Development of Site-Specific Water Quality Objectives for Faro Mine Complex: Phase I Evaluation

Report Prepared for:

Assessment and Abandoned Mines Branch
Energy, Mines and Resources
Government of Yukon
Whitehorse, Yukon

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1.0 INTRODUCTION

1.1 Background

Preliminary water quality predictions for the Faro Mine complex have indicated that generic Canadian water quality guidelines (CWQG) for protection of aquatic life (CCME 199) are likely to be exceeded for some parameters (Senes 2006). While the generic guidelines were designed to be protective of all aquatic biota inhabiting a wide range of habitat types and chemical conditions across Canada, they may be either over- or under-protective on a site-specific basis. As a result, site-specific water quality objectives (SSWQOs) are sometimes developed (CCME 2003) to take into account local biological and chemical characteristics that may influence the manifestation of contaminant effects.

Two distinct strategies are recognized by the Canadian Council of Ministers of the Environment (CCME 2003) for water quality guideline development: a) antidegradation, and b) use protection. The antidegradation strategy seeks to protect aquatic resources by preventing water quality impairment (a science-based approach), while the use protection strategy may allow for some degradation of water quality provided designated water uses are protected and reasonable preventative/mitigative measures are taken (a policy-based approach). Water uses that need to be considered include drinking water supplies, recreation, aquatic life, wildlife, agriculture, and tissue quality (the latter being for protection of human and wildlife consumers). Industrial water use is another possible designation that is officially recognized by U.S. authorities (e.g., follow applicable links presented at http://www.epa.gov/waterscience/standards/about/uses.htm), but is less openly acknowledged by Canadian authorities. Where it applies, protection of aquatic life tends to be the most sensitive (limiting) water use and CCME guidance for SSWQO development is focused accordingly (CCME 2003). However, other procedures can be followed in cases where there is agreement that a water body is designated for other uses. Thus, SSWQO are developed to provide the level of protection necessary to maintain the water body for its most sensitive designated use(s). They are used as the benchmarks against which current or predicted water quality is compared. Discharge limits and assessment criteria can also be developed, through back calculation based on effluent dilution, to ensure protection of the designated water use(s).

1.2 Development of SSWQO at Faro

Recognizing that future water quality at the Faro Mine complex may not achieve generic CWQGs for protection of aquatic life some parameters, the Faro Mine Closure Planning

Office initiated a project in 2007 to consider the development of SSWQOs. The project was divided into three phases:

Phase I: Review existing information regarding current and future water quality, as well as toxicological and ecological data to identify contaminants for which SSWQOs should be developed and, for each one, identify the best general approach for doing so;

Phase II: Develop detailed methodology for deriving the SSWQOs, based on established procedures and taking into account input from expert reviewers; and

Phase III: Follow the methodology developed in Phase II to develop the SSWQOs and make specific recommendations for their implementation at the Faro mine complex.

Minnow Environmental Inc. was retained by the Faro Mine Closure Office to complete Phase I, which will recommend whether or not SSWQO development should proceed for any contaminants of current or future concern (COCs) at Faro and, if so, recommend the general approach and schedule.

1.3 Overview

The Phase I SSWQO project involved:

- summarizing existing and predicted contaminant concentrations downstream of the Faro Mine complex relative to water quality guidelines for protection of aquatic life to identify contaminants of current or future concern (COCs);
- reviewing and summarizing information regarding the aquatic chemistry of each COC;
- summarizing data describing the toxicity of each COC to sensitive organisms;
- identifying the water quality factors that modify the toxicity of each COC;
- summarizing data for potential modifying factors in surface waters near the Faro Mine complex;
- indicating, of the species identified as being sensitive to each COC, which ones are known to be or are potentially present in surface waters at or near Faro; and
- making recommendations based on the above as to which, if any, COCs may be amenable to SSWQO development, and which of the available procedures for SSWQO development is/are most appropriate.

A more detailed description of the specific tasks involved in the Phase I SSWQO development project is presented in the sections that follow.

The conclusions of this assessment may be affected by new information generated through updates to CWQG for COCs (e.g., new CWQGs for Cd and Zn are anticipated in 2008), as well as updates to water quality predictions and re-evaluation of background station data during the Environmental Assessment process associated with closure of the Faro Mine complex.

1.4 Document Organization

Section 2.0 describes the approach and methods used in this evaluation, as well as relevant background technical information deemed necessary to support the descriptions of steps taken in the assessment. Section 3.0 identifies the contaminants considered to be of current or future concern at Faro (COCs). The aquatic chemical and toxicological characteristics of each COC are discussed in Section 4.0. Section 5.0 describes background (reference area) water quality and the concentrations in surface waters near the Faro Mine complex of substances that are known to modify the toxicity of each COC. Section 6.0 summarizes and integrates the above information for each parameter to identify which, if any, procedure is expected to yield a SSWQO that is higher than the respective generic CWQG. Information sources cited throughout the document are listed in Section 7.0.

2.0 APPROACH AND METHODS

2.1 Identification of Contaminants of Concern

Contaminants of current or future concern (COCs) were identified from reviews of:

1. GLL (2005), which listed 31 “parameters of interest” for the Faro Mine and presented a rationale for subsequent SSWQO development for zinc (GLL 2006).

2. Senes (2006), which involved a prospective risk assessment based on predictions of future water quality by SRK Consulting Inc., and

3. Minnow (2007), which reviewed, summarized and assessed historical water, sediment and biological data.

Minnow (2007) compared contaminant concentrations recently measured in surface waters downstream of the Faro Mine complex to generic Canadian Water Quality Guidelines (CWQG) for protection of aquatic life. Although interpretation of data for cadmium and silver was limited by poor (high) analytical method detection limits (MDLs) relative to the generic CWQG for those substances, the evaluation generally indicated that no substances possessing a CWQG are routinely found at concentrations exceeding the guideline at the present time (Minnow 2007). Also, water quality predictions reported by Senes (2006) were compared to generic CWQG to identify substances for which mean annual predicted concentrations are expected to exceed the applicable guideline in receiving waters in the future even after site remediation (Table 2.1). The predictions were made by SRK Consulting Inc. for a plausible remediation scenario under two independent water quality conditions (Future 2, Future 3) involving different assumptions of contaminant loading from waste rock and tailings (Senes 2006). These were considered representative of the range of uncertainty inherent in the prediction methodology, with the Future 2 condition having the greatest probability of representing peak future contaminant loading from the site. Substances that were predicted to substantially exceed CWQG (e.g., by more than 5X) included cadmium (Cd), copper (Cu), iron (Fe), lead (Pb), sulphate, and zinc (Zn) (Table 2.2).

Sulphate has no CWQG, but British Columbia has established an “alert level” of 50 mg/L and a maximum criterion of 100 mg/L for protection of aquatic life (BCMOE 2000, 2006). The alert level was used for screening Faro surface water data (Tables 2.1 and 2.2). However, the B.C. guidelines were partly based on toxicity (at 250 mg/L sulphate) to a bass species that does not occur in British Columbia or the Yukon. Also, attempts to verify other key study results upon which the BC guidelines were based (Davies et al. 2003; Davies and Hall 2007;

Table 2.1: Future maximum monthly concentrations predicted for two independent, future water quality conditions based on a plausible site remediation scenario (data from SRK presented by Senes (2006))[a]. Future 2 conditions are considered most likely.

Parameter	Units	Water Quality Criteria[b]	Predicted Water Quality			
			Future 2		Future 3	
			Rose	Vangorda	Rose	Vangorda
Aluminum	mg/L	0.1[c]	0.09	0.08	0.43	0.08
Arsenic	mg/L	0.005	0.004	0.005	0.022	0.005
Cadmium	mg/L	0.00003[e]	0.0029	0.0030	0.0070	0.0040
Cobalt	mg/L	0.004[d]	0.0033	0.0030	0.0083	0.0060
Copper	mg/L	0.002[e]	0.019	0.004	0.133	0.019
Iron	mg/L	0.3	1.0	0.2	4.9	0.4
Lead	mg/L	0.002[e]	0.014	0.015	0.015	0.015
Manganese	mg/L	1.0[d,e]	0.06	0.03	0.68	0.28
Nickel	mg/L	0.065[e]	0.014	0.016	0.017	0.019
Silver	mg/L	0.0001	0.0001	0.0003	0.0002	0.0003
Sulphate	mg/L	50[d]	151	14.3	486	17.8
Zinc	mg/L	0.03	0.445	0.052	4.92	0.727

[a] Each value for Rose Creek is the maximum concentration predicted for any month at either X2 or X14 under each future condition. Each value for Vangorda Creek is the maximum predicted concentration for any month at either V27 or V8 under each future condition.

[b] Canadian water quality guideline unless indicated otherwise (footnote c)

[c] for pH >6.5

[d] BCMOE (2000, 2006) "alert" concentration

[e] based on hardness of 100 mg/L, which is conservative because it is less than the mean hardness of stations exhibiting elevated levels of mine-related contaminants.

Table 2.2: Factors by which predicted receiving water concentrations (Table 2.1) are expected to exceed applicable generic water quality criteria. Shading indicates predicted concentrations will be at least 5 times higher than the respective criterion. Factors rounded to nearest whole number.

Parameter	Units	Water Quality Criteria[d]	Factor by Which Water Quality Criterion will be Exceeded			
			Future 2		Future 3	
			Rose	Vangorda	Rose	Vangorda
Aluminum	mg/L	0.1[a]	1	1	4	1
Arsenic	mg/L	0.005	1	1	4	1
Cadmium	mg/L	0.00003[b]	97	100	233	133
Cobalt	mg/L	0.004[c]	1	1	2	2
Copper	mg/L	0.002[b]	10	2	66	10
Iron	mg/L	0.3	3	1	16	1
Lead	mg/L	0.002[b]	7	8	8	8
Manganese	mg/L	1.0[b,c]	0	0	1	0
Nickel	mg/L	0.065[b]	0	0	0	0
Silver	mg/L	0.0001	1	3	2	3
Sulphate	mg/L	50[c]	3	0	10	0
Zinc	mg/L	0.03	15	2	164	24

[a] for pH >6.5

[b] based on hardness of 100 mg/L, which is conservative because it is less than the mean hardness of stations exhibiting elevated levels of mine-related contaminants.

[c] BCMOE (2000, 2006) "alert" level

[d] Canadian water quality guideline unless indicated otherwise (footnote c)

Davies 2007), as well as other data presented in the scientific literature (Mount and Gulley 1992; Singleton 2000), indicate that sulphate concentrations less than 500 mg/L are unlikely to harm aquatic biota and higher concentrations can be tolerated by many species. Sulphate concentrations downstream of the Faro Mine complex have rarely exceeded 500 mg/L in the past (only occasionally at X14; Minnow 2007), nor are they expected to exceed this level under most future remediation scenarios (Table 2.1). Furthermore, it is presently uncertain if, in the absence of CWQGs for substances, guidelines from other jurisdictions such as B.C. will be formally applied to future water quality evaluations at Faro. Therefore, Cd, Cu, Fe, Pb, and Zn (not sulphate) were identified as future COCs for which it was considered relevant to consider SSWQO development. Of these, Cd and Zn are believed to be of greatest concern, while Cu, Fe, and Pb may be of less or no concern on the basis of updated water quality predictions which should be available in early 2008 (D. Hockley, SRK Consulting, November 2007, pers. comm.).

2.2 Background Concentrations

In highly mineralized areas, surface waters may have natural concentrations of substances that consistently or occasionally exceed generic CWQG. In such cases, it is unreasonable to expect that generic water quality guidelines will be consistently achieved and the concentration representing the upper limit of regional background concentrations can be established as a SSWQO (Appendix A.1). Therefore, data for reference stations monitored at Faro during the period 2005-2007, inclusive, were summarized and the 95th percentile of each COC was computed (Appendix B). The 95th percentile statistic was selected based on a previous evaluation of this versus alternative statistics for characterizing background data at Faro (Minnow 2007). The 95th percentiles presented in this report differ from those reported by Minnow (2007) based on different data sets – only data for the period 2005-2007 were used herein and data collected for special projects (with low analytical method detection limits) were also incorporated to augment available routine monitoring data. The 95th percentiles were compared to generic CWQG to identify any COCs that are naturally elevated in local surface waters.

2.3 Toxicology of COCs

For SSWQO development procedures other than the Background Calculation Procedure (Appendix A), an understanding of the toxicology of the COCs is required. In particular, the Water Effect Ratio Procedure (WERP) and Biotic Ligand Modelling (BLM; Appendix A) involve characterization of the water quality factors that influence the bioavailability (and therefore toxicity) of each COC, while the Recalculation Procedure (RCP) and Resident Species Procedure (RSP; Appendix A) involve characterization of relative species sensitivities to each COC. Therefore, available literature related to the aquatic toxicity of each COC was summarized, including aqueous chemistry, concentrations associated with toxicity to different aquatic species, and the factors that modify aquatic toxicity. This information was then assessed with respect to its relevance to surface waters near the Faro Mine complex as discussed in more detail in the following sections.

2.3.1 Information Sources

Toxicity data and supporting information were obtained from the Canadian (CCREM 1987, CCME 1999) and American (USEPA 1976, 1985, 1986, 1987, 2001, 2007, 2008) water quality guideline documents. As most of the current CWQG for metals were developed two decades ago, this information was updated by downloading more recent toxicity data from the U.S. EPA's EcoTox database (http://cfpub.epa.gov/ecotox). As this database did not include the most recent 2-3 years of published scientific literature, additional searches were done using readily available electronic search engines including those available through the University of Guelph (e.g., Scholars Portal) and internet (e.g., Google Scholar). Key scientific articles were electronically downloaded, including most recent relevant publications, those reporting toxicity at low concentrations, and those that focused on characterization of toxicity modifying factors or SSWQO development for the same contaminants. Toxicity study results were not evaluated in detail with respect to the quality acceptability criteria typically employed by regulatory agencies when developing water quality guidelines (e.g., CCME 1991). Nor was there any attempt to separately report or to standardize the specific toxicity endpoints (e.g., LC50, EC10, NOEC, IC25) for all studies; this was not necessary for the purpose for which the data were used (i.e., identifying the variety of species sensitive to low concentrations of each COC). Neverthess, much of the data listed in tables presented in Section 4.0 were taken from U.S. EPA water quality criteria documents (and referenced accordingly) and thus were subjected to detailed scrutiny prior to incorporation into those documents.

2.3.2 Aquatic Chemistry and Behaviour

The identified COCs are metals (Section 2.1), which may exist as free cationic metal ions as well as inorganic and organic complexes and compounds (Janssen et al. 2003, Batley et al. 2004, Hyne et al. 2005). Metal speciation is influenced by water quality factors, such as pH, alkalinity, oxidation-reduction state, and the amount and type of organic material present (Hockett and Mount 1996, Di Toro et al. 2001). For example, some surface waters naturally contain organic acids or minerals that form relatively stable, nontoxic chemical complexes with certain metals (Giesy and Alberts 1982, Hockett and Mount 1996, Di Toro et al. 2001; De Schamphelaere et al. 2005). Metal speciation affects the bioavailability, and thus, toxicity of metals in aqueous solutions (Campbell and Stokes 1985, Campbell 1995, Batley et al. 2004, Niyogi and Wood 2004). Therefore, information regarding aqueous chemical speciation was summarized for each COC.

2.3.3 Mode of Toxicity

Toxicity of most metals is generally thought to be due to the free metal ion (Campbell 1995; Paquin et al. 2002, Niyogi and Wood 2004). Most metals appear to exert toxicity by binding to specific sites on respiratory cell surfaces (e.g., fish gills), where they interfere with regulation of sodium or calcium (Paquin et al. 2002, Niyogi and Wood 2004). The consequence of this ionoregulatory dysfunction is that there is a re-distribution of ions and water between the internal fluid compartments of the fish, such as a decrease in levels of plasma sodium, chloride, and other ions, which in turn triggers a sequence of events that potentially leads to cardiovascular collapse and death (Paquin et al. 2002). There are at least two types of saturable binding sites: high-affinity, low capacity binding sites versus low-affinity, high-capacity binding sites, which have different but overlapping properties and may vary in relative importance in acute versus chronic exposures (Niyogi and Wood 2004). Metals differ in their relative binding affinities for the specific ion transport binding sites, explaining relative differences in toxicity (Paquin et al. 2002, Niyogi and Wood 2004). For this project, the specific mode of action for each of the COCs was briefly summarized to provide the basis for discussion of water quality factors that modify toxicity.

