biological Test Methods and Guidance Documents published by Environment and Climate Change Canada (e.g. ECCC 1990a, 1998, 1999a). Several other useful sources of information for toxicity testing, analytical methods, and data interpretation are available (e.g. ECCC 1994, 2005; ASTM 2004; OECD 1993; USEPA 1985a, 1985b, 1985c, 1995, 2002a, 2002b) and should be consulted when necessary.

When consulting test protocols, evaluating toxicological information, and deriving WQGs, it is important to be aware of the following limitations:

- The study may have been performed prior to the development of an appropriate standardized test protocol. This does not necessarily invalidate the study.
- Standardized test protocols consider only a few well-studied aquatic species and biological processes.
- The ability to extrapolate toxicological results from one aquatic species to another (i.e. comparative ecotoxicology) is limited.
- There may be limited knowledge of the effects of metabolites and other environmentally transformed products of the parent chemicals.
- Protocols developed so far do not consider cumulative, synergistic, or antagonistic effects of chemicals or compensatory responses of organisms (such as acclimation, adaptation, or reduced density-dependent mortality among juveniles).
- The predictability of laboratory exposures and effects to aquatic ecosystems is still challenging.[23]

The analytical quantification of substances in the aquatic environment can often be very complex. Substances of importance for guideline development (e.g. metals, pesticides, and toxic by-products) are often found in trace amounts and few analytical apparati are set to attain such low detection limits. Also, only a few techniques are known today to

[22] CCME 2007, Part II, Section 1, Page 5
[23] CCME 2007, Part II, Section 1, Page 5

differentiate reliably, especially at environmental levels, between the different species of a substance, rendering the study of speciation in the aquatic environment difficult. Contamination, sampling procedure, sample preservation, storage, pre-concentration, and filtration may all be sources of errors rendering the task of achieving precision and accuracy complex. A thorough investigation of the data (technique and reliability) must be performed before considering the measured concentrations as acceptable values for a guideline derivation.[24]

Toxicity tests deviating from standard test protocols should be examined for their merit, and best scientific judgement should be used in deciding if the toxicity test is acceptable for use in WQG derivation.[25]

It is likely that specific standard toxicity tests (e.g. Environment Canada's Biological Test Methods and Guidance Documents) can be adapted for use with other closely related species. However, additional quality assurance data should be collected to assure the validity of the toxicity test when non-standard species are used.

4.1.1 Considering Bioavailability in Laboratory Studies

The bioavailability, or access that a substance has to the biological processes of an organism and, therefore, the toxicity of substances, can depend on chemical speciation (especially for, but not limited to, metals) and water chemistry (e.g. presence of organic matter, pH, etc.). Conditions under which the substance in question is bioavailable and how a change in conditions can or might change the bioaccessibility of the substance need to be investigated and are key factors to consider in guideline development. The interactions with other variables, i.e. the parameters affecting speciation and/or toxicity are important considerations.

Toxicological studies need to be conducted under conditions where the substance is bioavailable, otherwise toxicity will be underestimated. Studies conducted under conditions where the substance is not readily available (e.g. due to binding to organic or inorganic ligands) must be examined on a case-by-case basis. If the bioavailable and non-bioavailable fractions are not well characterized, identified, and quantified, these studies should not be considered acceptable for guideline development.

The bioavailability issue is particularly relevant to metals. The conditions under which the metal is bioavailable and bioreactive should be examined. Studies may report metal concentrations as total, filtered, dissolved, free, or bioavailable, and attention should be given to the analytical methodology used. The metal fraction (and species) used in the toxicity testing process should be clearly articulated if a study is used in guideline development.

From geochemical, biological, and analytical perspectives, the term "bioavailable fraction" is context-specific (i.e. not generalizable) and quantitatively elusive (Meyer 2002). Until it is possible to quantify in a scientifically defensible manner the bioavailable fraction of a

[24] CCME 2007, Part II, Section 2, Page 3
[25] CCME 2007, Part II, Section 1, Page 5

substance in the environment, [B.C. WQGs] will be derived based on chemical speciation-specific approaches. This may include the total and/or filtered fraction or chemical species.[26]

4.2 Toxicological Data Quality Classification

Toxicological data are classified as primary, secondary or unacceptable (see sections 4.2.1, 4.2.2, and 4.2.3, respectively) based on the suitability, usefulness, and reliability of the information. Appendix 1[27] provides an excel template for classifying data. Only primary data must be used to meet the minimum data set required for the derivation of Type A1 WQGs. For Type A2 and Type B, primary or secondary data may be used. Unacceptable data cannot be used in any derivation procedure.

The main exposure route(s) of an organism to a substance must be clearly stated in the WQG technical report to assist in the appropriate use of the WQG value. For this reason, in the evaluation of toxicity tests, it should be determined if organisms were fed during the study to allow the evaluation of the influence of food availability on the toxicity of the substance. Food availability can influence toxicity by:

- providing organic carbon to which substances can bind, thus reducing water column toxicity in the test chamber;
- by serving as an additional source of potentially toxic substances; and/or
- by keeping the organisms healthier than if they were not fed and thus better able to withstand toxic stress.

Similarly, the particulate matter or dissolved organic carbon (DOC) content of the test water should be noted and evaluated.

4.2.1 *Primary Data*

Primary data are those that are based on toxicity tests that are scientifically defensible. Toxicity tests must employ currently acceptable laboratory or field practices of exposure and environmental controls. Other types of tests using more novel approaches [(e.g. omics including genomics, proteomics, metabolomics)] will be evaluated on a case-by-case basis.

As a minimum requirement for primary data, substance concentrations must be measured at the beginning and end of the exposure period. Calculated substance concentrations or measurements taken in stock solutions are unacceptable in primary data. Test concentrations must be below the water solubility limit of the substance.

Measurements of abiotic variables such as temperature, pH, dissolved oxygen, water hardness (including Ca^{2+} and Mg^{2+} concentrations), salinity, dissolved organic matter (DOM), and the presence of other relevant substances should be reported so that any TMFs can be included in the evaluation process.

[26] CCME 2007, part II, section 1, page 3-4

[27] Available at: https://www2.gov.bc.ca/assets/gov/environment/air-land-water/water/waterquality/water-quality-guidelines/derivation-protocol/bc_wqg_aquatic_life_derivation_protocol_appendix1.xlsx

> For any toxicity test to generate primary data, appropriate replicates and dilution steps need to be completed. Generally, [non-renewed] static laboratory tests are not classified as primary data unless it can be shown that substance concentrations did not change during the test and that appropriate environmental conditions for the test species were maintained.[28]

Endpoints should be ecologically relevant toxic endpoints. These generally include, but are not limited to, reproduction, growth, development, and survival of young and adults. Other endpoints (e.g. behaviour, deformities, endocrine-disrupting effects, etc.) will be evaluated on a case-by-case basis. These can be included if it can be shown that those effects are a result of exposure to the substance in question, lead to an ecologically relevant negative impact, and are scientifically sound.

> In primary studies, the response and survival of controls (both positive [reference toxicants] and negative [uncontaminated conditions]) must be measured and reported, and should be appropriate for the life stage used. For standard test species (e.g. fathead minnow [*Pimephales promelas*], *Daphnia magna*, etc.), accepted control mortality rates should be considered for comparison to the treatment levels or concentrations. For nonstandard test species, the control mortality rate of the test should be used for comparison against the treatment levels of concentrations, provided the species has undergone previous reference toxicant testing to determine the acceptability of the species' response.
>
> A clear dose-response relationship should be demonstrated in the study. Studies with limited treatment levels may be considered if other toxicological studies support the effect level.
>
> Controlled microcosm and mesocosm studies are acceptable and are ranked according to the applicable categorization criteria. A clear dose-response relationship should be experimentally established and effects reasonably apportioned to the substance. As field studies generally have too many uncontrollable and recordable variables, they should not be used in WQG derivation. However, while not directly contributing to the actual WQG value derivation, field studies can play a significant role in evaluating and validating toxicological endpoints obtained in the laboratory and the final WQG.
>
> Statistical procedures used to analyze the data must be reported and be of an acceptable scientific standard.[29]

4.2.2 *Secondary Data*

> Secondary toxicity studies are those that cannot be classified as primary, but are still of acceptable quality and documentation. Secondary studies may employ a wider array of methodologies (e.g. measuring toxicity while tests species are exposed to additional stresses such as low temperatures, lack of food, or high salinity).

[28] CCME 2007, Part II, Section 1, Page 6
[29] CCME 2007, Part II, Section 1, Page 6

> All relevant environmental variables that modify toxicity must be measured and reported. The survival of controls must be measured and reported [and be appropriate].
>
> Static tests, calculated substance concentrations, and measurements taken in stock solutions are generally acceptable. Test concentrations must be below the water solubility limit of the substance. Evaluation criteria include the nature of the substance (e.g. volatility, complexation/chelating potential [especially for metals], stock solution analysis, nominal stock solution, and dilution series).
>
> Appropriate test replication is necessary; however, pseudo-replication may be acceptable for secondary data. Pseudo-replication refers to taking multiple measurements on the same experimental unit and treating each measurement as an independent data point. For example, a common form of pseudo-replication in aquatic toxicity testing is to have just one aquarium for each treatment in a test and then treat each fish exposed within that aquarium as a replicate. A true replicate is the smallest experimental unit to which a treatment is independently applied. Therefore, it is the aquarium in this example that is the replicate.[30]

Preferred test endpoints are the same as those listed for primary data.

4.2.3 *Unacceptable Data*

> Toxicity data that do not meet the criteria of primary or secondary data are unacceptable for WQG derivation purposes. Unacceptable data cannot be used to fulfill minimum data set requirements for any derivation procedure; … [the reasons for rejecting these data must be clearly stated].
>
> Data that are initially classified as unacceptable because insufficient information was reported in the study to assess the adequacy of the test design, procedures, or results, etc., may be upgraded to secondary or primary classification if ancillary information is available from related studies or obtained directly from the author(s).[31]

4.3 Preferred Effect Levels

The results of toxicity tests are generally reported as either hypothesis testing summary statistics (LOEC, NOEC, and MATC) or regression-based statistical data (EC_x and LC_x). NOECs, LOECs and MATCs have been criticized for being arbitrary, and posing the risk of both under- and over-protection (Van der Hoeven 1997; Laskowski 1995; Crane and Newman 2000; Isnard et al. 2001; Landis and Chapman, 2011; van Dam et al. 2012).

> [Therefore,] toxicity endpoints obtained through regression-based statistical data evaluation (e.g. EC_x values identifying no- or low-effect thresholds) are preferred over endpoints obtained through hypothesis-based statistical data evaluation (i.e. NOEC [no-observed-effect-concentration] and LOEC [lowest-observed-effect-concentration values]). When the desired regression based EC_x values are not presented in a toxicological study of

[30] CCME 2007, Part II, Section 1, Page 6-7
[31] CCME 2007, Part II, Section 1, Page 7

interest but sufficient information is provided, the desired EC_x values should be calculated for WQG derivation where feasible.

The use of toxicity data from a test where an insufficient concentration range on the higher end has been tested (i.e. [unbounded NOEC] where the results are expressed as "toxic concentration is greater than *x*") are generally acceptable, as they will not result in an under-protective WQG. These types of data are best used as supporting evidence for other studies and to help to fill minimum data requirements for guideline derivation. However, scientific judgement must be applied in their evaluation as primary or secondary data and in how many such data points should be included in the WQG derivation. Issues to consider are the percentage of "greater than" data points compared to the whole data set and how they compare to other data. It must be reasonable to assume that the tested organism is insensitive toward the tested substance.

Toxicity data from tests where an insufficient concentration range on the lower end has been tested (i.e. [unbounded LOEC] where the results are expressed as "toxic concentration is less than *x*") are not acceptable, as they may result in an under-protective guideline.[32]

Generally, the preferred endpoints for developing long-term chronic WQGs are the respective EC_x of a standard test (e.g. published by ECCC, OECD, USEPA, or ASTM) or another test deemed acceptable, where the EC_x value has been derived by regression analysis of the toxicological data and it has been demonstrated to be at or near the no-effects threshold. However, it is understood that this information may not always be readily available in sufficient quantity to meet the minimum data requirements and low-effect data may be included to satisfy the minimum data requirements.

For the development of the long-term chronic WQGs, a threshold level for no negative effects is generally defined as 10% or less of the exposed individuals of a species (i.e. EC_{10}), unless a more appropriate no-effects threshold is defined for the test species in a generally accepted standardized test protocol (i.e. the most appropriate EC_x representing a no-effects threshold for the species). The default level of 10% is chosen to allow comparison of results and support statistical robustness. Similarly, a threshold level for negative effects is generally defined as an effect level on more than 15% – 20% of the exposed individuals of a species (i.e. low-effect level: EC_{15} – EC_{20}). Accepted endpoints can be lethal or nonlethal.

The accepted endpoints for the development of short-term acute WQGs are LC_{50} or equivalent (i.e. EC_{50} for immobility).[33]

The acceptable endpoint effect levels for the different types of WQGs are discussed further in Section 5.

[32] CCME 2007, Part II, Section 1, Page 5-6
[33] CCME 2007, Part II, Section 1, Page 7

4.4 Exposure Period Definition

Data from laboratory experiments can be classified as long or short-term based on the descriptions below.

4.4.1 *Long-Term*

The following exposure periods are generally considered long-term. Shorter exposure periods may be classified as long-term exposures and used in the derivation of the long-term chronic WQGs on a case-by-case basis, using best scientific judgement.

Fish and Amphibians

For fish and amphibians, exposure periods involving juvenile or adult stages of ≥ 21 days in duration, or periods involving eggs and larvae of ≥ 7 days, are considered long-term. An example of a standard toxicity test in this category is the fathead minnow 7-day larval growth and survival test (USEPA 2002a; ECCC, 1992b).

Aquatic Invertebrates

Acceptable data for aquatic invertebrates include nonlethal endpoints from test durations of ≥ 96-h for shorter-lived invertebrates (e.g. *Ceriodaphnia dubia*) (USEPA 2002b; ECCC 1992a), nonlethal endpoints of ≥ 7 day duration for longer-lived invertebrates (e.g. crayfish), and lethal endpoints from tests of ≥ 21 day duration for longer-lived invertebrates. Lethal endpoints from shorter-lived invertebrates from tests < 21-day exposure periods will be considered on a case-by-case basis.

Plants

Acceptable studies for plants are restricted to aquatic and semi-aquatic plants. Plants that are normally found in the riparian zone will be considered on a case-by-case basis. Plants that would normally be found in terrestrial environments are excluded. The exposure of the plants to the test substance must be through the water column. All tests for *Lemna* sp. following standard test protocols (e.g. ECCC 1999b) are generally considered long-term exposures and are acceptable in the derivation of long-term WQGs. Data for other species will be considered on a case-by-case basis.

Algae

All toxicity tests with algae with exposure durations of longer than 24 hours are considered long-term because of the length of the algal life cycle compared to the duration of the exposure. Algal tests with exposure periods shorter than 24 hours and severe effects will be considered on a case-by-case basis. For example, growth and inhibition tests (72-h) and 96-h cell density counts with *Pseudokirchneriella subcapitata* following standard test protocols (e.g. ECCC 1992c) are acceptable for long-term WQG derivation.[34]

[34] CCME 2007, Part II, Section 1, Page 8

4.4.2 *Short-Term*

In general, exposure periods of 96 hours or less are considered appropriate for the derivation of a short-term exposure WQG.

Fish and Amphibians

For fish and amphibians, the effect level for the derivation of a short-term WQG is an LC_{50}. Examples of standard toxicity tests for this category are the 96-h rainbow trout (*Oncoryhnchus mykiss*) LC_{50} (ECCC 1990a), the 96-h threespine stickleback (*Gasterosteus aculeatus*) LC_{50} (ECCC 1990b), or the 96-h fathead minnow LC_{50} (ECCC 1992b).[35]

Where 96-h data are not available, sub-chronic exposure periods (96-h to 7 d) can be used to fulfill the minimum data requirement.

Aquatic Invertebrates

For aquatic invertebrates, the effect level for the derivation of a short-term WQG is a short-term LC_{50} or equivalent (i.e. EC_{50} for immobility). Examples of standard toxicity tests for this category are the 48-h *Daphnia magna* LC_{50} (EC 1990c).[36]

Where acute data are not available, sub-chronic exposure periods (e.g. 72-h *Daphnia magna* LC_{50}) can be used to fulfill the minimum data requirement.

Aquatic Plants

Because of the general lack of toxicity data for aquatic plants, these tests are considered on a case-by-case basis.

Algae

Because of the rapid cell division rate (reproduction rate) in algae, they generally (but not always) have a high resiliency during short-term exposures. Therefore, algal toxicity tests with exposure periods longer than approximately 24 hours are generally considered inappropriate for inclusion in the derivation of short-term WQGs. Algal tests with exposure periods shorter than 24 hours and severe effects should be included in the short-term data set, but each test must be considered and evaluated on a case-by-case basis emphasizing ecological relevance. Algal tests equal to or less than 48 hours may be included in the derivation of the short-term WQG if plant requirements are not met with algal studies equal to or less than 24 hours.[37]

4.5 Minimum Toxicological Data Requirements

Each derivation method has a minimum toxicological data requirement, as specified in the detailed methodologies (see Tables 4.1 and 4.2), depending on whether the WQG is for freshwater or marine environments and whether it is for short-term or long-term exposures. Type A1 long-term chronic WQGs have the most stringent data requirements using only primary, regression-based, no-effect or very low-effect data. Type A2 chronic WQGs can use both primary and secondary no-effect and low-

[35] CCME 2007, Part II, Section 1, Page 8
[36] CCME 2007, Part II, Section 1, Page 8
[37] CCME 2007, Part II, Section 1, Page 8

effect data with a wider range of acceptable effect levels. Type A1 and A2 short-term acute WQGs are similarly differentiated by the quality of data available (see Table 3.1).

Type B WQGs are based on the extrapolation from the lowest available and acceptable toxicity endpoint and require less data points. Type B WQGs can be developed using a minimum of four primary or secondary endpoints.

Table 4-1. Summary of minimum data set requirements for long-term chronic and short-term acute exposure water quality guidelines for freshwater environments.

		Minimum # of data points	Data Class	Required Groups	Preference of Acceptable Endpoints Data
Chronic	A1	10	Primary	F, I, P, A	EC_x/IC_x representing a no-effects threshold > EC_{10}/IC_{10} > $EC_{11\text{-}20}/IC_{11\text{-}20}$
	A2	7	Primary, Secondary	F, I, P	EC_x/IC_x representing a no-effects threshold > EC_{10}/IC_{10} > $EC_{11\text{-}25}/IC_{11\text{-}25}$ > MATC > NOEC > LOEC >nonlethal $EC_{26\text{-}49}/IC_{26\text{-}49}$ > nonlethal EC_{50}/IC_{50}
	B	4	Primary, Secondary	F, I	Most appropriate EC_x/IC_x representing a low-effect threshold > $EC_{15\text{-}25}/IC_{15\text{-}25}$ > LOEC > MATC > nonlethal $EC_{26\text{-}49}/IC_{26\text{-}49}$ > nonlethal EC_{50}/IC_{50} > LC_{50}
Acute	A1	10	Primary	F, I, A	Acceptable LC_{50} or equivalent (e.g. EC_{50} for immobility in small invertebrates)
	A2	6	Primary, Secondary	F, I	
	B	4	Primary, Secondary	F, I	

F: Fish, including at least 1 salmonid and 1 non-salmonid
I: Aquatic or semi-aquatic invertebrates, including at least 1 planktonic crustacean. For semi-aquatic invertebrates, the life stages tested must be aquatic.
P: Aquatic plants, at least 1 freshwater vascular plant or freshwater algal species.
A: Amphibian, aquatic life stage

Table 4-2. Summary of minimum data set requirements for long-term chronic and short-term acute exposure water quality guidelines for marine environments.

		Minimum # of data points	Data Class	Required Groups	Preference of Acceptable Endpoints Data
Chronic	A1	10	Primary	F, I, P	EC_x/IC_x representing a no-effects threshold > EC_{10}/IC_{10} > $EC_{11\text{-}20}/IC_{11\text{-}20}$
	A2	6	Primary, Secondary	F, I, P	Most appropriate EC_x/IC_x representing a no-effects threshold > EC_{10}/IC_{10} > $EC_{11\text{-}25}/IC_{11\text{-}25}$ > MATC > NOEC > LOEC > $EC_{26\text{-}49}/IC_{26\text{-}49}$ > nonlethal EC_{50}/IC_{50}
	B	4	Primary, Secondary	F, I	Most appropriate EC_x/IC_x representing a low-effect threshold > $EC_{15\text{-}25}/IC_{15\text{-}25}$ > LOEC > MATC > $EC_{11\text{-}25}/IC_{11\text{-}25}$ > $EC_{26\text{-}49}/IC_{26\text{-}49}$ > nonlethal EC_{50}/IC_{50} > LC_{50}
Acute	A1	6	Primary	F, I	Acceptable LC_{50} or equivalent (e.g. EC_{50} for immobility in small invertebrates)
	A2	6	Primary, Secondary	F, I	
	B	4	Primary, Secondary	F, I	

F: Fish, including at least 1 temperate species.
I: Aquatic invertebrates, including at least 2 studies on 2 or more marine species from different classes, at least 1 of which is a temperate species.
P: Plants, at least 1 temperate marine vascular plant or marine algal species.

4.5.1 *Minimum number of data points*

Type A WQGs use a statistical approach called the species sensitivity distribution (SSD). The statistical power of an SSD curve increases with the number of data points however toxicological data are often sparse. The minimum number of data points necessary to fit an SSD curve and generate a stable estimate of the 5th percentile is between 10 and 23 (see Section 5.1.1) (Solomon et al. 1996; Wheeler et al. 2002; Zhao and Chen 2016). A minimum of 10 data points is therefore required for a Type A1 WQG. Seven data points is the minimum for Type A2 WQGs, although more are preferred. Type A2 WQGs derived from less than 10 data points will have a higher level of uncertainty associated with the WQG estimate.

When the minimum toxicological data requirements for Type A WQGs cannot be met, the procedure for a Type B WQG (Section 5.2) is used if adequate data exist.

4.5.2 *Geographical Origin of Test Species*

To derive WQGs that are highly relevant to B.C. ecosystems, studies on species native to B.C. are preferred. If there are too few studies on B.C. native species, additional species will be added based on the following preference for geographic location:

B.C. native species > B.C. introduced (non-invasive[38]) species > Canadian native species > Canadian introduced (non-invasive) species > non-Canadian species

Non-Canadian species must be acceptable surrogates for B.C. native species (e.g. must be representative of a taxonomic group in B.C.) and laboratory tests must be conducted under exposure conditions representative of temperate B.C. waters. There is some evidence that the sensitivity of tropical species relative to temperate species is dependent upon the substance tested (Kwok et al., 2007; Jin et al., 2015). Therefore, tropical species will be considered on a case-by-case basis after considering the substance in question and the experimental conditions.