2.3.4 Modifying Factors

Water quality factors can influence metal toxicity in two ways. As noted above, factors such as pH, alkalinity, oxidation-reduction state, and the presence/absence of complexing ligands may influence the speciation of metals and thus affect the proportion of toxic, free metal ion in solution. Water quality factors may also influence toxicity by competing with metal ions for binding sites on cell surfaces (Paquin et al. 2002, Niyogi and Wood 2004). Many studies from which data were taken to develop water quality criteria for protection of aquatic life involved toxicity tests in laboratory water with low or limited concentrations of substances known to ameliorate metal toxicity. Consequently, metal toxicity in natural surface waters containing such modifying factors may be lower than would be predicted by the generic guidelines. For each of the COCs, the water quality factors that influence metal speciation and metal binding on cell surfaces of biota were summarized. The concentrations of potential modifying factors measured in the surface waters near Faro (2005-2007) were summarized and discussed in terms of how they may influence SSWQO development.

2.3.5 Species Sensitivity and Relevance to Faro

The data used to derive existing water quality guidelines in Canada and the U.S. were reviewed to identify the species most sensitive to each COC. The toxicity data for sensitive species and endpoints were tabulated and augmented by any recent literature which also showed toxicity at low concentrations. The relevance to the Faro site of species identified as being sensitive to each COC was evaluated based on the general geographic distribution of each species, if such information was readily available, and/or any previous study reports that documented the presence of the same or closely related species near the Faro Mine complex.

2.4 Parameter-Specific Evaluation for Potential SSWQO Development

Background concentrations of each COC were compared to the generic CWQG to determine the potential for developing a SSWQO that exceeds the current generic CWQG using the background concentration procedure (BCP; Appendix A). Aquatic biota found or expected to be found at Faro were compared to those shown in the scientific literature to be sensitive to each COC to determine the potential for using the recalculation procedure (RCP; Appendix A) or resident species procedure (RSP; Appendix A) for SSWQO development. Concentrations measured at the Faro Mine complex of water quality factors known to affect the toxicity of each COC were also evaluated to determine if the WERP or BLM predictions (Appendix A) will be likely to result in a SSWQO greater than the CWQG.

The above information was summarized for each COC to develop and present a rationale for identifying any COCs that may be amenable to development of a SSWQO and proposing the most suitable procedure.

3.0 CONTAMINANTS OF CONCERN

As noted in Section 2.1, Minnow (2007) concluded that contaminants having a CWQG (protection of aquatic life) have rarely been measured at concentrations exceeding such values in receiving waters downstream of the Faro Mine complex, although improved analytical detection limits are necessary to verify this for some contaminants (e.g., cadmium and silver). Based on water quality predictions presented by Senes (2006), it was evident that cadmium, copper, iron, lead and zinc are the contaminants of greatest future concern (i.e., predicted to exceed CWQG by greatest magnitude) sometime in the future, even after the remediation associated with mine closure. Therefore, this project investigates the potential development of SSWQG for these substances.

4.0 AQUATIC CHEMISTRY AND TOXICITY OF COCS

4.1 Cadmium

4.1.1 Aquatic Chemistry

Cadmium is a relatively rare element, ranking 64th in relative abundance of elements in the earth's crust at concentrations of about 0.2 mg/kg (CCREM 1987). It is a minor nutrient for plants at low concentrations (Lane and Morel 2000, Lee et al. 1995, Price and Morel 1990), but has no nutritional value for animals (USEPA 2001). Cadmium concentrations in uncontaminated surface waters are typically <0.1 ug/L (Hollis et al. 2000a). In most well oxygenated fresh waters with low organic carbon content, free divalent cadmium (Cd^{2+}) will be the predominant form (USEPA 2001).

Cadmium has only a moderate affinity for dissolved organic material and binding decreases with increased water hardness (Pentinnen et al. 1998). Adsorption of cadmium onto soil particles, silicon or aluminum oxides is strongly pH-dependent, increasing as conditions become more alkaline (USEPA 2008). However, cadmium has considerably less affinity for such absorbents than copper, zinc, and lead and is therefore relatively more mobile in aquatic environments (USEPA 2008). The portion of cadmium absorbed to mineral surfaces (e.g., clay) or organic materials may be released in the dissolved state when sediments are disturbed (USEPA 2008). Cadmium found in association with carbonate minerals, precipitated as stable solid compounds, or co-precipitated with hydrous iron oxides may be less readily mobilized by re-suspension of sediments or biological activity (USEPA 2008). Desorption from sediment particles may also be enhanced at pH below 6-7 (Van Deuren et al. 2002). Overall, precipitation of cadmium by carbonate or hydroxide and formation of soluble complexes by chloride, sulfate, carbonate, and hydroxide is usually of limited importance in aquatic environments (USEPA 2001).

4.1.2 Mode of Toxicity and Modifying Factors

The Cd^{2+} ion is the most bioavailable and toxic form of Cd to aquatic organisms (Pagenkopf 1983). Cadmium is taken up through chloride cells in external membranes of fish (e.g., gills) and invertebrates (Verbost et al. 1989, Wicklund-Glynn et al. 1994; Silvestre et al. 2004). These are the cells regulating calcium uptake from water (Galvez et al. 2006). At cell surfaces, Cd^{2+} competes with Ca^{2+} for specific, high-affinity calcium-binding sites (Playle 1998, Playle et al. 1993a, 1993b; Croteau and Luoma 2007), although low-affinity, high-capacity binding sites may also play a role in cadmium toxicity, at least for some aquatic species (Niyogi et al. 2004). Binding site characteristics are altered in chronic versus acute exposures to calcium or cadmium (Hollis et al. 2000a, 2000b; Niyogi et al. 2004).

The effect of the above is that cadmium blocks calcium uptake and causes calcium deficiency (hypocalcemia) (Roch and Maly 1979; Verbost et al. 1987, 1989). Unlike for zinc (Section 4.2.2), the inhibition of calcium uptake by cadmium may not be readily reversible (Reid and McDonald 1988). Internally, cadmium accumulates primarily in kidney, liver, and gills of fish (Benoit et al. 1976, Hollis et al. 1999, 2000a, 2000b).

At acute waterborne concentrations, Cd severely disrupts Ca homeostasis, which ultimately leads to death (Verbost et al. 1987; Reid and McDonald 1988). Cadmium effects may also be manifested as reduced growth (Peterson et al. 1983, Borgmann and Ralph 1986) and/or reduced reproductive performance (Dobson 1992, Brown 1994), although growth is not always a more sensitive endpoint than survival (Eaton et al. 1978, Anderson 1980; Hollis et al. 2000a, 2000b). Survival was the most sensitive test endpoint for *Hyallela azteca* exposed to cadmium in water whereas *H. azteca* exposed to dietary cadmium showed growth effects at concentrations lower than those affecting survival (Ball et al. 2006). This suggests that dietary cadmium may have a different mechanism of toxicity than water-borne cadmium. Reproduction of *Daphnia magna* was affected at aqueous cadmium concentrations almost an order of magnitude lower than those causing mortality (Barata and Baird 2000). Typically, the maximum cadmium concentration that can be tolerated by a species is 0.002 times the acute lethal concentration or higher (Marshall 1979, U.S. EPA 2001). Among fish, swim-up alevins or young juveniles appear to be more sensitive to cadmium exposure than younger or older life-stages (Benoit et al. 1976, Chapman 1978, Eaton et al. 1978, Buhl and Hamilton 1991).

Cadmium toxicity to biota may be greater from waterborne than dietary exposure (Abel and Barlocher 1988, Spry et al. 1988, Sofyan et al. 2007) or the converse (Roy and Hare 1999). Greatly elevated Cd levels in diet were required compared to waterborne exposures to achieve the same gill burden in rainbow trout, suggesting that the gut provides a better barrier than the gill to internal accumulation (Szebedinszky et al. 2001). At similar cadmium burden in gills, trout with dietary pre-exposure had lower accumulation in liver and kidney of new Cd from water than in trout pre-exposed to cadmium via water only, indicating the route of cadmium exposure affects cadmium uptake and tissue distribution characteristics (Szebedinszky et al. 2001).

Among various water quality characteristics that could potentially influence Cd uptake and toxicity (e.g., hardness, pH, alkalinity, and dissolved organic matter (DOM)), hardness is often reported to be the dominant factor (Calamari et al. 1980; Hollis et al. 1997, 2000b;

Pentinnen et al. 1998; Hansen et al. 2002). Cadmium binding affinity at uptake sites (Niyogi et al. 2004, Croteau and Luoma 2007) and acute Cd toxicity (Carroll et al. 1979; Pascoe et al. 1986, Hollis et al. 2000a, 2000b) decrease as water hardness (particularly the calcium component) increases. This is likely because calcium out-competes Cd for binding sites on external cell surfaces (Spry and Wiener 1991). Carroll et al. (1979) found that calcium, but not magnesium, sodium, sulphate, or carbonate, reduced the acute toxicity of cadmium. Cadmium toxicity to fish and invertebrates was reduced 4- to 8-fold by 3- to 4-fold increases in water hardness or calcium concentration (Calamari et al. 1980, Penttinen et al. 1998, Hollis et al. 2000b, Hansen et al. 2002).

Cadmium toxicity to bull and rainbow trout was up to 3 times lower at pH 6.5 than 7.5, suggesting that H^+ ions may also compete with Cd for binding sites on fish gills (Hansen et al. 2002). Cadmium toxicity to a variety of organisms in very hard water was either lower or remained the same when tested at progressively lower water pH from 8-8.5 to 7-7.5 or 6-6.5 (Schubauer-Berigan et al. 1993). Water pH had only a slight influence on benthic invertebrate species tested over a pH range of 3.5 to 4.5 (Mackie 1989). Effects of pH >8 on cadmium toxicity are still poorly characterized (Niyogi and Wood 2004). Water pH appears to affect the number and/or characteristics of metal binding sites of algae, whereas the binding characteristics for higher organisms is believed to be independent of the test medium characteristics (Heijerick et al. 2002a, Francios et al. 2007). Thus, protons appear to have a negligible to moderate protective influence on cadmium toxicity as pH declines from 8 to 6.

Giesy et al. (1977) found that dissolved organics substantially reduced the toxicity of cadmium to daphnids, but had little effect on cadmium toxicity to fish. Cadmium complexation with dissolved organic matter (DOM of 7.58 mg/L) eliminated cadmium effects on *Ceriodaphnia dubia* survival and reproduction that had been observed in water with similar hardness but lower DOM (low DOM concentration not reported) (Brooks et al. 2004). A small (<2-fold) reduction in cadmium toxicity to *Daphnia magna* was observed in lake water with high humic acid (measured as dissolved organic carbon - DOC 20 mg/L) compared to water with low DOC (2.0 mg/L) when water hardness was low (≤20 mg/L as CaCO3), but DOC had no effect on toxicity at higher water hardnesses (50-250 mg/L) (Penttinen et al. 1998). Similarly, cadmium toxicity to zebrafish was reduced by humic acids in soft water, but not in hard water (Meinelt et al. 2001). Some studies have reported enhanced cadmium toxicity in the presence of DOM, perhaps due to the effects of DOM on other water quality factors affecting cadmium toxicity (e.g., DOM effect on pH or available calcium), rather than directly affecting cadmium toxicity (Penttinen et al. 1998). For example, a slight increase in cadmium toxicity was observed among *Daphnia pulex* exposed to cadmium in soft water when 0.5 mg/L humic acid was added (Stackhouse and Benson 1988). This may have been due to complexation of available Ca^{2+} by the humic acid, thereby reducing the mitigating influence of calcium on cadmium toxicity. The same study showed that cadmium toxicity was reduced by more than 3-fold when humic acid concentrations were increased 10-fold to 50 mg/L. The influence of DOC on cadmium toxicity may also depend on the size (high versus low molecular weight; Voets et al. 2004) and type (e.g., humic versus fulvic acids; Koukal et al. 2003) of DOC present. Generally, the affinity of cadmium for dissolved organic matter is believed to be much less than that of some other metals such as copper (Penttinen et al. 1998, Niyogi and Wood 2004).

Overall, hardness (particularly the calcium component) appears to be the most important modifier of Cd toxicity compared to other cations (Na+, H+) or DOM/DOC

Unlike for some metals, pre-exposure to sublethal cadmium concentrations followed by subsequent exposure to lethal cadmium concentrations has not always resulted in the increased tolerance (higher LC50s) indicative of an acclimation response. Szebedinszky et al. (2001) reported that dietary, but not aqueous pre-exposure to low cadmium concentrations significantly reduced acute cadmium toxicity to rainbow trout. Cadmium tolerance increased by 15- to 20-fold among adult rainbow trout pre-exposed to sublethal cadmium concentrations in water (Stubblefield et al. 1999). Juvenile trout tested in the same study were more tolerant of cadmium than adults, and pre-exposure increased juvenile tolerance by a much smaller amount (≤2 times) (Stubblefield et al. 1999). Acclimation resulted in changes in cadmium tolerance of <5 times for trout and other species tested in other studies (Stubblefield et al. 1999). Acclimation occurred among juvenile rainbow trout previously exposed to sublethal levels of cadmium in soft water (hardness of 30-54 mg/L, calcium 10-20 mg/L; Hollis et al. 2002b) but not in harder water (hardness of 64-128, calcium 25-50 mg/L; Hollis et al. 2002b) nor in softer water (hardness 20 mg/L, calcium 5 mg/L; Hollis et al. 2000a). Variable results in acclimation tests may pertain to differences in fish size (and associated metabolic differences) and/or pre-exposure cadmium concentration relative to the toxic level for each species and life-stage (Stubblefield et al. 1999).

4.1.3 Sensitive Species and Relevance to Faro

The maximum cadmium concentration predicted for surface water downstream of Faro after closure and remediation was approximately 7 ug/L (Table 2.1), so toxicity data were tabulated for organisms showing cadmium toxicity at concentrations of 10 ug/L or less (Table 4.1). The USEPA (2001) used an average hardness-toxicity relationship to adjust acute toxicity test results reported in the literature to a common hardness of 50 mg/L (Table 4.1).

Table 4.1: Summary of studies reporting cadmium toxicity to aquatic biota at concentrations ≤ 10 µg/L.