4.5.3 *Number of Studies*

For all WQGs, data for each required taxonomic group (e.g. fish, invertebrates, plants) must come from more than one study to ensure that no systematic error or bias is included in the data set. For freshwater WQGs, at least two studies are required for each taxonomic group. For marine WQGs, data for the fish and invertebrate taxonomic groups must come from separate studies (to be congruous with CCME, 2007; See Tables 5.1-5.4).

4.5.4 *Taxonomic Coverage*

Despite the greater taxonomic diversity of invertebrates compared to vertebrates, and the greater taxonomic diversity of marine ecosystems compared to freshwater ecosystems, the minimum data requirements for vertebrates are equal to or higher than for invertebrates, and are equal for freshwater and marine ecosystems. The respective minimum data requirements are a compromise between the scientific desire for an extensive data set resembling the taxonomic diversity and the reality of data availability.

[38] B.C. invasive species are those listed on the Aquatic Invasive Species of British Columbia database available at: https://catalogue.data.gov.bc.ca/dataset/aquatic-invasive-species-of-british-columbia.

> In freshwater systems, salmonids are generally considered to be among the most sensitive fish and are routinely tested. They are, therefore, included in the minimum data requirement. With respect to invertebrates, Ephemeroptera (mayfly), Plecoptera (stonefly), and Trichoptera (caddisfly) often represent the sensitive end of the insect community spectrum with respect to contaminant exposure and water quality parameters (Versteeg et al. 1999). [Given their sensitivity, a benthic macro-invertebrate (i.e. Ephemeroptera, Plecoptera or Trichoptera) is required for Type A1 WQGs to reduce residual uncertainty]. However, because these insects, as well as amphibians, are not routinely used in toxicity tests, they are not included in the minimum data requirement [for Type A2 WQGs]. [39]

Both amphibians and freshwater mollusks are required for Type A1 WQGs. Populations from both of these taxonomic groups have undergone extensive decline over the past several decades. Amphibians are now considered the most threatened class of vertebrates (Wake and Vredenburg 2008) and have shown high sensitivity to some groups of contaminants (Kerby et al. 2010). Similarly, about 10% of freshwater mollusk species are on the IUCN Red List of Threatened Species (Lydeard et al. 2004) and in North America, nearly 70% of all freshwater mussel species are endangered, threatened, of special concern or already extinct (Williams 1993). Mollusks have shown high sensitivity to some contaminants such as metals and ammonia (Cope et al. 2008). Given their conservation status and sensitivity to contaminants, both amphibians and freshwater mollusks are required for a Type A1 WQG.

4.5.5 *Freshwater and Marine Species*

> As substances can elicit different toxic effects in freshwater and marine environments because of the fundamental differences in the chemistry of these two types of waterbodies, freshwater toxicity data and marine toxicity data are used to derive the freshwater and marine WQGs, respectively. However, to compensate for the paucity of marine toxicity data for many substances, for substances for which no significant influence on chemical behaviour can be shown or reasonably anticipated, and where no differences in toxicity toward freshwater and marine organisms (by comparison of similar taxonomic groups) can be seen, toxicity data from freshwater organisms may be used on a case-by-case basis to broaden the marine database.[40]

[39] CCME 2007, Part II, Section 1, Page 9
[40] CCME 2007, Part II, Section 1, Page 9

5. GUIDELINE DERIVATION

In B.C., two approaches can be used to derive WQGs based on the quantity and quality of toxicity data available for a given substance: a SSD approach referred to as Type A (see Section 5.1); and a deterministic approach referred to as Type B (see Section 5.2). The flow chart in Figure 5.1 shows the steps necessary to derive each WQG type. Type A WQGs are the preferred approach and can be further categorized as either Type A1 or Type A2. The data requirements of Type A1 WQGs are most stringent in an effort to reduce residual uncertainty. Type A1 WQGs are derived from primary data using only no-effect or very low-effect endpoints and a minimum of 10 data points. Type A2 WQGs are consistent with the CCME Type A approach, allowing fewer and a broader range of data (Table 5.2). Type B WQGs are derived when the Type A minimum data requirements cannot be satisfied. A minimum of four primary and/or secondary data points are required for Type B WQGs. The minimum data requirements for both Type A and Type B WQGs are summarized in Tables 5.1 – 5.4. Once the toxicity data have been compiled, evaluated, and classified, the appropriate WQG derivation method can be determined.

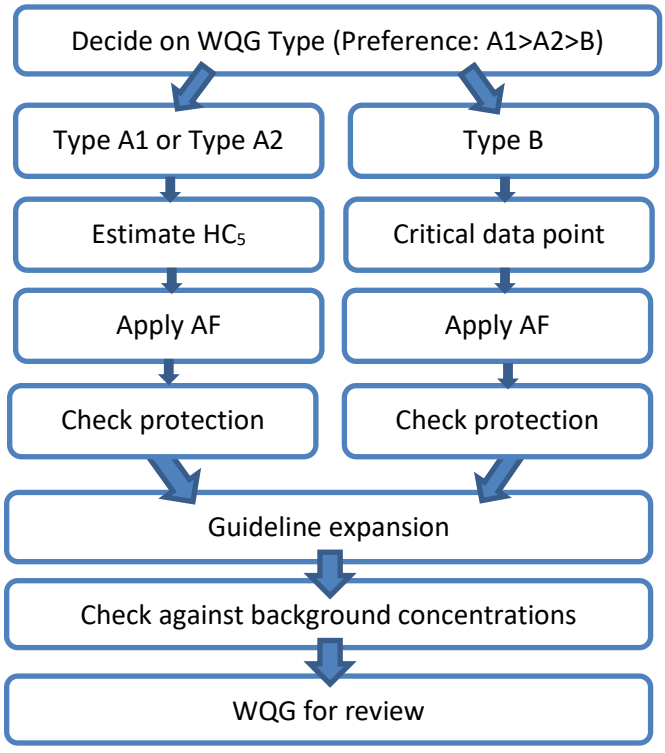

Figure 5.1. Water quality guideline derivation flow chart.

Table 5-1. Minimum data set requirements for the derivation of long-term chronic WQGs for freshwater environments.

<table>
<tr><th></th><th colspan="3">WQG Type</th></tr>
<tr><th></th><th>Type A1</th><th>Type A2</th><th>Type B</th></tr>
<tr><td>Fish</td><td colspan="2">At least two studies on three or more species, including at least one salmonid and one non-salmonid.</td><td>Two species, including at least one salmonid and one non-salmonid.</td></tr>
<tr><td rowspan="2">Aquatic Invertebrates</td><td rowspan="2">At least three aquatic or semi-aquatic invertebrates, at least one of which must be a planktonic crustacean, one must be a mollusk and one must be a mayfly, caddisfly or stonefly. For semi-aquatic invertebrates, the life stages tested must be aquatic.</td><td>At least three aquatic or semi-aquatic invertebrates, at least one of which must be a planktonic crustacean.</td><td>Two aquatic or semi-aquatic invertebrates, at least 1 of which must be a planktonic crustacean.</td></tr>
<tr><td colspan="2">It is desirable, but not necessary, that one of the aquatic invertebrate species be a mayfly, caddisfly, or stonefly. For semi-aquatic invertebrates, the life stages tested must be aquatic.</td></tr>
<tr><td rowspan="3">Plants</td><td colspan="2">At least one study on a freshwater vascular plant or freshwater algal species.</td><td>Toxicity data for plants are highly desirable, but not necessary.</td></tr>
<tr><td colspan="3">The substance is considered phyto-toxic if a toxicity study indicates a plant or algal species is among the most sensitive species.</td></tr>
<tr><td colspan="2">For phyto-toxic substances, three studies on freshwater plant or algal species are required.</td><td>For phyto-toxic substances, two studies on freshwater plant or algal species are required.</td></tr>
<tr><td>Amphibians</td><td>At least one amphibian. Data must represent fully aquatic stages.</td><td colspan="2">Toxicity data for amphibians are highly desirable, but not necessary. Data must represent fully aquatic stages.</td></tr>
<tr><td rowspan="3">Preferred Endpoints</td><td>Acceptable endpoints representing no- or very low-effect thresholds in the following order:
most appropriate EC_x/IC_x representing a no-effects threshold > EC_{10}/IC_{10} > $EC_{11\text{-}20}/IC_{11\text{-}20}$</td><td>Acceptable endpoints representing the no- or low-effect thresholds in the following order:
most appropriate EC_x/IC_x representing a no-effects threshold > EC_{10}/IC_{10} > $EC_{11\text{-}25}/IC_{11\text{-}25}$ > MATC > NOEC > LOEC > $EC_{26\text{-}49}/IC_{26\text{-}49}$ > nonlethal EC_{50}/IC_{50}</td><td>An acceptable endpoint representing a low-effect threshold for a species is used as the critical data point.
Preferred endpoints are ranked in the following order: most appropriate EC_x/IC_x representing a low-effect threshold > $EC_{15\text{-}25}/IC_{15\text{-}25}$ > LOEC > MATC > $EC_{26\text{-}49}/IC_{26\text{-}49}$ > nonlethal EC_{50}/IC_{50} > LC_{50}</td></tr>
<tr><td colspan="3">If multiple data exist for the same species, the lowest endpoint for the most sensitive life stage will be represented. If there is more than one comparable record (i.e. same species, same life stage, same endpoint, same exposure duration), then the species effects endpoint is to be represented by the geometric mean of these records.</td></tr>
<tr><td colspan="3">Less-preferred endpoints may be added sequentially to fulfill the minimum data requirement condition.</td></tr>
<tr><td>Data Quality Requirements</td><td>Primary no-effect and very low-effect level data.</td><td>Primary and secondary no-effect and low-effect level data.</td><td>Primary and secondary low-effect data.</td></tr>
</table>

Table 5-2. Minimum data set requirements for the derivation of long-term chronic WQGs for marine environments.

	WQG Type		
	Type A1	**Type A2**	**Type B**
Fish	At least three studies on three or more temperate marine fish species.	At least three studies on three or more marine fish species, at least one of which is a temperate species.	At least two studies on two or more marine fish species, at least one of which is a temperate species.
Aquatic Invertebrates	At least two studies on two or more temperate marine species from different classes.	At least two studies on two or more marine species from different classes, at least one of which is a temperate species.	At least two studies on two or more marine species.
Plants	At least one study on a temperate marine vascular plant or marine algal species.		Toxicity data for plants are highly desirable, but not necessary.
	The substance is considered phyto-toxic if a toxicity study indicates a plant or algal species is among the most sensitive species.		
	For phyto-toxic substances, three studies on non-target freshwater plant or algal species are required.		For phyto-toxic substances, two studies on non-target freshwater plant or algal species are required.
Preferred Endpoints	Acceptable endpoints representing no- or low-effect thresholds in the following order: most appropriate EC_x/IC_x representing a no-effects threshold > EC_{10}/IC_{10} > $EC_{11\text{-}20}/IC_{11\text{-}20}$	Acceptable endpoints representing the no- or low-effect thresholds in the following order: most appropriate EC_x/IC_x representing a no-effects threshold > EC_{10}/IC_{10} > $EC_{11\text{-}25}/IC_{11\text{-}25}$ > MATC > NOEC > LOEC > $EC_{26\text{-}49}/IC_{26\text{-}49}$ > nonlethal EC_{50}/IC_{50}	An acceptable endpoint representing a low-effect threshold for a species is used as the critical study. Preferred endpoints are ranked in the following order: most appropriate EC_x/IC_x representing a low-effect threshold > $EC_{15\text{-}25}/IC_{15\text{-}25}$ > LOEC > MATC > $EC_{26\text{-}49}/IC_{26\text{-}49}$ > nonlethal EC_{50}/IC_{50} > LC_{50}
	If multiple data exist for the same species, the lowest endpoint for the most sensitive life stage will be represented. If there is more than one comparable record (i.e. same species, same life stage, same endpoint, same exposure duration), then the species effects endpoint is to be represented by the geometric mean of these records.		
	Less-preferred endpoints may be added sequentially to fulfill the minimum data requirement condition.		
Data Quality Requirements	Primary no-effect and very low-effect level data.	Primary and secondary no-effect and low-effect level data.	Primary and secondary low-effect data.

Table 5-3. Minimum data set requirements for the derivation of short-term acute WQGs for freshwater environments.

	WQG Type		
	Type A1	**Type A2**	**Type B**
Fish	At least two studies on three or more species, including at least one salmonid and one non-salmonid.		Two species, including at least one salmonid and one non-salmonid.
Aquatic Invertebrates	At least three aquatic or semi-aquatic invertebrates, at least one of which must be a planktonic crustacean, one must be a mollusk and one must be a mayfly, caddisfly or stonefly. For semi-aquatic invertebrates, the life stages tested must be aquatic.	At least three aquatic or semi-aquatic invertebrates, at least one of which must be a planktonic crustacean.	Two aquatic or semi-aquatic invertebrates, at least 1 of which must be a planktonic crustacean.
		It is desirable, but not necessary, that one of the aquatic invertebrate species be either a mayfly, caddisfly, or stonefly. For semi-aquatic invertebrates, the life stages tested must be aquatic.	
Plants	Toxicity data for plants are highly desirable, but not necessary.		
	The substance is considered photo-toxic if a toxicity study indicates a plant or algal species is among the most sensitive species and 2 studies on freshwater plants or algal species are required.		
Amphibians	At least one amphibian. Data must represent fully aquatic stages.	Toxicity data for amphibians are highly desirable, but not necessary. Data must represent fully aquatic stages.	
Preferred Endpoints	Acceptable LC_{50} or equivalent (e.g. EC_{50} for immobility in small invertebrates).		
Data Quality Requirements	Primary LC_{50} (or equivalents) data are acceptable.	Primary and secondary LC_{50} (or equivalents) data are acceptable.	Primary and secondary LC_{50} (or equivalents) data are acceptable.

Table 5-4. Minimum data set requirements for the derivation of short-term acute WQGs for marine environments.

	WQG Type		
	Type A1	**Type A2**	**Type B**
Fish	At least three studies on three or more temperate marine fish species.	At least three studies on three or more marine fish species, at least one of which is a temperate species.	At least two studies on two or more marine fish species, at least one of which is a temperate species.
Aquatic Invertebrates	At least two studies on two or more temperate marine species from different classes.	At least two studies on two or more marine species from different classes, at least one of which is a temperate species.	At least two studies on two or more marine species.
Plants	At least 1 study on a temperate marine vascular plant or marine algal species.		Toxicity data for marine plants are highly desirable, but not necessary.
	The substance is considered photo-toxic if a toxicity study indicates a plant or algal species is among the most sensitive species and 2 studies on non-target freshwater plant or algal species are required.		
Preferred Endpoints	Acceptable LC_{50} or equivalent (e.g. EC_{50} for immobility in small invertebrates).		
Data Quality Requirements	Primary LC_{50} (or equivalents) data are acceptable.	Primary and secondary LC_{50} (or equivalents) data are acceptable.	Primary and secondary LC_{50} (or equivalents) data are acceptable.

5.1 Type A Water Quality Guidelines

5.1.1 *General Approach*

Type A WQGs are developed whenever the minimum data requirements are met. The Type A WQG derivation method involves fitting a statistical distribution to the species sensitivity data (Posthuma et al., 2002). The resultant SSD is used to estimate the hazard concentration to 5% of the species (HC_5), which is defined as the intercept of the 5th percentile of the y-axis with the fitted SSD curve (see Figure 5.2). The HC_5 value is divided by an assessment factor to determine a draft WQG value. A final screening of this value is then undertaken to ensure the guiding principles for chronic or acute WQGs are met (see section 5.1.6 for more information on the protection clause).

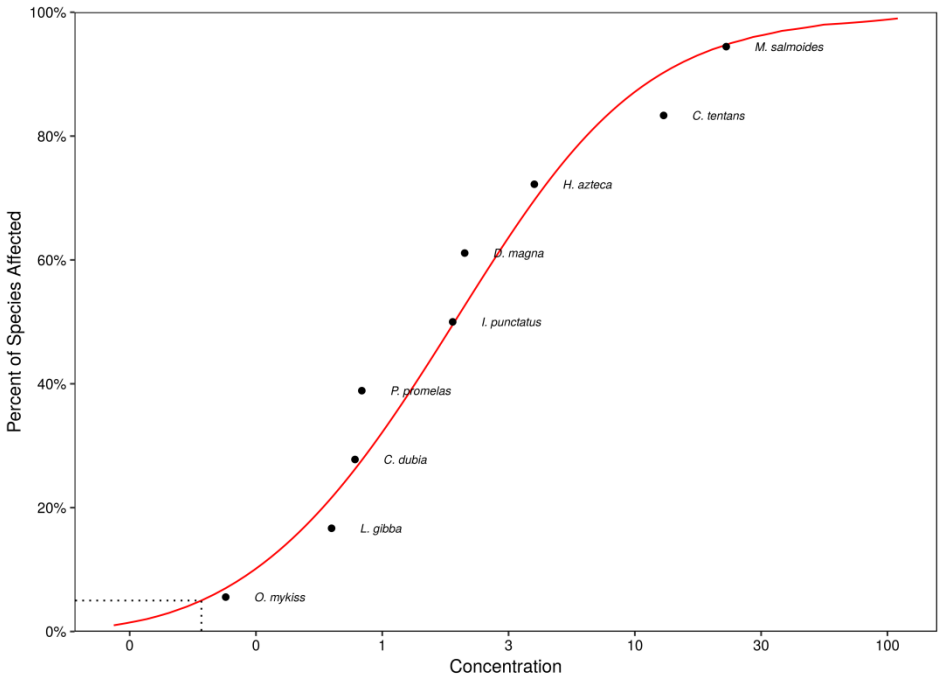

Figure 5.2. An example of a Type A species sensitivity distribution curve (data from CCME 2015). The substance concentration is on the X axis (log scale) and the percent of species affected is on the Y axis. The dotted line in the bottom left corner represents the HC_5 value of the curve.

Type A1 WQGs are based on EC_x/IC_x effect levels at or near the no-effects threshold. Only the most sensitive acceptable endpoint for a given species is included in the analysis (see Section 5.1.2). Each species is represented only once. If there is more than one comparable record (i.e. same species, same life stage, same endpoint, same exposure duration), the species effect concentration is represented by the geometric mean of these records. The acceptable endpoints can be traditional endpoints (e.g. growth, reproduction, and survival) as well as non-traditional endpoints (e.g. behaviour and physiological changes), but only if the ecological relevance of the non-traditional endpoints can be demonstrated (see Section 3.1.1 for more detailed discussion of non-traditional endpoints).

As toxicological studies may be scarce for many substances and organisms, a wider range of acceptable effect levels are available for Type A2 WQGs derivation to fulfill the minimum data requirements for deriving an SSD. This can result in a data set that includes long-term non-lethal toxicity tests with effects ranging from 0% to 50% of a test population and long-term lethal toxicity tests with effects ranging from

0% to 25% of a test population. It is important to identify and clearly label the different data points (i.e. identify species, effect, endpoint, and data classification) in any summary compilation and graphical representations of the distribution so they can be distinguished and analyzed, if necessary, to reveal different patterns and anomalies.

5.1.2 *Acceptable Endpoints*

Type A1 Long-Term Chronic Guideline

Type A1 WQGs are based on no-effect or very low-effect primary data and represent conditions of low risk to aquatic life, based on the information available at the time of their derivation. Though the preferred endpoint for long-term exposure studies is the no-effect EC_x/IC_x, other endpoints may be included to meet the minimum data requirement. The following list provides the acceptable endpoints for a Type A1 WQG, listed from high to low preference.

Most appropriate EC_x/IC_x representing a no-effects threshold > EC_{10}/IC_{10} > $EC_{11\text{-}20}/IC_{11\text{-}20}$

The no- and low-effect endpoints (i.e. up to EC_{20}/IC_{20}) can be lethality endpoints.

Type A1 Short-Term Acute Guideline

The acceptable endpoints for the development of the Type A1 short-term (generally ≤96 h) exposure WQGs are the LC_{50} or equivalent (i.e. EC_{50} for immobility) of a short-term exposure standard test (e.g. published by ECCC, OECD, USEPA, or ASTM), or another acceptable test, where the LC_{50} value has been derived by regression analysis of the toxicological data.

Type A2 Long-Term Chronic Guideline

Type A2 chronic WQGs are synonymous with the Type A chronic WQG described in the CCME aquatic life WQG derivation protocol (CCME 2007).

> The preferred endpoints in the derivation of Type [A2] long-term exposure guidelines is the most appropriate acceptable long-term exposure EC_x/IC_x of a standard test (e.g. published by ECCC, OECD, USEPA, or ASTM), or another acceptable test, where the EC_x value has been derived by regression analysis of the toxicological data and it has been demonstrated to be at or near the no-effects threshold. Though the preferred endpoint for long-term exposure studies is the no-effect EC_x, it is understood that it may not always be available in sufficient quantity to meet the minimum data requirement. The less preferred endpoints may be added to the data set sequentially in the following order if the more preferred endpoint for a given species is not available:

Most appropriate EC_x/IC_x representing a no-effects threshold > EC_{10}/IC_{10} > $EC_{11\text{-}25}/IC_{11\text{-}25}$ > MATC > NOEC > LOEC > nonlethal $EC_{26\text{-}49}/IC_{26\text{-}49}$ > nonlethal EC_{50}/IC_{50}

> The [no- and] low-effect endpoints (i.e. up to EC_{25}/IC_{25}) can be lethality endpoints.[41]

[41] CCME 2007, Part II, Section 3.1, Page 1-2

Type A2 Short-Term Acute Guideline

Type A2 acute WQGs are synonymous with the Type A acute WQG described in the CCME aquatic life WQG derivation protocol (CCME 2007).

The acceptable endpoints for the development of the Type A2 short-term (generally ≤96 h) exposure WQGs are the LC_{50} or equivalent (i.e. EC_{50} for immobility) of a short-term exposure standard test (e.g. published by ECCC, OECD, USEPA, or ASTM), or another acceptable test, where the LC_{50} value has been derived by regression analysis of the toxicological data.

5.1.3 *Derivation Methodology*

Toxicity data are first compiled according to the guidance provided in Section 4, then categorized as short- or long-term exposures, marine or freshwater studies, and classified as primary, secondary, or unacceptable. The evaluation and classification of data is documented in a table (a template is given in Appendix 1) and must be included as an appendix to the technical report for the WQG. To the extent possible, data should be standardized for TMFs to reflect the concentration that would elicit the response in the most appropriate condition (e.g. most sensitive or most common environmental conditions).