Source	Type	Scientific Name	Common Name	Hardness (mg/L as CaCO3)	Effect Concentration [a] (µg/L)	Hardness Adjusted Value [b] (µg/L)	Reference [c]
EPA 2001	Acute	*Oncorhynchus mykiss*	Rainbow trout (1130 mg)	30.2	0.38	0.63	Stratus Consulting 1999
		Oncorhynchus mykiss	Rainbow trout (659 mg)	29.3	0.47	0.81	Stratus Consulting 1999
		Oncorhynchus mykiss	Rainbow trout (1150 mg)	31.7	0.51	0.81	Stratus Consulting 1999
		Daphnia magna (genotype A)	Water flea	170	3.6	1.04	Baird et al. 1991
		Oncorhynchus mykiss	Rainbow trout (263 mg)	30.7	0.71	1.17	Stratus Consulting 1999
		Daphnia magna (genotype B)	Water flea	170	4.5	1.30	Baird et al. 1991
		Morone saxatilis	Striped bass (larva)	34.5	1	1.46	Hughes 1973
		Salvelinus confluentus	Bull trout (76.1 mg)	30.7	0.91	1.49	Stratus Consulting 1999
		Salvelinus confluentus	Bull trout (218 mg)	30.2	0.9	1.50	Stratus Consulting 1999
		Oncorhynchus mykiss	Rainbow trout (289 mg)	89.3	2.85	1.58	Stratus Consulting 1999
		Salvelinus confluentus	Bull trout (221 mg)	31.7	1	1.59	Stratus Consulting 1999
		Salmo trutta	Brown trout	43.5	1.4	1.61	Spehar and Carlson 1984a,b
		Morone saxatilis	Striped bass (63 d)	285	10	1.70	Palawski et al. 1985
		Salvelinus confluentus	Bull trout (200 mg)	29.3	0.99	1.71	Stratus Consulting 1999
		Salvelinus fontinalis	Brook trout	42	<1.5	<1.79	Carroll et al. 1979
		Oncorhynchus mykiss	Rainbow trout (juv)	41	1.5	1.84	Buhl and Hamilton 1991
		Oncorhynchus mykiss	Rainbow trout (299 mg)	30	1.29	2.17	Stratus Consulting 1999
		Oncorhynchus mykiss	Rainbow trout (parr)	23	1	2.20	Chapman 1978
		Daphnia magna (genotype A-1)	Water flea	170	9	2.59	Baird et al. 1991
		Daphnia magna (genotype A-2)	Water flea	170	9	2.59	Baird et al. 1991
		Oncorhynchus mykiss	Rainbow trout	43.5	2.3	2.65	Spehar and Carlson 1984a,b
		Oncorhynchus tshawytscha	Chinook salmon (juv)	21	1.1	2.66	Finlayson and Verrue 1982
		Daphnia magna (<24hr)	Water flea	170	9.5	2.74	Guilhermino et al. 1996
		Oncorhynchus mykiss	Rainbow trout (fry)	9.2	<0.5	<2.80	Cusimano et al. 1986
		Oncorhynchus mykiss	Rainbow trout	31	1.75	2.85	Davies 1976
		Oncorhynchus tshawytscha	Chinook salmon (juv)	25	1.41	2.85	Chapman 1982
		Oncorhynchus mykiss	Rainbow trout (swim-up)	23	1.3	2.86	Chapman 1975, 1978
		Morone saxatilis	Striped bass (fingerling)	34.5	2	2.92	Hughes 1973
		Salvelinus confluentus	Bull trout (72.7 mg)	89.3	6.06	3.36	Stratus Consulting 1999
		Oncorhynchus mykiss	Rainbow trout (8.8 g)	44.4	3	3.39	Phipps and Holcombe 1985
		Oncorhynchus tshawytscha	Chinook salmon (swim-up)	23	1.8	3.96	Chapman 1975, 1978
		Tanytarsus dissimilis	Chironomid	47	3.8	3.98	Anderson et al. 1980
		Oncorhynchus kisutch	Coho salmon (juv)	41	3.4	4.16	Buhl and Hamilton 1991
		Salvelinus confluentus	Bull trout (84.2 mg)	30	2.89	4.86	Stratus Consulting 1999
		Morone saxatilis	Striped bass (63 d)	40	4	5.02	Palawski et al. 1985
		Oncorhynchus kisutch	Coho salmon (parr)	22	2.7	6.22	Chapman 1975
		Oncorhynchus tshawytscha	Chinook salmon (smolt)	23	>2.9	>6.39	Chapman 1975, 1978
		Oncorhynchus tshawytscha	Chinook salmon	23	3.5	7.71	Chapman 1975, 1978
		Oncorhynchus mykiss	Rainbow trout (smolt)	23	4.1, >2.9	9.03, >6.39	Chapman 1975
		Daphnia magna (<24hr)	Water flea	51	9.9	9.70	Chapman et al. Manuscript
		Utterbackia imbecilis	Mussel (juv)	39	9	11.59	Keller and Zam 1991
		Simocephalus serrulatus	Water flea	11.1	7	32.33	Giesy et al. 1977
		Daphnia magna	Water flea	-	<1.6	-	Anderson 1948
		Daphnia magna (one instar)	Water flea	-	2.47	-	Barata and Baird 2000
		Daphnia magna (two instars)	Water flea	-	3.34	-	Barata and Baird 2000
		Oncorhynchus mykiss	Rainbow trout	-	6	-	Kumada et al. 1973
		Oncorhynchus mykiss	Rainbow trout	-	6	-	Kumada et al. 1980
		Oncorhynchus mykiss	Rainbow trout (2 mo)	-	6.6	-	Hale 1977
		Oncorhynchus mykiss	Rainbow trout	-	7	-	Kumada et al. 1973
	Chronic	*Daphnia magna*	Water flea	103	0.21	0.12	Chapman et al. Manuscript
		Daphnia magna	Water flea	53	0.15	0.15	Chapman et al. Manuscript
		Daphnia magna	Water flea	209	0.44	0.15	Chapman et al. Manuscript
		Hyalella azteca	Amphipod	280	0.98	0.28	Ingersoll and Kemble Unpublished
		Daphnia magna	Water flea	130	<1.9	<0.92	Borgmann et al. 1989
		Oncorhynchus mykiss	Rainbow trout (270d)	250	4.3	1.31	Brown et al. 1994
		Salvelinus fontinalis	Brook trout	37	1.7	2.17	Sauter et al. 1976
		Salvelinus fontinalis	Brook trout	44	2	2.25	Eaton et al. 1978
		Oncorhynchus kisutch	Coho salmon (Lk. Supr)	44	2.1	2.31	Eaton et al. 1978
		Oncorhynchus tshawytscha	Chinnok salmon	25	1.6	2.61	Chapman 1975
		Salvelinus fontinalis	Brook trout	44	2.4	2.64	Benoit et al. 1976
		Chironomus tentans	Midge	280	10	2.80	Ingersoll and Kemble Unpublished
		Daphnia magna	Water flea	150	7.1	3.13	Bodar et al. 1988b
		Aplexa hypnorum	Snail	45.3	3.5	3.72	Holcombe et al. 1984
		Jordanella floridae	Flagfish	47.5	4.4	4.59	Carlson et al. 1982
		Jordanella floridae	Flagfish	47.5	5	5.18	Carlson et al. 1982
		Daphnia pulex	Water flea	65	7.5	6.17	Niederlehner 1984
		Aplexa hypnorum	Snail	45.3	5.8	6.24	Holcombe et al. 1984
		Jordanella floridae	Flagfish	44	5.8	6.37	Spehar 1976a
		Salmo trutta	Brown trout	44	6.7	7.33	Eaton et al. 1978
		Catostomus commersoni	White sucker	44	7.1	7.80	Eaton et al. 1978
		Oncorhynchus kisutch	Coho salmon (West Coast)	44	7.2	7.87	Eaton et al. 1978
		Salmo salar	Atlantic salmon	23.5	4.5	7.92	Rombough and Garside 1982
		Salvelinus namaycush	Lake trout	44	7.4	8.09	Eaton et al. 1978
		Esox Lucius	Northern pike	44	7.4	8.09	Eaton et al. 1978
		Micopterus dolomieui	Smallmouth bass	44	7.4	8.12	Eaton et al. 1978
		Pimephales promelas	Fathead minnow	44	10	10.99	spehar and Fiandt 1986
	Plants	*Asterionella formosa*	Diatom	-	2	-	Conway 1978
		(mixed sp.)	Algae	11.1	5	-	Giesy et al. 1979
		Scenedesmus quadracauda	Diatom	-	6.1	-	Klass et al. 1974
		Chara vulgaris	Alga	-	9.5	-	Heumann 1987
		Salvina natans	Fern	-	10	-	Hutchinson and Czyrska 1972
		Lemna valdiviana	Duckweed	-	10	-	Hutchinson and Czyrska 1972
Other Recent Data		*Oncorhynchus mykiss*	Rainbow trout	30	0.35	-	Hansen et al. 2002
		Salvelinus confluentus	Bull trout	30	0.83	-	Hansen et al. 2002
		Hydra viridissima	Green hydra	20	3	-	Holdway et al. 2001
		Hydra vulgaris	Pink hydra	20	8.5	-	Holdway et al. 2001

[a] Data represent a variety of endpoints (eg. LC50 or ED50 for acute tests and some chronic tests, as well as highest no-effect concentrations or chronic threshold values for chronic tests).

[b] As reported by USEPA (2001) for hardness of 50 mg/L as $CaCO_3$

[c] References cited in USEPA 2001, except for more recent studies which are listed in Section 7.0.

Lowest effect concentrations (greatest toxicity) were reported for rainbow trout (≥0.6 ug/L), *Daphnia magna* (≥1 ug/L), striped bass (≥1.4 ug/L), Chinook salmon (≥3 ug/L), Coho salmon (≥4 ug/L), other salmonid species (≥1.5 ug/L), and the chironomid *Tanytarsus dissimilus* (4 ug/L), with all other species showing hardness-adjusted acute toxicity at concentrations >10 ug/L. The same data reported at the original test hardness values are also presented in Table 4.1. Hardness-adjusted chronic toxicity thresholds were <10 ug/L for most of the 21 fish and invertebrate species for which data were available (USEPA 2001). Additional toxicity tests conducted since that time (2001) have also shown cadmium toxicity at low concentrations (Table 4.1).

Some of the aquatic biota that are sensitive to cadmium are not found in Canada, let alone rivers of the Yukon (Table 4.2). Of all the species which appear to be very sensitive to cadmium, Chinook salmon is the only one previously reported in waters downstream of the Faro Mine complex (Table 4.2). That other salmonid species are listed in Table 4.2 suggests that this group is among the most sensitive of aquatic species to cadmium and a variety of salmonids have been found in previous studies near Faro. While the other species in Table 4.2 have not been specifically reported in previous studies, some of the same genera have been observed (e.g., *Hydra, Tanytarsus*). Also, the ubiquitous distribution of other species (e.g., *Chironomus* sp., *Chlorella sp., Selenastrum capricornutum*, the latter now known as *Pseudokirchneriella subcapitata*) does not preclude their presence. Therefore, CWQGs developed to protect sensitive aquatic species against the effects of cadmium cannot be considered overly protective with respect to the species assemblage found, or potentially present, near the Faro Mine complex.

4.2 Zinc

4.2.1 Aquatic Chemistry

Zinc ranks as the 24th most abundant element in the earth's crust, with an average concentration of 70 mg/kg (CCREM 1987). Zinc concentrations in uncontaminated freshwater are usually ≤ 10 ug/L (USEPA 1987). Zinc occurs in many forms in surface waters, including suspended and dissolved forms. Complexes of zinc with the common ligands of surface waters are soluble in neutral and acidic solutions, so it is readily transported and considered one of the most mobile of the heavy metals (USEPA 1987). Aqueous pH is an important factor influencing zinc speciation (Heijerick et al. 2002b, 2003; Wilde et al. 2006). In freshwater with pH around 6.0, the dominant inorganic forms are the free ion (98%) and zinc sulphate (2%), whereas at pH of 9.0, the dominant forms are the monohydroxide ion (78%), zinc carbonate (16%), and the free ion (6%) (Turner et al. 1981). Decreasing alkalinity and pH both favour the free metal ion (CCME 1999), which is

Table 4.2: Vertebrate and invertebrate species sensitive to cadmium at concentrations ≤ 10 µg/L.

Type	Scientific Name	Common Name	Cadmium Toxicity Range (µg/L) [a]		Distribution	Preferred Habitat	Related Species Found Near Faro
Fish	*Oncorhynchus mykiss*	Rainbow trout	0.35 - 27	Hansen et al. 2002, Chapman 1975, 1978	Native to west coast of North America. Introduced throughout North America and worldwide.	Gravelly streams required for spawning. Found in rivers and lakes.	Round whitefish (*Prosopium cylindraceum)* , Arctic grayling (*Thymallus arcticus)* , Lake whitefish (*Coregonus clupeaformis)* , Chinook salmon (*Oncorhynchus mykiss)*
	Salvelinus confluentus	Bull trout	0.83 - 6.06	Hansen et al. 2002, Stratus Consulting 1999	Native of Western North America	Cold, clear headwater lakes and streams	
	Oncorhynchus tshawytscha	Chinook salmon	1.1 - 57	Finlayson and Verrue 1982, Hamilton and Buhl 1990	Native to Northern Pacific Ocean and Tributaries, introduced to Great Lakes and world wide. **Found in Pelly R, Anvil Cr, Vangorda Cr, Blind Cr.**	Spawn in gravelly freshwater rivers and streams, then return to the ocean.	
	Salmo trutta	Brown trout	1.4 - 16.5	Spehar and Carlson 1984a,b, Brown et al. 1994	Native to Europe and western Asia, introduced to North America	Clear, cool, well-oxygenated streams and lakes. Temperatures up to 24^{o}C	
	Salvelinus fontinalis	Brook trout	<1.5 - 5,080	Carroll et al. 1979, Holcombe et al. 1983	Northeastern North America	Clear, cool, well-oxygenated streams and lakes. Temperatures below 20^{o}C	
	Oncorhynchus kisutch	Coho salmon	2.1 - 17.5	Eaton et al. 1978, Chapman 1975	Native to Northern Pacific Ocean, introduced to the Great Lakes.	Spawn in gravelly freshwater streams, then return to the ocean or lakes.	
	Salmo salar	Atlantic salmon	4.5 - 156	Rombough and Garside 1982	North Atlantic Ocean basin	Gravelly streams for spawning and as young. Return to the sea or lake and deeper, cool water.	
	Salvelinus namaycush	Lake trout	7.4	Eaton et al. 1978	Northern North America	Deep, cool lakes, prefer temperatures of about 10 ^{o}C	
	Morone Saxatilis	Striped bass	1 - 10	Hughes 1973, Palawski et al. 1985	North American coastal waters of the Atlantic Ocean	-	-
	Jordanella floridae	Flagfish	4.4 -2,500	Carlson et al. 1982, Spehar 1976a,b	Southeastern North America	Vegetated sloughs, ponds, lakes and sluggish streams	-
	Catostomus commersoni	White sucker	7.1 - 1,110	Eaton et al. 1978, Duncan and Klaverkamp, 1983	Central and Northern North America	Warmer shallow lakes, bays, and tributary rivers	Longnose sucker (*Catostomus catostomus*)
	Micopterus dolomieui	Smallmouth bass	7.4	Eaton et al. 1978	Originally eastern central North America, introduced throughout the US, Europe and Asia.	Rocky and sandy areas of lakes and rivers in moderately shallow water, retreating to greater depth during the summer	-
	Esox lucius	Northern pike	7.4	Eaton et al. 1978	Much of the upper northern hemisphere	Clear, warm, heavily vegetated rivers and bays	-
	Pimephales promelas	Fathead minnow	10 - 73,500	Spehar and Fiandt 1986, Pickering and Henderson 1966	Central North America	Still waters of ponds to flowing waters of streams	-
Invertebrates	*Daphnia magna*	Water flea	0.15 - 360	Chapman et al. Manuscript, Fargasova 1994a	Temperate North America, Europe, Asia, Africa	Small weedy lakes, saline ponds, temporary ponds. Tolerant of low oxygen.	Order Cladocera [b]
	Hyalella azteca	Amphipod	0.98	Ingersoll and Kemble Unpublished	North America: northern Canada to southern Chile	Benthic in ponds are streams, wide thermal tolerance, usually in association with macrophytes or detritus.	-
	Asterionella formosa	Diatom	2	Conway 1978, Rachlin et al. 1982	-	-	-
	Hydra viridissima	Hydra	3	Holdaway et al. 2001	Widespread in freshwaters	Benthic attached, and planktonic larvae	*Hydra* sp.
	Aplexa hypnorum	Snail	3.5 - 93	Holcombe et al. 1984, Phipps and Holcombe 1985	Central and northern Canada	Temporary pools with mud and vegetation	Class Gastropoda, Family Valvatidae
	Tanytarsus dissimilis	Chironomid	3.8	Anderson et al. 1980			*Tanytarsus* sp.
	Daphnia pulex	Water flea	7.5 - 319	Niederlehner 1984, Elnabarawy et al. 1986	Temperate North America	Small ponds with abundant organic matter	Order Cladocera [b]
	Simocephalus serrulatus	Water flea	7 - 24.5	Giesy et al. 1977, Spehar and Carlson 1984a,b	North America: widespread in western Canada	Lakes with macrophytes; acid tolerant	Order Cladocera [b]
	Hydra vulgaris	Hydra	8.5	Holdaway et al. 2001	Widespread in freshwaters	Benthic attached, and planktonic larvae	*Hydra* sp.
	Utterbackia imbecilis	Mussel	9 - 115	Keller and Zam 1991, Keller Unpublished	Central and southern U.S.	Ponds, lakes, and muddy-bottomed pools in rivers and streams	*Pisidium* sp.
	Chironomus tentans	Midge	10	Ingersoll and Kemble Unpublished	North America: widespread	Soft organic substrates in lakes and rivers	Family Chironominae
Plants	(mixed sp.)	Alga	5	Giesy et al. 1979	genera are widespread	-	various green and blue-green algae and diatoms
	Chara vulgaris	Alga	9.5 - 56.2	Heumann 1987			
	Salvina natans	Fern	10	Hutchinson and Czyrska 1972			
	Lemna valdiviana	Duckweed	10	Hutchinson and Czyrska 1972			

[a] As reported by US EPA (2001). References are for low and high ends of data range, respectively.
[b] Cladocera reported at Faro:
Alona sp
Bosmina longirostris
Chydorus gibbus
Daphnia sp
Eurycercus (Bullatifrons) sp

considered to be the form responsible for aquatic toxicity (Heijerick et al. 2002a, De Shamphelaere et al. 2005).

In the presence of dissolved organic materials, much of the dissolved zinc may be in the form of zinc-organic complexes (Lu and Chen 1977; De Schamphelaere et al 2005), but some studies have shown that only a small proportion of zinc was complexed by organic ligands (CCME 1999; De Shampheleare and Janssen 2004). As with inorganic zinc speciation, formation of organo-zinc complexes is probably at least partially pH-dependent. Greater Zn sorption to dissolved organic carbon likely occurs at higher pH and higher concentrations of dissolved organic matter relative to the amount of zinc present (USEPA 1987; CCME 1999; Heijerick et al. 2003; De Shamphelaere et al. 2005).

Thus, zinc speciation is controlled by pH, alkalinity, and the concentrations of various other inorganic and organic ligands (CCME 1999). Most the zinc introduced to aquatic environments is eventually partitioned into sediment by sorption onto hydrous iron and manganese oxides, clay minerals, and organic materials (USEPA 1987).

4.2.2 Mode of Toxicity and Modifying Factors

Zinc is an essential micronutrient for all living organisms, being involved in nucleic acid synthesis and occurring in many enzymes (CCME 1999). Aquatic toxicity of zinc is primarily attributable to waterborne exposure to the free ion (Zn^{2+}) (Heijerick et al. 2002a, De Shamphelaere et al. 2005), although aquatic invertebrates and some fish may also be affected by ingestion of suspended, particulate, or colloidal zinc. Consequently, zinc toxicity is influenced by the factors affecting Zn speciation. Also, although most studies have focused on the toxic effects of water-borne zinc exposures, dietary exposure to zinc may play a significant role in zinc toxicity to aquatic biota. Reproductive effects on *Daphnia magna* were observed after feeding on algae that had been exposed to 28 ug/L Zn, (De Shamphaleare et al. 2004), whereas daphnid toxicity occurred at concentrations of >100 ug/L in water-only tests conducted at the same laboratory (Heijerick et al. 2005b).

At extremely high waterborne levels, zinc causes gross morphological alterations at the teleost gill (Skidmore and Tovell 1972) and fish usually die within a few hours as a result of tissue hypoxia from impaired gas exchange (Lappivaara et al. 1995). At lower waterborne concentrations that more realistically reflect contaminated environments, zinc (Zn^{2+}) disrupts calcium uptake by chloride cells of the gills (Spry and Wood 1985, Heijerick et al. 2002b, Hogstrand et al. 1995, 1996). Calcium and zinc appear to compete for binding sites at the channels that regulate calcium uptake, such that while elevated levels of zinc inhibit calcium uptake, the converse is also true (Santore et al. 2002). Inhibition of calcium uptake leads to hypocalcemia, which may culminate in the death of the fish within a few days, depending on the zinc concentration. Survival of fish is equally or more sensitive than other endpoints in chronic exposures to zinc (De Shampheleare and Janssen 2004). Swim-up alevins or young juveniles appear to be more sensitive to zinc than either younger or older life stages (Buhl and Hamilton 1990).

As with fish, it appears that zinc toxicity to aquatic invertebrates (Heijerick et al. 2005b; Muyssen et al. 2006) and algae (Wilde et al. 2006) may occur through disruption of calcium regulation by chloride cells. However, there may be differences in the specific mechanisms controlling acute versus chronic toxicity, at least for some invertebrates (Pauluskis and Winner 1988; Heijerick et al. 2005b).

The aquatic toxicity of zinc is influenced by a number of chemical factors including pH, hardness and the presence of other inorganic or organic ligands (USEPA 1987, CCME 1999). These affect zinc toxicity either by influencing the speciation of zinc or by influencing the sorption or binding of available zinc to biological tissues (e.g., competition for binding sites at gills).

As noted above (Section 4.2.1), aqueous pH influences zinc speciation, which in turn influences zinc toxicity to aquatic biota. Toxicity test results expressed on the basis of total zinc concentrations often show a pattern of reduced toxicity at basic compared to neutral pH because formation of hydroxides at elevated pH reduce the quantities of available Zn^{2+} ion (Heijerick et al. 2002b, Santore et al. 2002). However, the toxicity of the free ion (Zn^{2+}) is reduced at lower pH as a result of increased competition between H^+ and Zn^{2+} for binding sites on biotic ligands, particularly at pH <7 (Santore et al. 2002; Heijerick et al. 2002a, 2003, 2005b; De Shampheleare and Janssen 2004; Wilde et al. 2006). The effects of pH on zinc toxicity to algae may be quite large, while pH effects on toxicity to invertebrates and fish are modest (Table 4.3). Aqueous pH appears to also affect the number and/or characteristics of binding sites of algae, whereas the binding characteristics for higher organisms is believed to be independent of the test medium characteristics (Heijerick et al. 2002a).

Hardness is probably the strongest single water quality factor influencing acute and chronic zinc toxicity to aquatic biota (Table 4.3 and USEPA 1987), with the calcium component of hardness being particularly effective in protecting fish from zinc toxicity (Santore et al. 2002). Magnesium appears to be as or more effective than calcium in reducing zinc toxicity to invertebrates and algae (Heijerick et al. 2002a; Niyogi and Wood 2004).

Dissolved organic carbon plays a varying role in mitigating zinc toxicity to aquatic biota (Table 4.3), with the magnitude of influence dependent on factors such as the quantity and

Table 4.3: Direction (arrows) and magnitude (in parentheses, if reported) of influence of water quality modifiers on zinc toxicity to aquatic biota. ↓ indicates toxicity was reduced by increase concentrations of modifier, whereas ↑ indicates toxicity was increased.

		Toxicity Modifiers									
Species	**Exposure Duration**	**Ca^{2+}**	**Mg^{2+}**	**Hardness (Ca versus Mg not specified)**	**Na^{+}**	**K^{+}**	**H^{+}**	**DOC**	**Humic Acid**	**Combined influence**	**Source**
Pseudokirchneriella subcapitata	3 d	↓ (1.7)	↓ (6-6.5)		↓ (2.1)	no	↓ (27, Zn^{2+})				Heijerick et al. 2002a
	3 d						√[a]	↓		↓ (20)	De Schamphelaere et al 2005.
Chlorella sp.	3 d						↓ (15, Zn^{2+})				Wilde et al. 2006
Daphnia magna	3 d			↓ (2)					↓ (1.4)	↓ (2.7)	Paulauskis and Winner 1988
	50 d			↓ (7)					↓ (4)	↓ (9)	Paulauskis and Winner 1988
	2 d	↓ (6.3)	↓ (2.1)		↓ (3.1)	no	no				Heijerick et al. 2002b
	21 d			↓↑[b]			↑ (2.2, Zn)	↓ (10)		↓ (30)	Heijerick et al. 2003
	21 d	↓ (2.7)	↓ (2.3)		↓ (1.6)		↓ (2.0, Zn^{2+})				Heijerick et al. 2005
	21 d	↓					√[a]	↓		↓ (6)	De Schamphelaere et al 2005.
Ceriodaphnia dubia	2 d			↓ (2)			↓ (~1.5, Zn)	↓ (<1.5)			Hyne et al. 2005
Rainbow Trout	30 d			↓ (5.4)						5.4	Alsop et al. 1999
	5 d			↓ (5)			↑ (~2, Zn)			↓ (11)	Hansen et al. 2002
				↓ (2)							Davies et al. 1993
	4 d			↓ (30)							Bradley and Sprague 1985
	30 d	↓ (12)	↓ (3)		↓ (>2)		↓ (2.0, Zn^{2+})			↓ (30)	De Schamphelaere and Janssen 2004
	30 d	↓					√[a]	↓		↓ (5)	De Schamphelaere et al 2005.
Bull Trout	5 d			↓ (5)			↑ (~2, Zn)			↓ (13)	Hansen et al. 2002
Mottled Sculpin	30 d			↓ (8.3)							Woodling et al. 2002 and Brinkman and Woodling 2005

[a] The influence of pH on zinc toxicity was tested but the magnitude of effect not explicitly reported.
[b] Toxicity was inceased at both high (>300 mg/L) and low (<100 mg/L) hardness

type of DOC present (e.g., humic versus fulvic acids), aqueous pH (affecting Zn sorption to DOC; Heijerick et al. 2003; De Shamphelaere et al. 2005), and the relative concentrations of zinc and other ions (e.g., calcium and magnesium; Paulauskis and Winner 1988). The protetctive effect of DOC is probably greater in soft than hard water (Paulauskis and Winner 1988, Vignault et al. 2004)

Numerous studies have shown that acclimation of aquatic organisms to low (non-toxic) levels of zinc results in greater tolerance to toxic zinc concentrations (Bradley et al. 1985, Hobson and Birge 1989; Hogstrand and Wood 1995, Anadu et al. 1989, Muyssen et al. 2002, Muyssen and Janssen 2005). This response happens and is lost quickly (i.e., within a few days of zinc exposure) (Bradley et al. 1985; Aanadu et al. 1989). As a result, acclimated organisms may be able to tolerate lethal concentrations up to 5 times higher than unacclimated organisms (Anadu et al. 1989; Muyssen et al. 2002). In some cases, sublethal rather than lethal tolerance may be increased by pre-exposure to zinc (Muyssen and Janssen 2005).

4.2.3 Sensitive Species and Relevance to Faro

The maximum zinc concentration predicted for surface water downstream of the Faro Mine complex after mine closure and remediation is approximately 5,000 ug/L (Table 2.1), which would be acutely toxic to most aquatic biota (USEPA 2007). Toxicity data were tabulated for organisms showing zinc toxicity at concentrations of 100 ug/L or less to identify the species most sensitive to zinc toxicity (Table 4.4). The toxicity data reported by each study (based on variable test conditions among studies) as well as hardness-adjusted values reported by USEPA (1987) are shown, but the latter values were available for only a few studies. Based on the original toxicity data, lowest effect concentrations (greatest toxicity) were reported for brown trout (≥4.9 ug/L), the Indian freshwater perch *Ambassis* (8.1 u/gL), the isopod *Asellus* (10 ug/L), and the water flea *Moina* (10 ug/L), although relative sensitivities may differ if hardness (data not available for all studies) is taken into account (Table 4.4). Overall, the data indicate that a variety of aquatic biota, including plant, invertebrate and fish species are affected by concentrations of zinc below 100 ug/L, at least when water hardness is low (e.g., <50 mg/L). While Chinook salmon is the only species on Table 4.5 reported in previous studies conducted at Faro, several of the genera are known to be present (e.g., *Cottus*, *Oncorhynchus*, *Tanytarsus*, *Daphnia*) (Table 4.5). Based on ubiquitous distribution, other organisms may be present, but have not been previously reported (e.g., *Chlorella, Hyallela*). Therefore, CWQGs developed to protect sensitive aquatic species against the effects of zinc cannot be considered overly protective with respect to the species assemblage found, or potentially present, near the Faro Mine complex.

Table 4.4: Summary of studies reporting zinc toxicity to aquatic biota at concentrations ≤ 100 μg/L.

		Scientific Name	Common Name	Hardness (mg/L as CaCO3)	Effect Concentration[a] (μg/L)	Hardness-Adjusted Values [b] (μg/L)	Reference [c]
US EPA 1987	Acute	*Ceriodaphnia reticulata*	Water flea	45	32	34.99	Carlson and Roush 1985
		Ceriodaphnia reticulata	Water flea	45	41	44.82	Carlson and Roush 1985
		Oncorhynchus mykiss	Rainbow trout (fry)	9.2	66	277	Cusimano et al. 1986
		Daphnia magna	Water flea	45	68	74.35	Mount and Norberg 1984
		Daphnia magna	Water flea	-	<71.95	-	Anderson 1948
		Ceriodaphnia reticulata	Water flea	45	76	83.1	Mount and Norberg 1984
		Oncorhynchus tshawytscha	Chinook salmon (juv)	21	84	175.2	Finlayson and Verrue 1982
		Salmo clarki	Cutthroat trout (fingerling)	-	90	-	Rabe and Sappington 1970
		Oncorhynchus mykiss	Rainbow trout (swim-up alevin)	23	93	179.6	Chapman 1975, 1978b
		Oncorhynchus tshawytscha	Chinook salmon (swim-up alevin)	23	97	187.3	Chapman 1975, 1978b
		Daphnia magna	Water flea	45.3	100	108.9	Biesinger and Cristensen 1972
	Chronic	*Jordanella floridae*	Flagfish	44	36.41	-	Spehar 1976a,b
		Daphnia magna	Water flea	211	46.73	-	Chapman et al. Manuscript
		Daphnia magna	Water flea	104	47.29	-	Chapman et al. Manuscript
	Plants	*Selenastrum capricornutum*	Green alga	-	30	-	Bartlett et al. 1974
		Selenastrum capricornutum	Green alga	-	40.4	-	Greene et al. 1975
		Selenastrum capricornutum	Green alga	-	50.9	-	Turbak et al. 1986
		Selenastrum capricornutum	Green alga	-	68	-	Greene et al. 1975
USEPA Ecotox Database (post US EPA 1987)		*Salmo trutta*	Brown trout	-	4.9 - 19.6	-	Sayer et al 1989
		Ambassis ranga	Indian freshwater perch	-	8.1	-	Gaikwad S.A. 1989
		Asellus aquaticus	Isopod	-	10	-	Migliore and Nicola Giudici 1990
		Moina macrocopa	Water flea	-	10	-	Wong. 1993
		Etroplus maculatus	Pearlspot	-	10.5	-	Gaikwad S.A. 1989
		Daphnia magna	Water flea	-	12.5	-	Paulaskis and Winner 1988
		Tilapia mossambica	Mozambique tilapia	-	12.6	-	Gaikwad S.A. 1989
		Insect community	Insect community	85	15	-	Clements et al. 1988
		Ceriodaphnia dubia	Water flea	-	20	-	Masters et al. 1991
		Moina irrasa	Water flea	-	25	-	Zou 1997
		Pimephales promelas	Fathead minnow	-	29	-	Norberg-King T.J. 1987
		Ephydatia fluviatilis	Freshwater sponge	30	32	-	Francis and Harrison 1988
		Moina irrasa	Water flea	-	33.79	-	Zou and Bu 1994
		Corbicula sp.	Asiatic clam	70.7	34	-	Farris et al. 1989
		Erpobdella octulata	Leech	15	60	-	Willis 1989
		Pimephales promelas	Fathead minnow	-	60	-	Gale et al. 1992
		Ceriodaphnia dubia	Water flea	-	60.5	-	Bitton et al. 1995
		Ceriodaphnia dubia	Water flea	-	65	-	Belanger and Cherry 1990
		Potamopyrgus jenkinsi	Snail	-	71.5	-	Dorgelo et al. 1995
		Colpidium sp.	Protozoan	-	72	-	Pratt et al. 1997
		Hyalella azteca	Scud	-	73	-	Phipps et al. 1995
		Ancylus fluviatilis	Mollusc		80		Willis 1988
		Moina macrocopa	Water flea	-	94	-	Pokethitiyook et al. 1987
		Oncorhynchus mykiss	Rainbow trout	-	95	-	Anadu et al. 1989
Recent Articles		*Oncorhynchus mykiss*	Rainbow trout	30	24-53	-	Hansen et al. 2002
		Daphnia magna	Water flea	-	28	-	DeSchamphelaere et al. 2004
		Salvelinus confluentus	Bull trout	30	30-80	-	Hansen et al. 2002
		Cottus bairdi	Mottled sculpin	46.3	32	-	Woodling et al 2002
		Tanytarsus dissimilis	Chironomid	46.8	36.8	-	Anderson et al. 1980
		Cottus bairdi	Mottled sculpin	48.6	38	-	Woodling et al 2002
		Chlorella sp.	Alga	40-48	52	-	Wilde et al 2006
		Ceriodaphnia dubia	Water flea	125	70	-	Hyne et al 2005

[a] Data represent a variety of endpoints (eg. LC50 or ED50 for acute tests and some chronic tests, as well as highest no-effect concentrations or chronic threshold values for chronic tests).