Type A WQGs are derived by fitting a model to the species sensitivity data and estimating the 5th percentile (i.e. the HC_5). Distributions are fit using maximum likelihood estimation (MLE) which is a method for fitting a statistical distribution to a univariate data set to maximize the likelihood that the observations came from the chosen statistical distribution. Distributions are fit to the data (see Figure 5.2) and either a single distribution is selected, or a weighted distribution is used to estimate the HC_5 (Schwarz and Tillmanns 2019). A weighted distribution is preferred for the following reasons:

1. it does not depend on the selection of a single best fitting distribution in situations where multiple distributions show a high goodness of fit;
2. in situations where two distributions have equal goodness of fit, but different individual HC_5 values, both will contribute to the weighted average used to determine the final HC_5; and
3. the weighted distribution is relatively stable and not greatly influenced by small changes in the data set.

The ENV has developed software specifically for deriving Type A WQGs using MLE. The *ssdtools* software package is written in R and available on Comprehensive R Archive Network (CRAN) (Thorley and Schwarz 2018) and allows the fitting of distributions to species sensitivity data. An accompanying Shiny Application (Dalgarno 2018) provides a web-based interface to facilitate the use of *ssdtools* by users not familiar with R. The *ssdtools* software and guidance documentation is available for download on CRAN and at:

- https://github.com/bcgov/ssdtools

The ssdtools shiny web application is available at:

- https://bcgov-env.shinyapps.io/ssdtools/

The estimated HC_5 is divided by an assessment factor (AF) (see Section 5.1.5) to create a draft WQG. The protection clause (see Section 5.1.6) is then applied to ensure the draft WQG provides the level of protection described in the guiding principles (see Section 1.3).

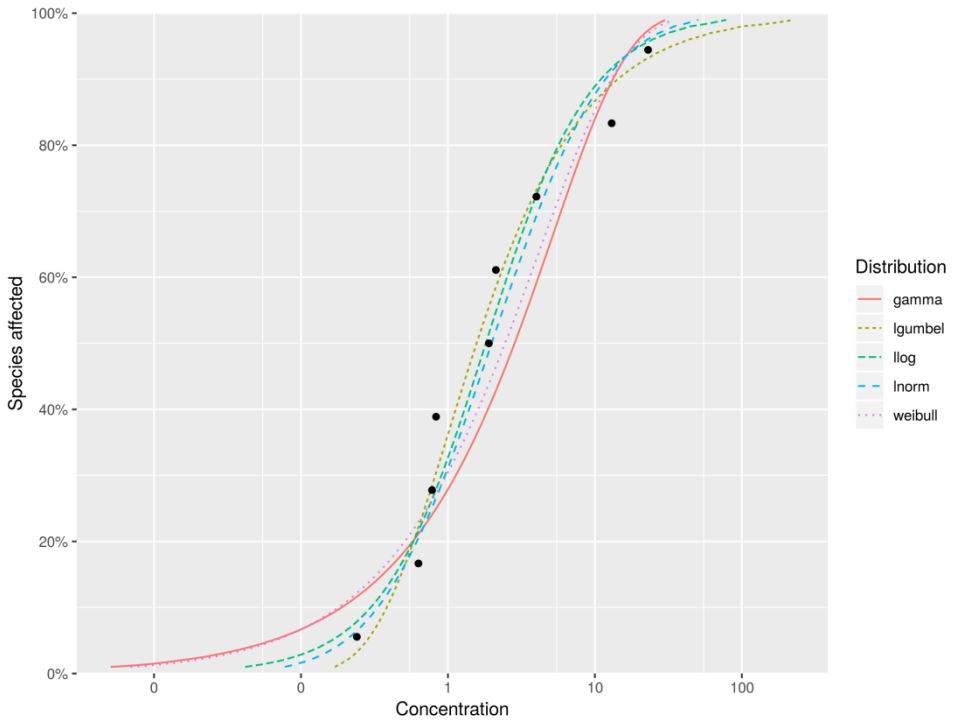

Figure 5.3. An example of several fitted distributions used in maximum likelihood estimation. Distributions plotted include Gamma, Gompertz, Log-Gumbel, Log-logistic, Log-normal, and Weibull.

5.1.4 *Limitations of the SSD Method*

Internationally, the SSD method is the most commonly applied WQG derivation method (Del Signore et al., 2016). However, it does have theoretical and statistical limitations. These limitations need to be recognized to ensure that the advice given to water managers is sound and to prompt further research into ecotoxicological methods for deriving WQGs.

The SSD methodology makes a number of assumptions that are not supported:

- *Species sensitivities can be modelled using a statistical distribution* (Posthuma et al. 2002). This is the underpinning assumption of the SSD approach but there is no mechanistic or physiological hypothesis that supports it and therefore the choice of statistical distribution cannot by informed by biological or ecotoxicological knowledge and is simply a statistical output.
- *Interactions between species do not influence the sensitivity distribution.* Species endpoints used to populate the SSD are taken from tests conducted on individual species. However, Larras et al. (2015) have shown that the species interactions can change species sensitivities (e.g. due to competition).
- *The selected species are representative of ecosystems and therefore can be used to extrapolate a no-effect estimate.* This assumption is problematic for the following reasons:
    - Only a small fraction of the species present in aquatic ecosystems is included in an SSD and these species are often those that are amenable to laboratory-based toxicity testing and may not represent the sensitivity distribution of all species (e.g. Besser et al. 2016).

- The selected species are not an unbiased sample. When using parametric statistics (i.e. using a sample to calculate estimates of a larger population) the sample must be random (Smith and Cairns 1993).
- The proportion of taxonomic groups required by quota systems is not ecologically realistic (i.e. 90% of species in aquatic ecosystems are in the 1st or 2nd trophic levels [e.g. algae and invertebrates] but about half of the species required by quota systems are in the 3rd trophic level [i.e. fish]) (Forbes and Calow 2002).

- *Exposure to concentrations at or below an HC_5 value estimated from low-effect data will not impact aquatic populations.* Population growth rates are dependent upon a myriad of factors and the "sensitivity of population growth rate to changes in individual survival, reproduction and development time varies as a function of life-cycle type and demographic state of the population" (see Forbes and Calow 2002, pg. 474). Therefore, the blanket assumption that a species can tolerate a low level of effect is unwarranted especially given the cumulative effect of multiple anthropogenic stressors.

The limitations of the SSD described above support the use of an AF as a counterbalance. A superior approach would be to develop a method for calculating predicted no-effect concentrations (PNECs) that does not rely on unsound assumptions. However, as a superior method has not yet been developed, B.C. will use the SSD approach coupled with the application of an AF.

5.1.5 *Assessment Factors*

The HC_5 value is divided by an AF to derive the draft WQG. The AF begins with a default value of five that may be reduced or increased depending upon the residual uncertainty of the WQG. The minimum AF to be applied to Type A WQGs is 2 to account for the extrapolation of laboratory testing to field conditions. This approach is similar to those used in other jurisdictions (e.g. EC, 2011).

Type A1 WQGs will have a lower AF than Type A2 WQGs given the more stringent data requirements. Generally, if an A1 WQG has at least 15 data points with complete taxonomic coverage (as given in Tables 5.1-5.4) and no additional residual uncertainty exists (e.g. substance has low toxicity, mode of action has been established and receptors have been tested, etc.), an AF of 2 would be applied.

The evaluation of uncertainty and resultant size of the AF should consider, at a minimum, the following (from EC 2011):

- the taxonomic and life stage representativity of the database;
- knowledge of the mode of action and persistence of the substance;
- whether or not the SSD includes no effect and low effect levels and/or lethal and non-lethal endpoints;
- statistical uncertainties of the HC_5 estimate; and
- the level of agreement between the estimated HC_5 and mesocosm and/or field studies.

5.1.6 *Protection Clause*

Next, the draft WQG is reviewed to ensure that it meets the level of protection described in the guiding principles for long-term chronic and short-term acute guidelines (see Section 1.3). Residual uncertainty should be accounted for with the use of the AF but by definition, the use of a 5th percentile of an SSD will mean that effect concentrations of one or more species are below the HC_5 if there are greater than 20

data points (Figure 5.4). The protection clause is a final test to ensure that there is no undue risk to a specific taxonomic group or species of conservation concern.

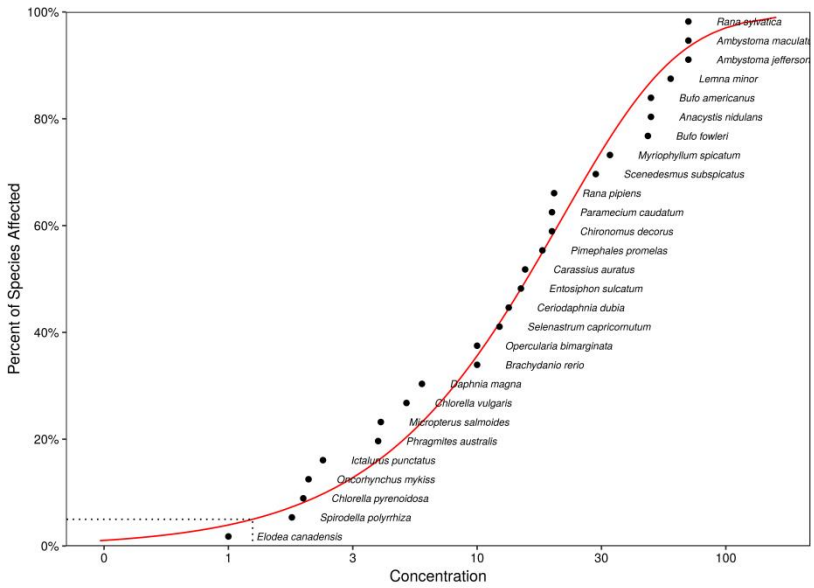

Figure 5.4. An example of a Type A species sensitivity distribution curve with greater than 20 data points (data from CCME 2009). The substance concentration is on the X axis (log scale) and the percent of species affected is on the Y axis. The dotted line in the bottom left corner represents the HC_5 value of the curve.

Long-Term Chronic WQG

The guiding principle for the long term chronic WQG is to be protective of all forms of aquatic life (all species, all life stages including multi-generational) from lethal and negative sub-lethal effects over indefinite exposures.

> The protection clause may be invoked if an acceptable single (or, if applicable, geometric mean) no-effect or low-effect level endpoint (e.g. EC_x for growth, reproduction, survival, or behavior) for a species at risk (as defined by [the B.C. Conservation Data Centre for species that occur in B.C. or] the Committee on the Status of Endagered Wildlife in Canada [COSEWIC]) is lower than the proposed guideline, then that endpoint becomes the recommended guideline value. If this endpoint is a moderate- or sever-effect level endpoint for a species at risk (i.e. EC_x with $x \geq 50\%$ or a lethality endpoint [LC_x]), then the guideline value shall be determined on a case-by-case basis.
>
> Similarly, if an acceptable single (or, if applicable, geometric mean) lethal-effects endpoint (i.e. LC_x where $x \geq 15\%$) for any species is lower that the proposed guideline, then that endpoint becomes the recommended guideline value.
>
> Furthermore, special consideration will be required if multiple endpoints for a single taxon (e.g. fish, invertebrates, or plant/algae) and/or an elevated number of secondary

studies are clustered around the 5th percentile. Best scientific judgement should be used in deciding when this situation is present (e.g. due consideration should be given to the percentage of data points in question to the whole data set) and in determining the best path forward to address this situation.[42]

Short-Term Acute WQG

The guiding principle for the short-term WQG is to be protective against severe effects including lethality (see Section 1.3). This differs from the objective of the short-term acute WQG of CCME that is designed to, "protect only a specified fraction of individuals from severe effects such as lethality for a defined short-term exposure period"[43]. The HC_5 calculated from an SSD of LC_{50} values can be interpreted as having an effect of 50% lethality for 5% of the species. The addition of an assessment factor will offer further protection but a final check is needed to ensure all species are protected against severe effects including lethality. This will be completed by comparing the draft WQG against the LC_{10} values (extracted from the reference) of the five most sensitive species as defined by LC_{50}. The WQG will be the lowest of these values: the draft WQG and the five LC_{10} values.