[b] As reported by USEPA (1987) for hardness of 50 mg/L as $CaCO_3$.

[c] References cited in USEPA 1987, except for recent articles where are listed in Section 7.0.

Table 4.5: Vertebrate and invertebrate species sensitive to zinc at concentrations ≤ 100 µg/L.

Type	Scientific Name	Common Name	Cadmium Toxicity Range (µg/L) [a]		Distribution	Preferred Habitat	Related Species Found Near Faro
Fish	*Salmo trutta*	Brown trout	4.9 - 19.6	Sayer et al. 1989	Native to Europe and western Asia, introduced to North America	Clear, cool, well-oxygenated streams and lakes. Temperatures up to 24°C	Salmonid species [c]
	Ambassis ranga	Indian freshwater perch	8.1	Gaikwad 1989	Asia	-	-
	Etroplus maculatus	Pearlspot	10.5	Gaikwad 1989	Asia, India, Sri Lanka	Lagoons and small streams	-
	Tilapia mossambica	Mozambique tilapia	12.6 - 1,600	Gaikwad 1989, Qureshi and Saksena 1980	Mozambique to South Africa	-	-
	Pimephales promelas	Fathead minnow	29 - 35,500	Norberg-King 1987, Mount 1966	Central North America	Still waters of ponds to flowing waters of streams	-
	Salvelinus confluentus	Bull trout	30-80	Hansen et al. 2002	Western North America	Clean, cool streams and lakes	Salmonid species [c]
	Cottus bairdi	Mottled sculpin	32 - 38	Woodling et al. 2002	Central to Northeastern North America	Cool streams and lakes	Slimy sculpin (*Cottus cognatus*)
	Jordanella floridae	Flagfish	36.4 - 1,500	Spehar 1976	Southeastern North America	Vegetated sloughs, ponds, lakes and sluggish streams	-
	Oncorhynchus mykiss	Rainbow trout	66 - 7,210	Cusimano et al. 1986, Sinley et al. 1974	Native to west coast of North America, Introduced throughout North America and worldwide.	Gravelly streams required for spawning. Found in rivers and lakes.	Salmonid species [c]
	Oncorhynchus tshawytscha	Chinook salmon	84 - 701	Finlayson and Verrue 1982, Chapman 1975, 1978	Native to Northern Pacific Ocean and Tributaries, introduced to Great Lakes and world wide. **Found in Pelly R, Anvil Cr, Vangorda Cr, Blind Cr.**	Spawn in gravelly freshwater rivers and streams, then return to the ocean.	Salmonid species [c]
	Salmo clarki	Cutthroat trout	90	Rabe and Sappington 1970	Central west coast of North America and inland varity in BC and Alberta	Gravelly lowland, coastal streams and lakes, inland alpine lakes and rivers, estuaries and near shore in the sea.	Salmonid species [c]
Invertebrates	*Moina macrocopa*	Cladoceran	10	Wong 1993	Asia	-	Order Cladocera [b]
	Asellus aquaticus	Isopod	10	Migliore and Nicola Giudici 1990	Temperate, world wide	Eutrophic streams and lakes, depositional areas.	-
	Daphnia magna	Cladoceran	12.5 - 798.9	Paulaskis and Winner 1988, Attar and Maly 1982	Temperate North America, Europe, Asia, Africa	Small weedy lakes, saline ponds, temporary ponds. Tolerant of low oxygen.	Order Cladocera [b]
	Ceriodaphnia dubia	Cladoceran	20 - 174.1	Masters et al. 1991, Carlson et al. 1986	Palearctic and North America	Nearshore of rocky lakes and ponds	Order Cladocera [b]
	Moina irrasa	Cladoceran	25	Zou 1997	Asia		Order Cladocera [b]
	Ephydatia fluviatilis	Freshwater sponge	32	Francis and Harrison 1988	Southern Canada, Europe	Benthic in ponds and streams, on hard substrates.	-
	Tanytarsus dissimilis	Chironomid	36.8	Anderson et al. 1980	-	-	*Tanytarsus* sp.
	Ceriodaphnia reticulata	Cladoceran	32 - 76	Carlson and Roush 1985, Mount and Norberge 1984	North America; Europe	Nearshore of lakes and ponds, among vegetation, ph 5.3 to 8.7	Order Cladocera [b]
	Corbicula sp.	Asiatic clam	34 - 6,040	Faris et al. 1989, Cherry et al. 1980	Asia; North America (introduced: US states)	Mud, sand substrates in lakes and rivers	-
	Erpobdella octulata	Leech	60	Willis 1989	Europe, Middle East	Benthic predators of molluscs	-
	Potamopyrgus jenkinsi	Snail	71.5	Dorgelo et al. 1995	Europe	Benthic algal grazer in streams	*Probythinella lacustris,* Class Gastropoda, Family Valvatidae
	Hyalella azteca	Amphipod	73	Phipps et al. 1995	North America: northern Canada to southern Chile	Benthic in ponds are streams, wide thermal tolerance, usually in association with macrophytes or detritus.	-
	Ancylus fluviatilis	Mollusc	80	Willis 1988	Europe	Running water	*Pisidium* sp.
Plants & Protazoa	*Selenastrum capricornutum* (= *Pseudokirchneriella subcapitata*)	Green alga	30 - 68	Bartlett et al. 1974, USEPA 1980	genera are widespread	-	Various green and blue-green algae and diatoms
	Chlorella sp.	Alga	52	Wilde et al. 2006			
	Colpidium sp.	Protozoan	72	Pratt et al. 1997			-

[a] Range identified from US EPA, 1987, Ecotox and recent literature. References for low and high ends of range respectively.

[b] Cladocera reported at Faro:
- *Alona* sp
- *Bosmina longirostris*
- *Chydorus gibbus*
- *Daphnia* sp
- *Eurycercus (Bullatifrons)* sp

[c] Salmonids reported at Faro:
- Round whitefish (*Prosopium cylindraceum*)
- Arctic grayling (*Thymallus arcticus*)
- Lake whitefish (*Coregonus clupeaformis*)
- Chinook salmon (*Oncorhynchus mykiss*)

4.3 Copper

4.3.1 Aquatic Chemistry

Copper ranks just behind zinc (25th) in relative abundance in the earth's crust and is usually found in the form of mineral ores at concentrations of <2% (CCREM 1987). It is a minor nutrient for both plants and animals, but becomes toxic to aquatic life at elevated concentrations. Natural background concentrations of copper in surface waters are usually less than 30 ug/L (USEPA 1985, 2007; CCREM 1987). Copper occurs in four oxidation states, 0, +1, +2, and +3, of which the most common aqueous form is +2 (cupric) (CCREM 1987, USEPA 1985, 2007). In well-aerated surface waters, copper may be present as free cupric ions or complexed with inorganic or organic ligands, depending on pH and the relative abundace of other ligands (CCREM 1987). Copper is generally more soluble in acidic waters and increasingly forms carbonate or hydroxide colloids, or precipitates, as pH and alkalinity increase (CCREM 1987, Janssen et al. 2003). Copper also has high affinity for manganese oxides, clays, and organic matter (CCREM 1987). As a result of its high reactivity, the free ionic form usually represents a small fraction of the total copper present in natural water (USEPA 1985, 2007; CCREM 1987, Scheinberg 1991).

4.3.2 Mode of Toxicity and Modifying Factors

The toxicity of copper is related primarily to the activity of the free cupric (Cu^{2+}) ion (USEPA 1985, Paquin et al. 2002, Boeckman and Bidwell 2006), although other ion species may be weakly toxic (e.g., $CuOH^+$ and $CuCO_3$; Erickson et al. 1996, De Schamphelaere and Janssen 2002, De Schamphelaere et al. 2002, 2006). The binding affinities of $CuOH^+$ and $CuCO_3$ for biotic ligands are approximately 4- to 10-fold lower relative to Cu^{2+} (Niyogi and Wood 2004), consistent with the relative toxicity of these forms.

As with a number of other metals, copper tends to bind to gills at very specific sites and interferes with the ability of the organism to regulate both ion uptake and efflux across the gill (Paquin et al. 2002). Although copper predominantly exists as a divalent metal (Cu^{2+}) in surface waters, it exhibits several potential pathways of uptake and transport (Kamunde et al. 2005) and ultimately inhibits (monovalent) Na^+ transport (Grosell et al. 2002, Paquin et al. 2002, Niyogi and Wood 2004). Sodium uptake from the water across the gills is essential for any water breathing freshwater animal as it serves to compensate for the diffusive loss of sodium from its concentrated extracellular fluids to the surrounding dilute environment (Grosell et al. 2002). Based on larger surface area relative to volume, sodium turnover (the rate at which sodium is taken up and lost) is greater among smaller than larger organisms and may account for generally greater sensitivity of aquatic invertebrates to copper toxicity relative to fish (Grosell et al. 2002). Pronounced sodium deficiency leads to cardiovascular collapse and death (Wilson and Taylor 1993).

The bioavailability of copper is known to be influenced by the concentrations of competing cations, by proton concentrations (pH), by dissolved ligands (such as dissolved organic carbon and inorganic carbonates), and by colloidal and particulate ligands (such as amorphous iron oxides and particulate organic carbon) (DeMayo and Taylor 1981). Although the interactions are complex, it has often been observed that the bioavailability of copper is particularly influenced by organic matter (e.g., dissolved and total organic carbon; Hering and Morel 1990; Allan 2002). In general, copper forms more stable complexes with natural organic compounds than does Cd, Pb, or Zn (Hart and Davies 1977). Organic matter has negatively charged surfaces and copper tends to associate by adsorption or covalent bonding (e.g., Davis 1984). The formation of organo-copper complexes decreases the bioavailable proportion the total copper present (De Schamphelaere et al. 2006).

Higher DOC (range 5-15 mg/L) and higher pH (range 6-7.8) reduced sublethal copper toxicity to rotifers by up to 12 times over the range of conditions tested (De Schamphelaere et al. 2006). Dissolved copper (≤0.45 um) effects on algae varied by up to 27 times in waters with different pH and DOC (De Schamphelaere and Janssen 2006). The toxicity of total copper to *Ceriodaphnia dubia* was reduced 45-fold by increasing DOC (fulvic acid) from 0.1 mg/L to 10 mg/L, whereas increased pH (5.5 to 8.4) reduced toxicity by only 5-fold (Hyne et al. 2005). Welsh et al. (1993) showed that copper toxicity to fathead minnows decreased by an order of magnitude when DOC concentrations increased by an order of magnitude. The biotic ligand model used by USEPA (2007) to update the water quality criterion for copper assumes that acute toxicity of total copper to aquatic biota will be reduced by about 8- to 9-fold over the range of 2-16 mg/L DOC.

Increased pH generally reduces the toxicity of total copper on aquatic biota (Schubauer-Berigan et al. 1993, De Schamphelaere et al. 2006). This is because increasing proportions of the total copper speciates to $CuOH^+$ and $CuCO_3$ relative to Cu^{2+} (De Schamphelaere et al. 2006). The biotic ligand model used by USEPA (2007) to update the water quality criterion for copper assumes that acute toxicity of total copper to aquatic biota will be reduced by about 8- to 9-fold over the range of pH 6.5 to 8.0. However, the toxicity of the Cu^{2+} ion increases with increasing pH, due to reduced competition by protons (H^+) for binding sites at Na^+ transport sites on cell surfaces (Borgmann et al. 2005, De Shamphelaere and Janssen 2006, De Shamphelaere et al. 2007). As might be expected based on copper's mechanism of effect, increased levels of aqueous sodium may also result in a small to moderate reduction in copper toxicity to aquatic biota (Erickson et al. 1996, DeSchamphelaers and Janssen 2002, Borgmann et al. 2005, DeSchamphelaere et al. 2007).

Studies investigating the effect of hardness or its component cations (Ca^{2+} and Mg^{2+}) on copper toxicity have produced equivocal results, in part because factors such as pH and alkalinity often co-vary with hardness (Gensemer et al. 2002, USEPA 2007). Generally, hardness has limited (typically ≤2-fold) or no protective effect on copper toxicity to biota[1], particularly when other factors are held constant (Welsh et al. 1993, Barata et al. 1998, De Shamphelaere and Janssen 2002, 2004, Gensemer et al. 2002, De Shamphelaere et al. 2003, Taylor et al. 2003, Sciera et al. 2004, De Shamphelaere and Janssen 2004, Hyne et al. 2005, Boeckman and Bidwell 2006, De Shamphelaere and Janssen 2006, USEPA 2007).

Fish appear to be more sensitive to copper after yolk-sac absorption compared to embryonic or newly hatched stages (Chapman 1978, Buhl and Hamilton 1990, Stasiunaite 2005). Fish appear to accumulate copper primarily in liver, with lesser amounts in other organs and tissues (Kamunde et al. 2003, 2005), although gill accumulations may vary considerably among species (Taylor et al. 2003).

4.3.3 Sensitive Species and Relevance to Faro

The maximum copper concentrations predicted to occur in surface waters downstream of the Faro Mine complex after closure and remediation were approximately 133 ug/L (Table 2.1), so data were tabulated for organisms showing toxicity at concentrations of 150 ug/L or less (Table 4.6). Fish and invertebrate data were normalized by USEPA (2007) using a biotic ligand model (BLM) to account for differences in test conditions among studies. Copper concentrations in that range (≤150 ug/L) are toxic to a wide variety of fish, invertebrate and plant species (Table 4.6). Lowest effect concentrations (greatest toxicity) were reported for the green alga *Chlorella pyrenoidosa* (1 ug/L), several cladoceran species (≥2.7 ug/L), the rotifer *Brachiorus* (3.5 ug/L), the diatom *Nitzschia* (5 ug/L), Chinook salmon (5.9 ug/L), the snails *Lithoglyphus virens* (6.7 ug/L) and *Campeloma* (9.8 ug/L), the caddisfly Clistoronia (7.7 ug/L), fathead minnow (9.4 ug/L), and the amphipod *Gammarus* (9.6 ug/L).

Some of the aquatic biota that are sensitive to copper are not found in Yukon rivers (Table 4.7a,b). Of all the species which appear to be very sensitive to copper, Chinook salmon and northern pike are the only ones previously reported in waters near the Faro mine complex

[1] Although water quality criteria for copper developed in Canada (CCREM 1987) and the United States (USEPA 1985) in the 1980s took water hardness into account, more recent data suggest that the supposed hardness relationship may have been attributable to other co-varying factors as pH and alkalinity (Gensemer et al. 2002, USEPA 2007).

Table 4.6: Summary of studies reporting copper toxicity to aquatic biota at concentrations ≤ 150 µg/L.

Source	Type	Scientific Name	Common Name	Hardness (mg/L as CaCO3)	BLM Normalized Species Mean Effect Concentration [a] (ug/L)	n	Reference [b]
US EPA 2007	Acute	*Daphnia pulicaria*	Cladoceran	-	2.73	24	
		Ceriodaphnia dubia	Cladoceran	-	5.93	24	
		Daphnia magna	Cladoceran	-	6	31	
		Lithoglyphus virens	Snail	-	6.67	1	
		Gammarus	Amphipod	-	9.6	2	
		Scapholeberis sp.	Cladoceran	-	9.73	1	
		Actinonaias	Freshwater mussel	-	11.33	2	
		Hyalella azteca	Amphipod	-	12.07	7	
		Juga plicifera	Snail	-	12.31	2	
		Ptychocheilus oregoneus	Northern squawfish	-	14.61	2	
		Physa integra	Snail	-	20.41	2	
		Etheostoma rubrum	Fountain darter	-	22.74	1	
		Oncorhynchus kisutch	Coho salmon	-	22.93	7	
		Oncorhynchus mykiss	Rainbow trout	-	22.19	37	
		Oncorhynchus tshawytscha	Chinook salmon	-	25.02	12	
		Oncorhynchus	Apache trout	-	32.54	1	Sources cited in US EPA 2007
		Oncorhynchus clarki	Cutthroat trout	-	32.97	11	
		Oncorhynchus gorbu	Pink salmon	-	40.13	3	
		Bufo boreas	Boreal toad	-	47.49	1	
		Lumbriculus variega	Worm	-	48.41	3	
		Utterbackia imbecillis	Freshwater mussel	-	52.51	8	
		Oncorhynchs merka	Sockeye salmon	-	54.82	10	
		Poeciliposis	Gila topminnow	-	56.15	1	
		Gila elegans	Bonytail chub	-	63.22	1	
		Salvelinus confluentus	Bull trout	-	68.31	5	
		Scaphirhynchus	Shovelnose sturgeon	-	69.63	1	
		Pimephales promelas	Fathead minnow	-	69.63	150	
		Xyrauchen texanus	Razorback sucker	-	78.66	2	
		Etheostoma	Greenthroat darter	-	82.8	1	
		Etheostoma flabella	Fantail darter	-	124.3	4	
		Ptychocheilus oregoneus	Northern squawfish	-	132.2	2	
	Chronic	*Brachiorus calyciflorus*	Rotifer	-	3.54	1	
		Daphnia pulex	Cladoceran	-	5.68	3	
		Oncorhynchus tshawytscha	Chinook salmon	-	5.92	1	
		Clistoronia magnifica	Caddisfly	-	7.67	1	
		Pimephales promelas	Fathead minnow	-	9.38	1	
		Campeloma decisum	Snail	-	9.77	2	
		Salvelinus fontinalis	Brook trout	-	12.5	2	
		Daphnia magna	Cladoceran	-	14.1	5	Sources cited in US EPA 2007
		Pimephales notatus	Bluntnose minnow	-	18	1	
		Ceriodaphnia dubia	Cladoceran	-	19.3	5	
		Catostomus commersoni	White sucker	-	20.9	1	
		Oncorhynchus mykiss	Rainbow trout	-	23.8	2	
		Lepomis macrochirus	Bluegill	-	27.2	1	
		Salmo trutta	Brown trout	-	29.9	1	
		Salvelinus namaycush	Lake trout	-	30.9	1	
		Esox lucius	Northern pike	-	60.4	1	
	Plants				**Effect Concentration**		
		Chlorella pyrenoidosa	Green alga	-	1	-	Steeman-Nielsen and Wium-Andersen 1970
		Mixed culture	Algae	-	5	-	Elder and Horne 1978
		Nitzschia palea	Diatom	-	5	-	Steeman-Nielsen and Wium-Andersen 1970
		Chlamydomonas reinhardtii	Green alga	90-133	12.2-43.0	-	Winner and Owen 1991a
		Chlamydomonas reinhardtii	Green alga	90-133	12.2-49.1	-	Winner and Owen 1991a
		Lemna minor	Duckweed	39	24	-	Taraldsen and Norberg-King 1990
		Microcystis aeruginosa	Blue-green alga	54.9	30	-	Bringmann 1975, Bringmann and Kuhn 1976, 1978a,b
		Chlamydomonas reinhardtii	Green alga	24	31.5	-	Schafer et al. 1993
		Selenastrum capricornutum	Green alga	16	38	-	Chen et al. 1997
		Chlorella vulgaris	Green alga	-	42	-	Rosko and Rachlin 1977
		Selenastrum capricornutum	Green alga	9.3	44.3	-	Blaise et al. 1986
		Selenastrum capricornutum	Green alga	9.3	46.4	-	Blaise et al. 1986
		Selenastrum capricornutum	Green alga	24.2	48.2	-	Radetski et al. 1995
		Selenastrum capricornutum	Green alga	9.3	48.4	-	Blaise et al. 1986
		Selenastrum capricornutum	Green alga	14.9	50	-	Bartlett et al. 1974
		Selenastrum capricornutum	Green alga	15	53.7	-	Turbak et al 1986
		Selenastrum capricornutum	Green alga	24.2	54.4	-	Radetski et al. 1995
		Selenastrum capricornutum	Green alga	14.9	58	-	Nyholm 1990
		Chlorella vulgaris	Green alga	-	62	-	Ferard et al. 1983
		Anabaena strain 7120	Blue-green alga	-	64	-	Laube et al. 1980
		Selenastrum capricornutum	Green alga	9.3	65.7	-	St. Laurent et al. 1992
		Selenastrum capricornutum	Green alga	9.3	69.9	-	St. Laurent et al. 1992
		Chlorella pyrenoidosa	Green alga	3.65-365	78-100	-	Bednarz and Warkowska-Dratnal 1985
		Selenastrum capricornutum	Green alga	14.9	85	-	Christensen et al. 1979
		Anabaena variabilis	Blue-green alga	65.2	100	-	Young and Lisk 1972
		Chrococcus paris	Blue-green alga	54.7	100	-	Les and Walker 1984
		Chlorella pyrenoidosa	Green alga	54.7	100	-	Steeman-Nielsen and Wium-Andersen 1970
		Chlorella vulgaris	Green alga	17.1	100	-	Bilgrami and Kumar 1997
		Lemna minor	Duckweed	-	119	-	Walbridge 1977
		Lemna minor	Duckweed	-	130	-	Brown and Rattigan 1979

[a] BLM values normalized based on chemistry:

Temp	20 °C
pH	7.5
Diss Cu	1.0 mg/L
DOC	0.5 mg/L
%HA	10
Ca	14.0 mg/L
Mg	12.1 mg/L

[b] References listed in USEPA 2007.

Table 4.7a: Vertebrate and invertebrate species sensitive to copper at concentrations ≤ 150 μg/L.

Type	Scientific Name	Common Name	Copper Toxicity Range (μg/L)[a]		Distribution	Preferred Habitat	Related Species Found Near Faro
Invertebrates	*Daphnia pulicaria*	Cladoceran	0.88 - 8.81	Lind et al. Manuscript 1978	Northern hemisphere, widespread in large and small lakes	Lake planktonic	Order Cladocera [c]
	Daphnia magna	Cladoceran	1.22 - 33.58	Long's MS Thesis, Baird et al. 1991	Temperate North America, Europe, Asia, Africa	Small weedy lakes, saline ponds, temporary ponds. Tolerant of low oxygen.	Order Cladocera [c]
	Ceriodaphnia dubia	Cladoceran	2.46 - 34.6	Belanger et al 1989, Oris et al. 1991	Palearctic and North America	Nearshore of rocky lakes and ponds	Order Cladocera [c]
	Daphnia pulex	Cladoceran	2.83 - 12.25	Winner et al 1985	Temperate North America	Small ponds with abundant organic matter	Order Cladocera [c]
	Brachiorus calyciflorus	Rotifer	3.54	Janssen et al. 1994	-	-	-
	Lithoglyphus virens	Snail	6.67	Nebeker et al. 1986b	Western U.S. and southwestern Canada	Larger lakes and rivers, on rocky bottoms and among vegetation	Class Gastropoda, Family Valvatidae
	Clistoronia magnifica	Caddisfly	7.67	Nebeker 1984b	Western North America	Lentic littoral in sediments and detritus	Order Trichoptera [d]
	Hyalella azteca	Amphipod	8.09 - 18.80	Welsh 1996	North America: northern Canada to southern Chile	Benthic in ponds are streams, wide thermal tolerance, usually in association with macrophytes or detritus.	-
	Campeloma decisum	Snail	8.73 - 4319	Arthur and Leonard 1970	Eastern and central North America	Lakes and slow flowing rivers on mud and sand bottom	Class Gastropoda, Family Valvatidae
	Gammarus	Amphipod	8.86 - 10.39	Arthur and Leonard 1970	At genus level, widespread	Freshwater and saltwater habitats	-
	Scapholeberis sp.	Cladoceran	9.73	Carlson et al. 1986	Europe, Asia, Canada (B.C.)	Small lakes and ponds, limnetic-littoral	Order Cladocera [c]
	Actinonaias	Freshwater mussel	10.36 - 12.39	Keller unpublished	Rare in Canada: southwestern Ontario	Larger rivers, in sand and gravel	*Pisidium* sp.
	Juga plicifera	Snail	12.31	Nebeker et al. 1986b	western U.S. and southwestern Canada	Small lakes and slow streams on mud bottoms	Class Gastropoda, Family Valvatidae
	Physa integra	Snail	19.09 - 21.81 -	Arthur and Leonard 1970	Great Lakes-St. Lawrence drainage of North America	Lake margins, on mud substrates	Class Gastropoda, Family Valvatidae
	Utterbackia imbecillis	Freshwater mussel	24.12 - 177.9	Keller unpublished, Keller and Zam 1991	Central and southern U.S.	Ponds, lakes, and muddy-bottomed pools in rivers and streams	*Pisidium* sp.
	Lumbriculus variega	Worm	37.81 - 55.39	Schubauer-Berigan et al. 1993	Europe	Organic sediments	-
Fish	*Oncorhynchus mykiss*	Rainbow trout	5.02 - 99.97	Welsh et al. 2000, Chakoumakos et al. 1979	Native to west coast of North America, Introduced throughout North America and worldwide.	Gravelly streams required for spawning. Found in rivers and lakes.	Salmonid species [b]
	Oncorhynchus tshawytscha	Chinook salmon	5.92 - 48.56	Chapman 1975, 1982, Welsh et al. 2000	Native to Northern Pacific Ocean and Tributaries, introduced to Great Lakes and world wide. **Found in Pelly R, Anvil Cr, Vangorda Cr, Blind Cr.**	Spawn in gravelly freshwater rivers and streams, then return to the ocean.	Salmonid species [b]
	Pimephales promelas	Fathead minnow	5.92 - 266.3	Welsh et al. 1993, Birge et al. 1983	Central North America	Still waters of ponds to flowing waters of streams	-
	Oncorhynchus clarki	Cutthroat trout	10.6 - 111.3	Chakoumakos et al. 1979	Central west coast of North America and inland varity in BC and Alberta	Gravelly lowland, coastal streams and lakes, inland alpine lakes and rivers, estuaries and near shore in the se.	Salmonid species [b]
	Oncorhynchus kisutch	Coho salmon	11.95 - 106.09	Mudge et al. 1993, Buckley 1983	Native to Northern Pacific Ocean, introduced to the Great Lakes.	Spawn in gravelly freshwater streams, then return to the ocean or lakes.	Salmonid species [b]
	Salvelinus fontinalis	Brook trout	12.5	Sauter et al. 1976	Northeastern North America	Clear, cool, well-oxygenated streams and lakes. Temperatures below 20°C	Salmonid species [b]
	Ptychocheilus oregonensis	Northern squawfish	12.54 - 17.02	Andros and Garton 1980	Central west coast of North America	Lakes, pools and rivers	-
	Pimephales notatus	Bluntnose minnow	18	Horning and Neiheisel 1979	Southeastern North America	Sandy and gravelly areas of lakes, ponds and streams	-
	Oncorhynchus gorbu	Pink salmon	19.70 - 78.76	Servizi and Martens 1978	Pacific and Arctic Oceans and tributaries	Spawn in streams and return to the ocean	Salmonid species [b]
	Catostomus commersoni	White sucker	20.9	McKim et al. 1978	Central and Northern North America	Warmer shallow lakes, bays, and tributary rivers	Longnose sucker (*Catostomus catostomus)*
	Etheostoma rubrum	Fountain darter	22.74	Dwyer et al. 1999	Southwestern Mississipi	Riffles of clear creeks and rivers, with moderate to swift current	-
	Oncorhynchs merka	Sockeye salmon	23.74 - 114.4	Servizi and Martens 1978	Northern Pacific Ocean and inland lakes along the west coast of North America	Open water of lakes, dependent on temperature	Salmonid species [b]
	Lepomis macrochirus	Bluegill	27.2 - 4200	Benoit 1975	Southeastern North America	Shallow, weedy, warm areas of lakes, ponds and rivers	-
	Salmo trutta	Brown trout	29.9	McKim et al. 1978	Native to Europe and western Asia, introduced to North America	Clear, cool, well-oxygenated streams and lakes. Temperatures up to 24°C	Salmonid species [b]
	Salvelinus namaycush	Lake trout	30.9	McKim et al. 1978	Northern North America	Deep, cool lakes, prefer temperatures of about 10 °C	Salmonid species [b]
	Oncorhynchus	Apache trout	32.54	Dwyer et al. 1995	-	-	Salmonid species [b]
	Poeciliposis	Gila topminnow	56.15	Dwyer et al. 1999	-	-	-
	Salvelinus confluentus	Bull trout	63.62 - 74.18	Hansen et al. 2000	Native of Western North America	Cold, clear headwater lakes and streams	Salmonid species [b]
	Esox lucius	Northern pike	60.4	McKim et al. 1978	Much of the upper northern hemisphere. **Found in Vangorda Cr, Pelly R**	Clear, warm, heavily vegetated rivers and bays	Northern pike (*Esox lucius*)
	Gila elegans	Bonytail chub	63.22	Dwyer et al. 1995	Southwestern United States	Pools and eddies of warm, often heavily silted, swift moving rivers.	-
	Xyrauchen texanus	Razorback sucker	63.78 - 97.0	Dwyer et al. 1995	Southwestern United States	Moderate depths of rivers with flow less than 0.5 m/s	-
	Scaphirhynchus	Shovelnose sturgeon	69.63	Dwyer et al. 1995	Southeastern North America	Bottom dweller, prefering high turbidity and large waters	-
	Etheostoma lepidum	Greenthroat darter	82.8	Dwyer et al. 1999	Southwestern United States	Gravel riffles of creeks and small rivers	-
	Ptychocheilus oregonensis	Northern squawfish	88.4 - 197.6	Dwyer et al. 1995	Central west coast of North America	Lakes, pools and rivers	-
	Etheostoma flabella	Fantail darter	117.7 - 136.6	Lydy and Wissing 1988	Central eastern North America	Gravel and boulder bottom creeks with slow to moderate flow.	-
Other	*Bufo boreas*	Boreal toad	47.49	Dwyer et al. 1999	Western North America	Near springs, streams, meadows and woodlands.	-

[a] As reported by US EPA (2007). References are for low and high ends of data range, respectively.

[b] Salmonids reported at Faro:
- Round whitefish (*Prosopium cylindraceum*)
- Arctic grayling (*Thymallus arcticus*)
- Lake whitefish (*Coregonus clupeaformis*)
- Chinook salmon (*Oncorhynchus mykiss*)

[c] Cladocera reported at Faro:
- *Alona* sp
- *Bosmina longirostris*
- *Chydorus gibbus*
- *Daphnia* sp
- *Eurycercus (Bullatifrons)* sp

[d] Tricoptera reported at Faro:
- Family *Hydropsychidae*
- Family *Glossosomatidae*
- Family *Brachycentridae*
- Family *Limnephilidae*
- Family *Hydroptilidae*
- Family *Rhyacophilidae*

Table 4.7b: Plant species sensitive to copper at concentrations ≤ 150 µg/L.

Scientific Name	Common Name	Copper Toxicity Range (µg/L)[a]		Distribution	Related Species Found Near Faro
Chlorella pyrenoidosa	Green alga	1 - 100	Steeman-Nielsen and Wium-Andersen 1970	genera are generally widespread	-
Mixed culture	Algae	5	Elder and Horne 1978		-
Nitzschia palea	Diatom	5	Steeman-Nielsen and Wium-Andersen 1970		-
Chlamydomonas reinhardtii	Green alga	12.2 - 49.1	Winner and Owen 1991a, Schafer et al. 1993		*Chlamydomonas sp.*
Lemna minor	Duckweed	24 - 1100	Taraldsen and Norberg-King 1990, Wang 1986		-
Microcystis aeruginosa	Blue-green alga	30	Bringmann 1975, Bringmann and Kuhn 1976, 1978a,b		-
Selenastrum capricornutum (= Pseudokirchneriella subcapitata)	Green alga	38 - 400	Chen et al. 1997, Blaylock et al. 1985		-
Chlorella vulgaris	Green alga	42 - 200	Rosko and Rachlin 1977, Young and Lisk 1972		-
Anabaena strain 7120	Blue-green alga	64	Laube et al. 1980		*Anabaena sp.*
Chrococcus paris	Blue-green alga	100	Les and Walker 1984		-
Anabaena variabilis	Blue-green alga	100	Young and Lisk 1972		*Anabaena sp.*

[a] As reported by US EPA (2007). References are for low and high ends of data range, respectively.