5.1.7 *Fulfillment of the Guiding Principles*

Using the SSD approach in the derivation of long-term chronic WQGs raises the question as to whether the resulting WQGs fulfill the guiding principle of protecting all forms of aquatic life and all aspects of their aquatic life cycles [Section 1.3].

In the SSD approach, the likelihood of a data point falling below a certain percentile on the y-axis is a function of sample size (i.e. the number of species and endpoints in the SSD in relation to the percentile). For example, with a data set of over 20 data points, at least one data point would fall below the 5th percentile. Therefore, setting the guideline for the 5th percentile alone could be interpreted as allowing for the impairment (and, theoretically, potential loss) of up to 5% of possible species, depending on the severity of the effects endpoints plotted. This issue is of more concern when plotting moderate- or severe-effects level data, but assumed to be less problematic when plotting low- or no-effect level data.

To address this concern, additional safeguards are taken in the development of WQGs when using the SSD approach:

- Data for all available relevant species are plotted.
- The lowest acceptable endpoint for appropriate negative effects for each species is plotted.
- No-effect data are preferred.

The protection clause [Section 5.1.6] can be applied if there is strong reason to believe the resulting WQG is not achieving the intended level of protection.

[42] CCME Part II, Section 3.1, Page 5
[43] CCME Part I, Page 2

While the HC_5 is often lower than the lowest observed low-effect toxicity value (especially for data sets with fewer than 15 data points), the larger the data set, the higher the probability that a low-effect data point will fall below this value, thereby implying that this species may not be sufficiently protected (depending on the kind and severity of effect associated with this data point). Although the WQG is derived preferentially with a no-effect data set (which may include some effects data, especially at the upper part of the concentration range) a low-effect or even a severe-effect endpoint may potentially fall below the recommended WQG value. Consequently, in certain situations, the protection clause may be applied.[44]

5.2 Type B Water Quality Guidelines

5.2.1 *General Approach*

The Type B WQG derivation method presented here is a modified version of the previous method used to derive B.C. WQGs (ENV 2012). It is a generic method of wide applicability that can be used when data are insufficient or inadequate to derive Type A WQGs. In this approach, the long-term chronic WQG is extrapolated from low-effect threshold data, while the short-term acute guideline is extrapolated from severe-effects threshold data.

This method is used to generate Type B WQGs for long- or short-term exposures and for both marine and freshwater environments. These WQGs can be upgraded to Type A WQGs when additional data are available and the minimum data requirements are met. WQGs should be set for the most sensitive or common environmental conditions after the toxicity data have been adjusted (normalized) to that condition according to the relevant TMFs (Section 3). All acceptable data points are plotted and the critical data point, defined as the lowest acceptable endpoint (i.e. the most sensitive LC_{50} or equivalent endpoint) for WQG derivation is identified.

To proceed with the derivation of a Type B WQG, the appropriate minimum physical, chemical, and toxicological data requirements (i.e. the requisite number of studies on fish, invertebrates, and plants, depending on the receiving waters [marine or freshwater]) must be met (see Tables 5.1 – 5.4).

The minimum toxicological data set can be met with primary and/or secondary data. If the minimum data requirement cannot be met, then no Type B WQG will be set. The critical data point used to derive the WQG can be of either primary or secondary quality.

The data requirements for Type B WQGs are summarized in Tables 5.1 – 5.4. The requirements are less restrictive than for Type A, both in terms of quality and quantity of data. This reflects the preference to develop Type A WQGs where possible but recognizes the need for another option when the derivation of a Type A WQGs is not possible.

[44] CCME 2007, Part II, Section 3.1, Page 5

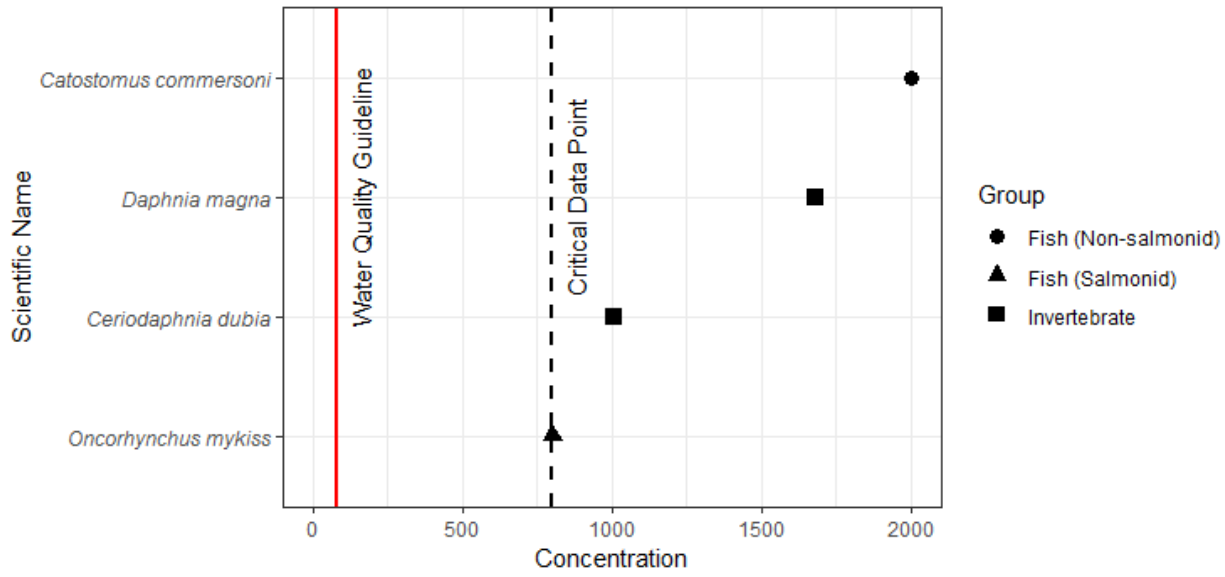

Figure 5.5. An example of a Type B WQG. The dotted line indicates the concentration of the critical data point and the red line indicates the WQG after an AF of 10 has been applied.

5.2.2 *Acceptable Endpoints*

Type B Long-Term Chronic Guidelines

The acceptable endpoints can be traditional endpoints (i.e. growth, reproduction, and survival), as well as non-traditional endpoints (e.g. behaviour and physiological changes), but only if the ecological relevance of these non-traditional endpoints can be demonstrated (see Section 3.1.1).

The preferred acceptable endpoint for Type B long-term chronic WQGs is the most appropriate EC_x/IC_x of a long-term exposure standard test (e.g. published by ECCC, OECD, USEPA, or ASTM), or another acceptable test, where the EC_x value has been derived by a regression analysis of the toxicological data and it has been demonstrated to be at or near the low-effect threshold. Though the preferred endpoint for long-term exposure studies is the EC_x, it may not always be available. Other endpoints are considered acceptable in a tiered approach for developing long-term chronic WQGs in the following order:

Most appropriate EC_x/IC_x representing a low-effect threshold > EC_{15-25}/IC_{15-25} > LOEC > MATC > nonlethal EC_{26-49}/IC_{26-49} > nonlethal EC_{50}/IC_{50}.

The low-effect endpoints (i.e. up to EC_{25}) can be lethality endpoints.

Type B Short-Term Acute Guidelines

The acceptable endpoints for the development of the Type B short-term acute guidelines are the LC_{50} or equivalent (i.e. EC_{50} for immobility) of a short-term exposure standard test (e.g. published by ECCC, OECD, USEPA, or ASTM), or another acceptable test, where the EC_{50} value has been derived by regression analysis of the toxicological data. The lowest scientifically defensible acceptable effects concentration from a short-term exposure study will be the critical data point for the derivation of the short-term acute Type B WQG.

5.2.3 *Derivation Methodology*

The lowest acceptable endpoint (i.e. the most sensitive low-effect endpoint) from a long-term exposure study provides the critical data point for the derivation of a Type B long-term chronic WQG. The lowest acceptable endpoint (i.e. the most sensitive LC_{50} or equivalent endpoint) from a short-term exposure study is the critical data point for the derivation of the Type B short-term acute WQG.

Long-term exposure studies generally show effects at lower concentrations than short-term exposure studies for the same endpoint and species. It is possible, however, that effect concentrations (including EC_{50}) from short-term exposure studies for a species can be below the effect concentrations from long-term exposure studies (due to differences in laboratory conditions, genetic strains, life-stages tested and etc.). In such cases, the long-term chronic WQG may not be sufficiently protective. This situation may occur more frequently for substances with limited data sets (i.e. candidate substances for Type B WQG derivation). While the preference is to derive Type B long-term chronic WQGs from the lowest acceptable endpoint of a long-term exposure study, professional judgement must be used in situations where a short-term endpoint is more sensitive. If it is decided that the resulting long-term chronic WQG would not be sufficiently protective, a suitable low-effect concentration from a short-term exposure study may be used as the critical effect concentration.

5.2.4 *Assessment Factor*

The critical effect concentration is divided by an AF to derive the draft long-term chronic or short-term acute WQG value. The size of the AF is determined after reviewing the following criteria:

- degree of certainty in the effect concentration used as the critical data point
- the taxonomic and life stage representativity of the database;
- knowledge of the mode of action and persistence of the substance; and
- the level of agreement between the lowest effect concentration and mesocosm and/or field studies.

The assessment factor for Type B WQGs will be in the range of 5 – 100 and the information used to inform the decision will be clearly documented in the technical report. The critical data point divided by the AF is used as the draft WQG moving forward.

5.2.5 *Protection Clause*

Long-Term Chronic WQG

The protection clause does not apply to long-term chronic WQGs derived using the B Type approach. Using the lowest effect concentration of the most sensitive species and life stage is the best starting estimate of a WQG when data are sparse. Uncertainty in the data set is accounted for by the assessment factor.

Short-Term Acute WQG

By definition, the selection of the lowest LC_{50} concentration will result in a WQG that does not protect 50% of the individuals of the most sensitive species from lethality. The guiding principle for the short-term WQG is to be protective against severe effects including lethality (see Section 1.3). Therefore a final check is made to ensure that the draft WQG is protective against lethality. This will be completed by comparing the draft WQG against the LC_{10} values (extracted from the studies) of the five (or four if only four data points) most sensitive species as defined by LC_{50}. The WQG will be the lowest of these values: the draft WQG and the five LC_{10} values.