(Table 4.7a,b). While the other species in Table 4.7 have not been specifically reported in previous studies conducted at Faro, some of the same genera have been observed (e.g., *Daphnia*, *Catostomus*, *Oncorhynchus*, *Hydra, Tanytarsus*). Also, the ubiquitous distribution of some other genera (e.g., *Hyallela, Gammarus*, *Chlorella*) does not preclude their presence. Therefore, CWQGs developed to protect sensitive aquatic species against the effects of copper cannot be considered overly protective with respect to the species assemblage found, or potentially present, near Faro.

4.4 Iron

4.4.1 Aquatic Chemistry

Iron is an important component of the earth's crust, ranking 4^{th} in relative elemental abundance. Concentrations of total iron found in oxygenated surface waters at neutral or alkaline pH generally range from 50 to 200 ug/L, with higher levels typically observed in water bodies heavily stained with dissolved humic compounds and acidic bog waters (Wetzel 2001). For pristine waterbodies in Quebec, historical 5^{th} and 95^{th} percentile total iron concentrations were 50 and 1,000 ug/L, respectively (DSEE 2000).

In surface waters, iron exists primarily in ferrous (Fe^{2+}) and ferric (Fe^{3+}) oxidative states. The physico-chemical processes that determine the predominant state of iron in a waterbody are complex, with several abiotic and biotic factors potentially influencing the relative concentrations of each form (Loeffelman et al. 1985; Davison 1993). Variables of principal importance to iron speciation include pH and reduction-oxidation (redox) potential (E_h), which in turn may be influenced by dissolved oxygen and carbon dioxide concentrations, temperature, and sulphur species (Loeffelman et al. 1985).

In general, ferrous iron forms tend to be more soluble than ferric iron forms. In well oxygenated waters at pH ranging from approximately 5.0 to 9.0, ferric (Fe^{3+}) iron is the most stable oxidative state. Under these conditions, hydrated ferric hydroxide ($Fe[OH]_3$) and/or oxyhydroxide (FeOOH), often referred to as hydrous iron oxides, are the predominant iron forms (Loeffelman et al. 1985; Davison 1993; Horowitz 1991). Ferric (oxy)hydroxide in surface waters commonly occurs as a relatively insoluble (< 1 pg/L at pH 7.0) ochre-coloured, amorphous precipitate that may remain in suspension or settle from solution as a loose floc (Wetzel 2001). Ferric (oxy)hydroxide may react strongly with different co-existing organic and inorganic matter to form complex colloidal and suspended particles (Hiraide et al 1988). The intense yellow-brown colouration of certain waters may, in part, be the result of iron-organic complexes (Wetzel 2001). Because colloidal particles of ferric (oxy)hydroxides can be relatively large and contain both positive and negative (i.e., hydroxyl) charges, various charged ions in solution, clay particles and organic colloids may be electro-statically attracted resulting in formation of larger aggregates that can rapidly settle. In this manner, phosphate and other metals, such as arsenic, cobalt and copper, lead, and zinc can often be adsorbed by and co-precipitated with this ferric (oxy)hydroxide precipitate, effectively removing them from solution. Although all particulate iron material can potentially settle to sediments, clay mineral fractions generally sink more rapidly than amorphous ferric (oxy)hydroxide and/or organic complexes (Davison 1993), with the rate of precipitation positively related to the concentration of iron, pH, presence of suspended particles and water temperature (Dave 1984).

4.4.2 Mode of Toxicity and Modifying Factors

Iron is an essential micronutrient for most aquatic organisms, its primary physiological functions involving electron transport in redox systems of respiration and photosynthesis, oxygen transport and/or storage, enzyme activation and as an oxygen carrier during nitrogen fixation (Guerinot and Yi 1994; Wetzel 2001). Adverse effects of aqueous iron to freshwater biota can be both direct (toxic) and indirect (physical). Direct toxic effects are generally attributed to ferrous (Fe^{2+}) iron through aqueous exposure (although conditions rarely support high enough levels of bioavailable Fe^{2+}) and/or through diet. Free Fe^{2+} ions are believed to produce reactive hydroxyl radicals in tissues which can result in lipid peroxidation of organelle membranes, mitochondrial dysfunctions, alterations in intracellular Ca^{2+}-homeostasis and DNA structure, reduced enzymatic activity and suppressed immune functions (van Anholt et al. 2002; Lappivaara et al. 1999, Payne et al. 2001; Baker et al. 1997; Lappivaara and Marttinen 2005). Overall, iron is generally not considered highly toxic to aquatic life in neutral pH, well oxygenated aquatic environments, mainly because the Fe^{2+} form does not usually occur at high enough concentrations under such conditions (Loeffelman et al. 1985; Linton et al. 2007).

Indirect physical effects may be associated with ferric (Fe^{3+}) iron and in particular, its tendency to precipitate as ferric (oxy)hydroxide and/or humic complexes (i.e., particles). Ferric iron and its complexes can precipitate on aquatic organism gill and egg membranes, physically clogging gills and chorion pores, respectively, and damaging gill epithelia effectively interfering with gas exchange and reducing oxygen uptake which in turn can delay hatching and/or cause mortality (Smith et al. 1973; Grobler et al. 1989; Wepener et al. 1992; Dalzell and Macfarlane 1999; Lappivaara et al. 1999; van Anholt et al. 2002; Peuranen et al. 2003). In addition, precipitation of ferric (oxy)hydroxides and iron-humic complexes can constrain food access by invertebrates (Gerhardt 1992; Randall et al. 1999) and alter the quality and structure of the benthic habitats (Loeffelman et al. 1985; Rasmussen and Lindegaard 1988). Overall, indirect physical effects due to iron precipitation are considered more ecologically detrimental than direct toxic effects in many aquatic environments, but especially in those also impacted by low pH (McNight and Feder 1984; Linton et al. 2007).

Direct toxicity and indirect adverse physical impacts associated with iron in aquatic environments can be influenced by a number of physico-chemical factors that are also important determinants of iron speciation. Because pH and E_h largely influence whether iron occurs in ferrous or ferric form, these factors are primary influences determining whether potential biological impacts associated with iron are direct (toxicity) or indirect (physical). Outside of these key influences, greater aqueous levels of hardness, suspended clay, sulphide and dissolved organic substances can also limit the bioavailability of ferrous iron and/or remove it from solution (Loeffelman et al. 1985; Lappivaara et al 1999; Peuranen et al. 2003; Van der Welle et al. 2006). Suspended clay particles and dissolved organic carbon can accelerate precipitation of ferric iron forms (Jones et al. 1993; Wetzel 2001), with resulting larger aggregate sizes less likely to adversely impact aquatic organisms through clogging of biological membranes (Smith et al. 1973). Moreover, adsorption of other, potentially more toxic, metal ions (e.g., arsenic, cobalt, copper, zinc) to ferric (oxy)hydroxide precipitate/complexes can limit the bioavailability of these metals and/or remove them from solution (Davison 1993; Weltens et al. 2000) reducing overall toxicity. Finally, by lowering the metabolism of aquatic organisms, cooler water temperature may limit adverse respiratory impacts associated with ferric (oxy)hydroxide precipitation on biological membranes (Peuranen et al. 2003).

Differentiation between ferrous and ferric forms of iron may be important for determining toxicological modes of action. However, very few studies report and/or differentiate ferrous iron from ferric iron concentrations. Except at high concentrations, no correlation between total iron and ferric or ferrous iron appear to exist (Ohio EPA 1998), precluding any predictions of iron speciation based on total iron concentrations. Like other readily hydrolysable metals, the ‘dissolved’ fraction of iron (i.e., that fraction < 0.45 mm in diameter; USEPA 1993) may contain discrete particles and therefore does not represent a true, ‘soluble’ fraction as for other metals (Loeffelman et al. 1985). As a result, dissolved metal fractions do not necessarily represent more ‘bioavailable’ forms of iron, which is generally considered to be the ferrous iron component (Loeffelman et al. 1985). Truly ‘dissolved’ iron concentrations (i.e., the amount of iron present strictly in chemical solution) are very low in pH neutral, well oxygenated waters (Horowitz 1991), with Forstner and Wittman (1979) reporting a typical range from 0.7 to 6.6 ug/L.

4.4.3 Sensitive Species and Relevance to Faro

The maximum iron concentrations predicted to occur in surface waters downstream of the Faro Mine complex after closure and remediation were approximately 5,000 ug/L (Table 2.1), so data were tabulated for organisms showing toxicity at concentrations within this range (Table 4.8). Iron is acutely toxic to freshwater invertebrates and fish only at relatively high concentrations, typically greater than 1 mg/L (Table 4.8), and often much higher. The stream-dwelling mayfly *Ephemerella subvaria* and fathead minnow showed highest acute sensitivity to iron at 320 ug/L (Table 4.8).

Some of the aquatic biota that are sensitive to iron are not found in Yukon rivers (Table 4.9). Of all the species which appear to be very sensitive to iron, northern pike is the only one previously reported in waters near the Faro Mine complex (Table 4.9). While the other species in Table 4.9 have not been specifically reported in previous studies conducted at Faro, some of the same genera have been observed (e.g., *Ephemerella*, *Daphnia*, *Thymallus*, *Oncorhynchus*). Therefore, CWQGs developed to protect sensitive aquatic species against the effects of iron cannot be considered overly protective with respect to the species assemblage found, or potentially present, near Faro.

4.5 Lead

4.5.1 Aquatic Chemistry and Behaviour

Lead is the 36^{th} most abundant element by concentration in the earth's crust and occurs naturally in aquatic ecosystems due to weathering (CCREM 1987). In the 1980s, the concentration of lead in Canadian surfaces waters ranged from <1 ug/L to 77 ug/L (CCREM 1987), with background concentrations typically being less than 5 ug/L (Nagpal 1987). The upper range of concentrations has likely declined in more recent years due to reductions in the production and use of leaded-gasoline since the late 1970s (CCREM 1987).

Lead exists in four environmentally-relevant oxidation states (0, 1+, 2+, and 4+), with 2+ being the predominant stable ionic species and 1+ being of minor importance (CCME 1999). Between pH 6 and 8, solubility of lead is a complex function of pH and dissolved CO2 (Demayo et al. 1980, CCREM 1987). Generally speaking, the solubility of lead increases at pHs below 6.5 and decreases under conditions of constant pH and increasing alkalinity. At pHs below 5 to 6, the concentration of the stable cations increases (CCME 1999) with Pb2+ presumed to be one of the most biologically available forms (Niyogi and Wood 2004). Alkalinity and hardness promote the insolubility of lead through an increased tendency to precipitate and bind to colloidal particles (Demayo et al. 1980). At the low lead concentrations typical of many aquatic environments, most of the lead in the dissolved phase

Table 4.8: Summary of studies reporting iron toxicity to aquatic biota at concentrations ≤ 5,000 µg/L.

	Scientific Name	Common Name	Hardness (mg/L as CaCO3)	Effect Concentration [a] (Total ug/L)	Reference [b]
US EPA 1976	*Ephemerella subvaria*	Mayflies	-	320	Warnick and Bell 1969
	Cyprinus carpio	Common Carp	-	900	Brandt 1948
	Esox lucius	Northern pike	-	1000	Doudoroff and Katz 1953
CCME 1999	*Pimephales promelas*	Fathead minnow	-	1500	Sykora et al. 1972
	Gammarus minus	Scud	-	3000	Sykora et al. 1972
USEPA Ecotox Database	*Salmo trutta*	Brown trout	-	100	Dalzell and Macfarlane 1999
			-	200	Dalzell and Macfarlane 1999
	Pimephales promelas	Fathead minnow	102.7	320	Birge et al. 1985
	Cyprinus carpio	Common carp	-	560	Alam and Maughan 1995
	Pimephales promelas	Fathead minnow	102.7	570	Birge et al. 1985
	Daphnia pulex	Water flea	93.6	700	
			93.6	960	
	Pimephales promelas	Fathead minnow	102.7	1010	
	Cyprinus carpio	Common carp	-	1220	Alam and Maughan 1995
	Daphnia pulex	Water flea	93.6	1310	Birge et al. 1985
	Cyprinus carpio	Common carp	-	1360	Alam and Maughan 1995
			-	2250	
	Daphnia pulex	Water flea	-	2800	Lee 1976
	Cyprinus carpio	Common Carp	-	3000	Alam and Maughan 1995
	Pimephales promelas	Fathead minnow	-	3700	Loeffelman et al. 1986
	Oncorhynchus mykiss	Rainbow trout	-	4400	Loeffelman et al. 1986
Literature	*Thymallus thymallus*	Grayling	-	1000	Peuranen et al. 2003

[a] Data represent a variety of endpoints (eg. LC50 or ED50 for acute tests).

Table 4.9: Species sensitive to iron at concentrations ≤ 5,000 µg/L.

	Scientific Name	Common Name	Iron Toxicity Range (µg/L)[a]		Distribution	Preferred Habitat	Related Species Found Near Faro
Invertebrates	*Ephemerella subvaria*	Mayflies	320	Warnick and Bell 1969	Eastern and central North America	Erosional, in streams.	Family Ephemerelidae
	Daphnia magna	Water flea	5900	Biesinger and Christensen 1972	Temperate North America, Europe, Asia, Africa	Small weedy lakes, saline ponds, temporary ponds. Tolerant of low oxygen.	Order Cladocera [b]
	Daphnia pulex	Water flea	700 - 17,350	Birge et al. 1985	Temperate North America	Small ponds with abundant organic matter	Order Cladocera [b]
	Gammarus minus	Scud	3000	Sykora et al. 1972	Eastern North America	Caves pools, springs, small streams	-
Fish	*Salmo trutta*	Brown trout	100 - 47,000	Dalzell and Macfarlane 1999	Native to Europe and western Asia, introduced to North America	Clear, cool, well-oxygenated streams and lakes. Temperatures up to 24°C	Salmonid species [c]
	Pimephales promelas	Fathead minnow	320 - 21,840	Birge et al. 1985	Central North America	Still waters of ponds to flowing waters of streams	-
	Cyprinus carpio	Carp	560 - 2250	Alam and Maughan 1995	Native to parts of Asia and Europe, introduced to North America	Wide range of habitat from streams to lakes	-
	Esox lucius	Northern pike	1000	Doudoroff and Katz 1953	Much of the upper northern hemisphere, **Found in Vangorda Cr, Pelly R**	Clear, warm, heavily vegetated rivers and bays	Northern pike (*Esox lucius*)
	Thymallus thymallus	Grayling	1000	Peuranen et al. 2003	Britian to Central Europe	Clear, cold water of creeks, rivers and lakes.	Salmonid species [c]
	Oncorhynchus mykiss	Rainbow trout	4,400 - 18,000	Loeffelman et al. 1986	Native to west coast of North America, Introduced throughout North America and worldwide.	Gravelly streams required for spawning. Found in rivers and lakes.	Salmonid species [c]
	Salvelinus fontinalis	Brook trout	12000	Smith et al. 1973	Northeastern North America	Clear, cool, well-oxygenated streams and lakes. Temperatures below 20°C	Salmonid species [c]

[a] As reported by USEPA Ecotox database, USEPA 1976, CCME 1999 and recent literature. References are for low and high ends of data range, respectively.

[b] Cladocera reported at Faro:
Alona sp
Bosmina longirostris
Chydorus gibbus
Daphnia sp
Eurycercus (Bullatifrons) sp

[c] Salmonids reported at Faro:
Round whitefish (*Prosopium cylindraceum*)
Arctic grayling (*Thymallus arcticus*)
Lake whitefish (*Coregonus clupeaformis*)
Chinook salmon (*Oncorhynchus mykiss*)

may be complexed by organic (e.g, humic acids) or inorganic (hydroxides, iron and manganese oxides, and clays) ligands, much of which will precipitate to sediments (Demayo et al. 1980, CCREM 1987, Bradley and Cox 1988). However, under conditions of low pH, lead adsorbed onto mineral particulates can be released into the water column (Brown et al. 1999).