5.3 Guideline Standardization and Expansion

Toxicity data may be standardized to allow the toxicity results conducted under different water quality conditions to be directly compared. Standardization accounts for TMFs, however the resultant draft WQG is only applicable to these standardized water chemistry conditions which typically represent either the most sensitive or most appropriate environmental condition. Standardized WQGs must be expanded through the use of an equation which reverses the procedure used to standardize the toxicity data. The equation requires the input of water chemistry variables allowing the WQG to be applied to site-specific TMF levels.

5.4 Comparison to Background Levels

Before a WQG is finalized, it is compared to the ambient concentrations to ensure the draft WQG value is appropriate for provincial applications. Any relevant TMFs should be included in this evaluation. If the draft WQG is below 50% of the background concentrations in minimally impacted waterbodies, then further investigation is warranted. However, it is common in B.C. for there to be a large range in background concentrations of naturally occurring substances. Therefore, it may be necessary for water managers to evaluate the WQG in respect to the conditions at a specific site.

5.5 Review and Approval

Several steps must be followed to formally establish WQGs as ENV policy. First, the draft technical report is completed and an internal review is conducted within the appropriate provincial and federal agencies. The technical report contains all required information pertaining to the substance of concern, the recommended WQGs, and their application, as described in earlier sections of this protocol document. Comments from the internal review are incorporated into the draft report.

Next, an external review of the draft WQG report is conducted. Scientific experts, other government and non-government stakeholders, ENV staff, and the general public are invited to comment on the draft report, which is made available on the ENV website. The Water Protection and Sustainability Branch maintains a list of external parties that have expressed interest in reviewing WQGs. Members of this list are notified of a draft WQG for review by email. All comments received during this review period are considered. When necessary, additional engagement may be conducted to address specific issues associated with the draft WQG report. The need for such engagement will be determined by the ENV. The external review period is generally one month but may be extended depending on circumstances at the discretion of ENV.

Once all comments and concerns have been addressed and documented, the final WQG document is presented to ENV Executive for approval. Once approved, the WQG is final and the technical report and any supporting documents are posted on the ENV website. All public and stakeholder review comments, and the ENV responses, are available upon request.

6. APPLICATION OF WATER QUALITY GUIDELINES

B.C.'s approved WQGs provide information on PNECs and low-risk conditions. It is recognized that there is a degree of uncertainty associated with the derivation of WQGs and the toxicity data they are based on, and that they may be under-protective or over-protective in certain situations. Therefore, exceeding WQG values does not necessarily mean adverse effects will occur but suggests further investigation may be warranted. While WQGs may provide the basis for standards (e.g. contaminated sites regulations standards), they are not directly enforceable on their own.

6.1 Averaging Period

Water quality is inherently dynamic and when assessing substance concentrations against long-term chronic WQGs, an averaging approach may be taken to allow for some flexibility. When using the averaging approach, the following principles should be followed to ensure adequate protection for aquatic life:

1. WQGs should be compared to the average of five evenly spaced (i.e. weekly) samples collected over a 30-day period
2. When WQGs are reliant on TMFs, a WQG should be calculated for each sample
3. No more than 20% (1 out of 5) of the individual samples should exceed the WQGs and this sample should not exceed the short-term WQG
4. If less than 5 samples are collected, then each sample should meet the long-term chronic WQGs.

It must be stressed that WQGs provide information to aid in the assessment of water quality and to inform natural resource management decisions. Users must employ their own professional scientific judgement in applying the WQGs for assessment or management decision purposes.

6.2 Application of Water Quality Guidelines for Mixtures

WQGs are generally derived using single-substance toxicity tests and may not be protective when mixtures of contaminants are present. Therefore, the application of WQGs to the mixtures of two or more substances may result in under/over protection. More information on the toxic interaction of the substances is required to resolve this issue. It has been shown that mixtures composed of 3 to 30 substances with the same, as well as different modes of toxic action can elicit significant toxic responses even when they are present at their individual EC_{01} concentrations (Vighi et al. 2003). By extension, while a WQG estimates the PNEC of a substance to protect the aquatic environment when acting singly, it cannot automatically be assumed that this is also a scientifically sound PNEC when multiple contaminants are present. It is, therefore, recommended to consider whole effluent toxicity measurements and the development of site-specific science-based environmental benchmarks (SBEBs) when multiple substances are present at concentrations close to their individual WQGs (ENV 2016).

6.3 Considering Background Concentrations

In some situations, background concentrations of naturally occurring substances may be higher at a specific site than the WQG. If it can be verified that the concentration is higher from natural causes (e.g. geology) rather than legacy projects, then the background concentration would be taken as the site-specific WQG unless another value such as a WQO or SBEB had been developed (ENV 2016).

6.4 Dissolved and Total WQGs

WQGs can be derived for either the dissolved (i.e. filtered using a 45µm filter) or total fraction (i.e. non-filtered) of a substance. The choice of chemical fraction is related to the mode of toxic action and the bioavailability of the substance. Some substances, such as metals, are not bioavailable when they are

adsorbed to suspended particulates. Therefore, WQGs based on the dissolved fraction may be developed. However, unless otherwise specified in the technical document, a WQG value refers to the total concentration of the substance in an unfiltered sample.

> The WQG can be applied to the dissolved fraction if it can be demonstrated that the relationship between this fraction and its toxicity is firmly established. Although the dissolved fraction is most often considered to represent a better estimate of the bioavailable fraction of the substance (which cannot reliably be quantified, as it is dependent on the system [environmental conditions and organisms involved]) than the total concentration, the total concentration often symbolizes a more appropriate measure for the conservative derivation of WQGs. It includes the already dissolved fraction, as well as the fraction that may in some cases become soluble when environmental conditions change. Ambient physical and chemical conditions often differ greatly from one location to another, or from effluent conditions. There is no guarantee that the particulate or bound fraction of a substance at one site will not dissolve at another site (i.e. bioaccessibility must be considered). Furthermore, while the bioavailability of a substance bound to a particulate (and, consequently, the toxicity of the particulate [i.e. non-dissolved] fraction) is lower than the dissolved fraction, it is not zero, and should, therefore, be considered.[45]

When using WQGs based on the dissolved fraction, one should consider the environmental fate of the particulate portion of the substance. In some situations, the particulate fraction may settle out and sediment quality guidelines may be used to assess the risk of the settled contaminant to aquatic life. If the particulate portion remains suspended, then downstream conditions should be considered as changing water chemistry conditions may cause the adsorbed fraction to dissolve and become bioavailable.

6.5 Additional Guidance

WQGs are predicted no-effect concentrations, representing low risk conditions, that have been extrapolated from the existing toxicological data sets according to the procedures described herein. WQGs provide the basis for water quality assessments and environmental benchmarks, to inform natural resource sector decisions, and promote stewardship of B.C.'s water resources. In applying the WQGs, professional judgement and critical thinking are required to ensure effective resource management.

[45] CCME 2007, Part II, Section 2, Page 3

REFERENCES

ASTM (American Society for Testing and Materials). 2004. Standard guide for conducting acute toxicity tests on test materials with fishes, macroinvertebrates, and amphibians. In: Annual Book of ASTM Standards. E279-96 (Reapproved 2002). American Society for Testing and Materials, Philadelphia.

Besser, J. M., Dorman, R. A., Hardesty, D. L., and Ingersoll, C. G. 2016. Survival and growth of freshwater pulmonate and nonpulmonate snails in 28-day exposures to copper, ammonia, and pentachlorophenol. Archives of Environmental Contamination and Toxicology, 70: 321-331.

Besser, J. M., Wang, N., Dwyer, F. J., Mayer, F. L., and Ingersoll, C. G. 2005. Assessing contaminant sensitivity of endangered and threatened aquatic species: Part II. Chronic toxicity of copper and pentachlorophenol to two endangered species and two surrogate species. Archives of Environmental Contamination and Toxicology, 48: 155-165.

CCME (Canadian Council of Ministers of the Environment). 1999a. Canadian environmental quality guidelines: Glossary. Canadian Council of Ministers of the Environment, Winnipeg, Manitoba. Available online at: http://ceqg-rcqe.ccme.ca/download/en/94

CCME (Canadian Council of Ministers of the Environment). 1999b. Protocols for deriving water quality guidelines for the protection of agricultural water uses (irrigation and livestock water). Canadian Council of Ministers of the Environment, Winnipeg, Manitoba. Available online at: http://ceqg-rcqe.ccme.ca/download/en/131

CCME (Canadian Council of Ministers of the Environment). 1999c. Protocol for the derivation of Canadian tissue residue guidelines for the protection of wildlife that consume aquatic biota. Canadian Council of Ministers of the Environment, Winnipeg, Manitoba. Available online at: http://ceqg-rcqe.ccme.ca/download/en/290

CCME (Canadian Council of Ministers of the Environment). 2003. Canadian water quality guidelines for the protection of aquatic life: guidance on the site-specific application of water quality guidelines in Canada: Procedures for deriving numerical water quality objectives. In: Canadian environmental quality guidelines, Canadian Council of Ministers of the Environment, 1999, Winnipeg.

CCME (Canadian Council of Ministers of the 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. Available online at: http://ceqg-rcqe.ccme.ca/download/en/220

CCME (Canadian Council of Ministers of the Environment). 2009. Canadian water quality guidelines for the protection of aquatic life: Boron. In: Canadian environmental quality guidelines, 1999. Canadian Council of the Ministers of the Environment, Winnipeg. Available at: http://ceqg-rcqe.ccme.ca/download/en/324/

CCME (Canadian Council of Ministers of the Environment). 2015. Canadian water quality guidelines for the protection of aquatic life: Silver. In: Canadian environmental quality guidelines, 1999. Canadian Council of the Ministers of the Environment, Winnepeg. Available at: http://ceqg-rcqe.ccme.ca/download/en/355/

CFR (Code of Federal Regulations), 2011. Appendix B to Part 136 – Definition and procedure for the determination of the method detection limit – Revision 1.11.

Chowdhury, B.A., and R.K. Chandra. 1987. Biological and health implication of toxic heavy metal and essential trace element interactions. Progress in Food and Nutrition Science, 11: 57-113.

Cope, W.G., Bringolf, R.B., Buchwalter, D.B., Newton, T.J, Ingersoll, C.G., Wang, N., Augspurger, T., Dwyer, F.J., Barnhart, M.C., Neves, R.J. and Hammer, E. 2008. Differential exposure, duration, and sensitivity of unionoidean bivalve life stages to environmental contaminants. Journal of the North American Benthological Society, 27: 451-462.

Crane, M., and Newman, M. C. 2000. What level of effect is a no observed effect? Environmental Toxicology and Chemistry, 19: 516-519.

Dalgarno, D. 2018. ssdtools: A shiny web app to analyse species sensitivity distributions. Prepared by Poisson Consulting for the B.C. Ministry of Environment and Climate Change Strategy. Available online at: https://poissonconsulting.shinyapps.io/ssdtools/

Del Signore, A., A. J. Hendriks, H. J. R. Lenders, R. S. E. W. Leuven, and A. M. Breure. 2016. Development and application of the SSD approach in scientific case studies for ecological risk assessment. Environmental Toxicology and Chemistry, 35: 2149-2161.

Dwyer, F. J., Mayer, F. L., Sappington, L. C., Buckler, D. R., Bridges, C. M., Greer, I. E., Hardesty, D. K., Henke, C. E., Ingersoll, C. G., Kunz, J. L., Whites, D. W., Augspurger, T., Mount, D. R., Hattala, K., and Neuderfer, G. N. 2005. Assessing contaminant sensitivity of endangered and threatened aquatic species: Part I. Acute toxicity of five chemicals. Archives of Environmental Contamination and Toxicology, 48: 143-154.

ECCC (Formerly: Environment Canada). 1990a. Biological test method: acute lethality test using rainbow trout. Environmental Protection Series, Report EPS 1/RM/9 (with amendments). Ottawa.

ECCC (Formerly: Environment Canada). 1990b. Biological test method: acute lethality test using threespine stickleback (*Gasterosteus aculeatus*). Environmental Protection Series, Report EPS 1/RM/10 (with 2002 amendments). Ottawa.

ECCC (Formerly: Environment Canada). 1990c. Biological test method: acute lethality test using *Daphnia* spp. Environmental Protection Series, Report EPS 1/RM/11 (with 1996 amendments). Ottawa.

ECCC (Formerly: Environment Canada). 1992a. Biological test method: test of reproduction and survival using the Cledoceran *Ceriodaphnia dubia*. Environmental Protection Series, Report EPS 1/RM/21 (with 1997 amendments). Ottawa.

ECCC (Formerly: Environment Canada). 1992b. Biological test method: test of larval growth and survival using fathead minnows. Environmental Protection Series, Report EPS 1/RM/21 (with 1997 amendments). Ottawa.

ECCC (Formerly: Environment Canada). 1992c. Biological test method: growth inhibition test using the freshwater alga *Selenastrum capricornutum*. Environmental Protection Series, Report EPS 1/RM/25 (with 1997 amendments). Ottawa.

ECCC (Formerly: Environment Canada). 1994. Manual of analytical methods. Vols. 1, 2, and 3. The National Laboratory for Environmental Testing, Burlington, ON.

ECCC (Formerly: Environment Canada). 1998. Biological test method: toxicity tests using early life stages of salmonid fish (rainbow trout). Environmental Protection Series, EPS 1/RM/28. Ottawa.

ECCC (Formerly: Environment Canada). 1999a. Guidance document on the application and interpretation of single-species tests in environmental toxicology. Environmental Protection Series, EPS 1/RM/34. Ottawa.

ECCC (Formerly: Environment Canada). 1999b. Biological test method: test for measuring the inhibition of growth using the freshwater macrophyte *Lemna minor*. Environmental Protection Series, EPS 1/RM/37. Ottawa.

ECCC (Formerly: Environment Canada). 2005. Guidance document on statistical methods for environmental toxicity tests. Environmental Protection Series, EPS 1/RM/46. Ottawa.

EC (European Commission). 2011. Common Implementation Strategy for the Water Framework Directive (2000/60/EC). Guidance Document No. 27. Technical Guidance for Deriving Environmental Quality Standards. Technical Report – 2011 – 055.

ENV (Formally: British Columbia Ministry of Environment). 2012. Derivation of water quality guidelines to protect aquatic life in British Columbia. Water Protection and Sustainability Branch, Environmental Sustainability and Strategic Policy Division, British Columbia Ministry of Environment. 34 p.

ENV (Formally: British Columbia Ministry of Environment), 2015. British Columbia Environmental Laboratory Manual, Section C, Metals. Environmental Monitoring, Reporting and Economics Knowledge Management Branch, British Columbia Ministry of Environment and Climate Change Strategy. 176 p.

ENV (Formally: British Columbia Ministry of Environment). 2016. A framework for the development and use of freshwater science-based environmental benchmarks for aquatic life in Environmental Management Act permitting for mines. Environmental Protection Division, Regional Operations Branch.

Forbes, V. E., and Calow, P. 2002. Species Sensitivity Distributions Revisited: A Critical Appraisal. Human and Ecological Risk Assessment: An International Journal, 8:473–492.

Health Canada. 2012. Guidelines for Canadian recreational water quality. Ottawa, Ontario.

Health Canada. 2017. Guidelines for Canadian drinking water quality – summary table. Water and Air Quality Bureau, Healthy Environments and Consumer Safety Branch, Health Canada, Ottawa, Ontario.

Isnard, P., Flammarion, P., Roman, G., Babut, M., Bastien, P., Bintein, S., Esserméant, L., Ferard, J.F., Gallotti-Schmitt, S., Saouter, E. and Saroli, M. 2001. Statistical analysis of regulatory ecotoxicity tests. Chemosphere, 45: 659-669.

Jin, X., Wang, Z., Wang, Y., Lv, Y., Rao, K., Jin, W., Giesy J.P., and Leung, K. M. 2015. Do water quality criteria based on nonnative species provide appropriate protection for native species? Environmental Toxicology and Chemistry, 34: 1793-1798.

Johnson, A. C., and Sumpter, J. P. 2016. Are we going about chemical risk assessment for the aquatic environment the wrong way? Environmental Toxicology and Chemistry, 35: 1609–1616.

Kerby, J. L., Richards-Hrdlicka, K. L., Storfer, A., and Skelly, D. K. 2010. An examination of amphibian sensitivity to environmental contaminants: are amphibians poor canaries? Ecology Letters, 13: 60-67.

Kwok, K. W., Leung, K. M., Lui, G. S., Chu, V. K., Lam, P. K., Morritt, D., Maltby, L., Brock, T.C.M., Van den Brink, P. J., Warne, M. St. J., and Crane, M. 2007. Comparison of tropical and temperate freshwater animal species' acute sensitivities to chemicals: implications for deriving safe extrapolation factors. Integrated Environmental Assessment and Management, 3: 49-67.

Landis, W. G., and Chapman, P. M. 2011. Well past time to stop using NOELs and LOELs. Integrated Environmental Assessment and Management, 7: vi-viii.

Larras, F., V. Gregorio, A. Bouchez, B. Montuelle, and N. Chèvre. 2015. Comparison of specific versus literature species sensitivity distributions for herbicides risk assessment. Environmental Science and Pollution Research, 23: 3042–3052.

Laskowski, R. 1995. Some good reasons to ban the use of NOEC, LOEC and related concepts in ecotoxicology. Oikos, 73: 140-144.

Lydeard, C., Cowie, R.H., Ponder, W.F., Bogan, A.E., Bouchet, P., Clark, S.A., Cummings, K.S., Frest, T.J., Gargominy, O., Herbert, D.G., Hershler, R., Perez, K.E., Roth, B., Seddon, M., Strong, E.E., and Thompson, F.G. 2004. The Global Decline of Nonmarine Mollusks. BioScience, 54:321–330.

Meyer, J.S. 2002. The utility of the terms "bioavailability" and "bioavailable fraction" for metals. Marine Environmental Research, 53: 417-423.

OECD (Organization for Economic Co-operation and Development). 1993. OECD guidelines for testing chemicals. Vols. 1, 2, and updates. OECD Publications, Paris.

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 Part C: Toxicology & Pharmacology, 133: 3-35.

Posthuma, L., Suter, G.W., and Traas, T.P. 2002. Species sensitivity distributions in ecotoxicology. Lewis Publishers, Boca Raton, FL, USA.

Schwarz, C.J. and A.R. Tillmanns. 2019. Improving statistical methods to derive species sensitivity distributions. Water Science Series, WSS2019-07, Province of British Columbia, Victoria.

Smith, E. P., and Cairns, J. 1993. Extrapolation methods for setting ecological standards for water quality: statistical and ecological concerns. Ecotoxicology, 2: 203–219.

Solomon KR, Baker DB, Richards RP, Dixon DR, Klaine SJ, LaPoint TW, Kendall RJ, Weisskopf CP, Giddings JM, Giesy JP, Hall LW, and Williams WM. 1996. Ecological risk assessment of atrazine in North American surface waters. Environmental Toxicology and Chemistry, 15: 31-74.

Thorley, J. and Schwarz, C. 2018. ssdtools: Species Sensitivity Distributions. R package version 0.0.1.9002.

USEPA. 1985a. Methods for measuring acute toxicity of effluents to freshwater and marine organisms. 3rd ed. EPA/600/4-85/013. U.S. Environmental Protection Agency, Office of Research and Development, Cincinnati, OH.

USEPA. 1985b. Short-term methods for estimating the chronic toxicity of effluents and receiving waters to freshwater organisms. 3rd ed. EPA/600/4-85/014. U.S. Environmental Protection Agency, Office of Research and Development, Cincinnati, OH.

USEPA. 1985c. Guidelines for deriving numerical national water quality criteria for the protection of aquatic organisms and their uses. U.S. Environmental Protection Agency, Office of Research and Development, Washington, DC.

USEPA. 1985d. 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.

USEPA. 1995. Short-term methods for estimating the chronic toxicity of effluents and receiving waters to west coast marine and estuarine organisms. 1st ed. EPA/600/R-95-136. U.S. Environmental Protection Agency, Office of Research and Development, Cincinnati, OH.

USEPA. 2002a. Short-term methods for estimating the chronic toxicity of effluents and receiving waters to marine and estuarine organisms. 3rd ed. October 2002. EPA-821-R-02-014. U.S. Environmental Protection Agency, Office of Water (4303T), Washington, DC.

USEPA. 2002b. Methods for measuring the acute toxicity of effluents and receiving waters to freshwater and marine organisms. 5th ed. 3rd ed. October 2002. EPA-821-R-02-012. U.S. Environmental Protection Agency, Office of Water (4303T), Washington, DC.

Van Dam, R. A., Harford, A. J., and Warne, M. S. J. 2012. Time to get off the fence: the need for definitive international guidance on statistical analysis of ecotoxicity data. Integrated Environmental Assessment and Management, 8: 242-245.

Van Der Hoeven, N. 1997. How to measure no effect. Part III: statistical aspects of NOEC, ECx and NEC estimates. Environmetrics: The Official Journal of the International Environmetrics Society, 8: 255-261.

Versteeg, D.J., S.E. Belanger, and G.J. Carr. 1999. Understanding single species and model ecosystem sensitivity: data-based comparison. Environmental Toxicology and Chemistry 18: 1329-1346.

Vighi, M., R. Altenburger, A. Arrhenius, T. Backhaus, W. Bödeker, H. Blanck, F. Consolaro, M. Faust, A. Finizio, K. Froehner, P. Gramatica, L.H. Grimme, F. Grönvall, V. Hamer, M. Scholze, and Walter, H. 2003. Water quality objectives for mixtures of toxic chemicals: problems and perspectives. Ecotoxicology and Environmental Safety, 54: 139-150.

Wake, D. B., and Vredenburg, V. T. 2008. Are we in the midst of the sixth mass extinction? A view from the world of amphibians. Proceedings of the National Academy of Sciences, 105(Supplement 1): 11466-11473.

Wang, N., Ingersoll, C. G., Dorman, R. A., Brumbaugh, W. G., Mebane, C. A., Kunz, J. L., and Hardesty, D. K. 2014. Chronic sensitivity of white sturgeon (*Acipenser transmontanus*) and rainbow trout (*Oncorhynchus mykiss*) to cadmium, copper, lead, or zinc in laboratory water-only exposures. Environmental Toxicology and Chemistry, 33: 2246-2258.

Wheeler JR, Grist EPM, Leung KMY, Morritt D, and Crane M. 2002. Species sensitivity distributions: data and model choice. Marine Pollution Bulletin, 45: 192-202.

Williams, J. D., M. L. Warren, K. S. Cummings, J. L. Harris, and R. J. Neves. 1993. Conservation status of freshwater mussels of the United States and Canada. Fisheries 18: 6–22.

Zhao, J., and B. Chen. 2016. Species sensitivity distribution for chlorpyrifos to aquatic organisms: Model choice and sample size. Ecotoxicology and Environmental Safety, 125: 161–169.