4.5.2 Mode of Toxicity and Modifying Factors

Lead has no nutritional role in organisms (Hodson 1988). Earlier studies state that organolead species are more toxic than inorganic lead (Demayo et al. 1980) while more recent work has focused on Pb^{2+}, which is believed to be one of the most bioavailable and therefore most toxic lead species (Niyogi and Wood 2004). Pb^{2+} gains entry into fish through voltage-independent calcium channels on the surface of gill chloride cells and subsequently causes a decrease in the activity of an ATP-driven baso-lateral calcium pump (Rogers and Wood 2004). Entry of lead through calcium channels occurs relatively quickly (3 hours) whereas inhibition of pump activity takes place over a longer time period (96 hours) (Rogers and Wood 2004). Similar to the effects of cadmium exposure, exposure to lead results in competitive inhibition of calcium absorption and a decrease in plasma and whole-body concentrations of this essential ion (Rogers et al. 2003; Rogers and Wood 2004). The mechanism of lead's acute toxicity also involves decreases in the plasma concentrations of sodium and chloride presumably due to the inhibitory effect of lead on the activity of the gill's baso-lateral Na+/K+ ATPase (Rogers et al. 2003). Conversely, plasma concentrations of magnesium, ammonia and the stress hormone cortisol have been shown to increase in lead-exposed fish (Rogers et al. 2003).

In studies using adult fish, the mechanism of acute toxicity for lead is believed to be ionoregulatory disturbances whereas respiratory functioning, as measured by plasma concentrations of oxygen tension and carbon dioxide, is unaffected (Rogers et al. 2003). However, smallmouth bass fingerlings that died as part of a 96-hr LC50 bioassay (96-hr LC50 determined as 29.0 mg/L Pb) exhibited several gill histopathologies, including excess mucous and fused filament tips, which are indicative of impaired respiration (Coughlan et al. 1986). A study by Sola et al. (1994) also reported structural alterations to gills such as massive oedema of the secondary lamellae in rainbow trout (average weight 240 g) exposed to 1 mg/L Pb for one day. In developing fish, chronic exposure to lead causes a black colouration of the caudal area ("black tail") and a curvature of the spine, both of which are signs of neurotoxicity (Demayo et al. 1980; Hodson et al. 1982), as well as a decrease in the activity of blood delta-aminolevulinic acid dehydratase, an enzyme required for the biosynthesis of hemoglobin (Hodson et al. 1982) and a common marker of lead exposure in animals (Goyer and Clarkson 2001).

Daphnia magna exposed to 0.9 mg/L Pb (pH 8.1 and hardness 217 mg/L as $CaCO_3$) for 10 days (with generous food rations) grew to be smaller in length and produced smaller sized progeny than *Daphnia* exposed to lower lead levels (0 – 0.27 mg/L; Enserink et al. 1995). In order to simulate conditions in the wild, parallel tests were also conducted in which *Daphnia* were given limited food rations (as an added stressor). *Daphnia magna* receiving limited food showed delayed ovary/egg development and produced smaller broods. In addition, *Daphnia* receiving more food coped with 0.9 mg/L lead exposures by allocating less energy to body maintenance, while those on limited rations allocated less energy to growth, indicating organism responses to lead may be affected by the presence of additional stressors.

Similar to fish, invertebrate gills consist of proteins that could offer sites for interaction with lead and subsequent toxic responses (Paquin et al. 2002).

A similar mechanism of inhibited calcium uptake as seen with fish has been observed for the unicellular alga, *Chlorella vulgaris* (Niyogi and Wood 2004). Lead absorbed by plants may inhibit photosynthesis in certain species (Demayo et al. 1980) while others are largely unaffected (Wundram et al. 1996). Photosynthetic activity of *Chlamydomonas reinhardtii* incubated with 10 mg/L Pb in a brine solution for four hours was virtually 100% whereas exposure to mercury, copper or cadmium resulted in significant inhibition. Other plant species were less resistant to lead; *Lepidium sativum* showed a 20% reduction in root growth within 8-days while leaf growth in *Lemna minor* was inhibited by approximately 60% over the same time span (Wundram et al. 1996).

A lead-only 96-hour exposure using rainbow trout observed the greatest accumulation in gills, followed by kidney and liver (Rogers et al. 2003). However, rainbow trout exposed for 30 days to river water contaminated with lead and other metals accumulated the greatest concentration of lead in bone followed by kidney, spleen, gills, and muscle (Camusso et al. 1995). *Hyalella azteca* did not show obvious evidence of long-term storage of lead in the form of insoluble granules after 8 days exposure to 480 nM (or 0.1 mg/L Pb) (Maclean et al. 1996) while macrophytes growing in contaminated waterbodies can accumulate lead in roots and shoots (Duman et al. 2007).

The toxicity of lead to carp eggs was greatly enhanced at water pH of 5.6 compared to 7.5 and was attributed to increased bioavailability of Pb (more Pb^{2+}) at lower pH (Strouthart et al. 1994). Similarly, lead toxicity to several species of fish and aquatic invertebrates was greatest at pH 6.3 and least at pH 8.3 (Shubauer-Berigan et al. 1993). Mortality rates of *Hyallela azteca* increased with increasing free lead concentrations during acute laboratory exposures (Freedman et al. 1980).

A change to the solubility of lead is reflected in higher acute and chronic toxicity in water of low compared to high hardness (CCME 1999). The presence of organic matter, calcium, magnesium, alkalinity, and sodium in water may decrease the amount of lead that is bioavailable for uptake at the gill through either binding or competitive interactions (McDonald et al. 2002). Although the concentration of lead accumulated in the gill has been shown to decrease as the concentration of calcium in the ambient water increases, the concentration of calcium required to inhibit lead uptake increases with lead concentration (Rogers and Wood 2004). The protective effect of calcium is similar to that seen with other divalent cations such as cadmium and zinc (Niyogi and Wood 2004) and involves competition as well as a reduction in the permeability of gill cell membranes thereby further limiting the entry of ions such as lead (Wood 2001). Even though organic matter can ameliorate the toxicity of lead to fish by reducing the concentration of the biologically available form, lower aquatic organisms have demonstrated an opposite trend (Kungolos et al. 2006). For instance, the toxicity of lead to the bioluminescent bacterium *Vibrio fischeri* increased in the presence of humic acids (Kungolos et al. 2006).

Acclimation to non-lethal levels of lead may result in increased tolerance of aquatic organisms. Evidence to this effect was demonstrated in rainbow trout exposed to 1 mg/L Pb; initial histopathological alterations to the gills, including massive oedema and degenerating chloride cells, was followed by a return to normal appearance after 6 days of continued exposure (Sola et al. 1994). This process of damage to tissue followed by repair may indicate that the fish have increased tolerance to lead (i.e. could potentially withstand a higher concentration) (McDonald and Wood 1993). Acclimation to lead by populations of macroinvertebrates is also a possibility. Crustaceans of the species *Asellus aquaticus* growing in rivers containing elevated concentrations of lead compared to *A. aquaticus* reared in rivers with lower levels showed better survival when challenged with increased lead concentrations (Demayo et al. 1980). Tolerance to lead by macroinvertebrates has also been demonstrated as decreased lead accumulation and alleviated growth inhibition (Demayo et al. 1980).

4.5.3 Sensitive Species and Relevance to Faro

The maximum concentration of lead in surface waters downstream of Faro after mine closure and remediation was predicted to be 15 ug/L (Table 2.1). Toxicity data were tabulated for organisms showing lead toxicity at concentrations of 150 ug/L or less to identify the species most sensitive to lead (Table 4.10). The lowest effect concentrations (greatest toxicity) were reported for *Daphnia magna* (12 ug/L), *Hyallela azteca* (16 ug/L), and rainbow trout (19 ug/L), with all other species showing toxicity at concentrations greater than 20 ug/L (Table 4.10). Overall, the data indicate that a variety of aquatic biota, including plant, invertebrate and fish species may be affected by lead concentrations below 150 ug/L. While none of the species on Table 4.11 were specifically reported in previous studies conducted at Faro, several of the same genera are known to be present (e.g., *Oncorhynchus*, *Daphnia*) (Table 4.5). Based on ubiquitous distribution, other organisms may be present, but have not been previously reported (e.g., *Hyallela, Gammarus, Chlorella*). Therefore, CWQGs developed to protect sensitive aquatic species against the effects of lead cannot be considered overly protective with respect to the species assemblage found, or potentially present, near Faro.

Table 4.10: Summary of studies reporting lead toxicity to aquatic biota at concentrations ≤ 150 µg/L.

		Scientific Name	Common Name	Hardness (mg/L as CaCO3)	Effect Concentration [a] (Total ug/L)	Reference [b]
US EPA 1984	Acute	*Gammarus pseudolimnaeus*	Amphipod	46	124	Spehar et al. 1978
		Gammarus pseudolimnaeus	Amphipod	48	140	Call et al. 1983
	Chronic	*Daphnia magna*	Cladoceran	52	12.26	Chapman et al. Manuscript
		Oncorhynchus mykiss	Rainbow trout	28	18.88	Goetti et al 1972, Davies and Everhart 1973, Davies et al 1976
		Lymnaea palustris	Snail	139	25.46	Borgmann et al. 1978
		Salvelinus fontinalis	Brook trout	44	83.08	Holcombe et al. 1976
		Oncorhynchus mykiss	Rainbow trout	35	101.8	Sauter et al. 1976
		Daphnia magna	Cladoceran	102	118.8	Chapman et al. Manuscript
		Daphnia magna	Cladoceran	151	128.1	Chapman et al. Manuscript
Ecotox		*Hyalella azteca*	Scud	-	16	Phipps et al. 1995
		Ceriodaphnia dubia	Water flea	20	51	Jop et al. 1995
		Ceriodaphnia dubia	Water flea	20	71	Jop et al. 1995
		Ceriodaphnia dubia	Water flea	20	99	Jop et al. 1995
		Ceriodaphnia dubia	Water flea	20	107	Jop et al. 1995
		Ceriodaphnia dubia	Water flea	-	150	Jop et al. 1995
Other Literature		*Hyalella azteca*	Scud	130	20.1-44.7	MacLean et al. 1996
		Oncorhynchus mykiss	Rainbow trout	-	125-200	MacDonald et al. 2002
		Chlorella pyrenoidosa	Green alga	-	140	Lin et al. 2007

[a] Data represent a variety of endpoints (eg. LC50 or ED50 for acute tests and some chronic tests, as well as highest no-effect concentrations or chronic threshold values for chronic tests).

[b] References cited in USEPA 1984, except more recent studies which are cited listed in Section 7.0.

Table 4.11: Species sensitive to lead at concentrations ≤ 150 µg/L.

	Scientific Name	Common Name	Lead Toxicity Range (µg/L) [a]		Distribution	Preferred Habitat	Related Species Found Near Faro
Invertebrates	*Daphnia magna*	Water flea	12.26 - 5000	Chapman et al. Manuscript, Bringmann and Kuhn 1959a,b	Temperate North America, Europe, Asia, Africa	Small weedy lakes, saline ponds, temporary ponds. Tolerant of low oxygen.	Order Cladocera [b]
	Hyalella azteca	Scud	16 - 5,000	Phipps et al. 1995, Freedman et al. 1980	North America: northern Canada to southern Chile	Benthic in ponds are streams, wide thermal tolerance, usually in association with macrophytes or detritus.	-
	Lymnaea palustris (= Stagnicola elodes)	Snail	25.46	Borgmann et al. 1978	Widespread in northern Canada	Ponds and streams with mud bottom and thick vegetation	Class Gastropoda Family Valvatidae
	Ceriodaphnia dubia	Water flea	51 - 340	Jop et al. 1995	Palearctic and North America	Nearshore of rocky lakes and ponds	Order Cladocera [b]
	Gammarus pseudolimnaeus	Amphipod	124 - 140	Spehar et al. 1978, Call et al. 1983	North American streams, widespread	Detritus and gravel	-
Fish	*Oncorhynchus mykiss*	Rainbow trout	18.88 - 542,000	Goetti et al 1972, Davies and Everhart 1973, Davies et al 1976	Native to west coast of North America. Introduced throughout North America and worldwide.	Gravelly streams required for spawning. Found in rivers and lakes.	Salmonid species [c]
	Salvelinus fontinalis	Brook trout	83.08 - 4,100	Holcombe et al. 1976	Northeastern North America	Clear, cool, well-oxygenated streams and lakes. Temperatures below 20^{o}C	Salmonid species [c]
	Oncorhynchus mykiss	Rainbow trout	125-200	MacDonald et al. 2002	Native to west coast of North America, Introduced throughout North America and worldwide.	Gravelly streams required for spawning. Found in rivers and lakes.	Salmonid species [c]
plants	*Chlorella pyrenoidosa*	Green alga	140 - 680	Lin et al. 2007	Widespread genus	-	Various green and blue-green algae and diatoms

[a] As reported by USEPA Ecotox database and recent literature. References are for low and high ends of data range, respectively.

[b] Cladocera reported at Faro:
- *Alona* sp
- *Bosmina longirostris*
- *Chydorus gibbus*
- *Daphnia* sp
- *Eurycercus (Bullatifrons)* sp

[c] Salmonids reported at Faro:
- Round whitefish (*Prosopium cylindraceum*)
- Arctic grayling (*Thymallus arcticus*)
- Lake whitefish (*Coregonus clupeaformis*)
- Chinook salmon (*Oncorhynchus tshawytscha*)

5.0 WATER QUALITY AT FARO

5.1 Background Concentrations

The upper range of background concentrations for cadmium slightly exceed the generic CWQG (Table 5.1). Background levels for copper, iron, lead, and zinc were similar to or less than the applicable CWQG (Table 5.1). Therefore, cadmium is the only COC for which a SSWQO could be derived using the background concentration procedure, although an increase of only 0.01 ug/L (SSWQO relative to CWQG) would be achievable (see also Section 6.1).

5.2 Modifying Factors

Water pH at both background and mine-exposed surface waters near Faro tend to be slightly alkaline (mean pH 7.7-7.9), with low DOC (<3.5 mg/L) (Table 5.2). Background stations generally exhibit moderate hardness (mean 77 mg/L), although this varies considerably within and among stations (range 7 to 261 mg/L). On average, water hardness at mine-exposed stations is about 3 times higher (234 mg/L) than at background stations and also exhibits considerable seasonal variability (54 to 812 mg/L) depending on the relative contributions of natural and mine-related sources. The components of hardness, calcium and magnesium, also tend to be about 3-4 times higher at mine-exposed than background stations (Table 5.2). Similarly, sodium and alkalinity concentrations are generally about three times higher downstream relative to upstream of the Faro Mine complex. The data suggest that some protective effect from hardness, its components (calcium, magnesium), sodium, and/or alkalinity may be available in mine-exposed compared to background waters near Faro. However, it should be noted this is an anthropogenic, not natural, condition and potential temporal changes in such modifying factors need to be evaluated with respect to predicted temporal changes in COCs. The potential influence of modifying factors on the toxic expression of each COC at Faro is discussed in Section 6.0.

Table 5.1: Background COC concentrations[a] relative to generic CWQG.

Parameter	Concentration (mg/L)	
	Background Concentration	CWQG
Cadmium	0.00004	0.00003[b]
Copper	0.002	0.002[b]
Iron	0.25	0.3
Lead	0.0005	0.002[b]
Zinc	0.016	0.03

[a] Data from 2005-2007, Appendix B
[b] Based on hardness of 100 mg/L

Table 5.2: Potential toxicity modifying factors in surface waters near Faro.

Parameter	Units	Concentration (mg/L) [a]							
		Background Stations				Mine-exposed Stations [b]			
		Mean	Median	Range	n	Mean	Median	Range	n
pH	pH units	7.70	7.77	6.45 - 8.44	87	7.89	7.90	7.2 - 8.5	124
Hardness	mg/L	77	59	7 - 261	87	234	197	54 - 812	130
Calcium	mg/L	22.3	17.6	2.07 - 77.8	93	64.3	55.8	16.1 - 243	131
Magnesium	mg/L	4.5	3.1	0.4 - 16	93	17.9	13.2	3.24 - 49.8	131
Sodium	mg/L	2.15	1.94	0.77 - 5.5	93	5.42	4.45	1.37 - 27.7	131
Alkalinity	mg/L	60	37	11.6 - 159	43	133	122	32.5 - 255	104
Dissolved Organic Carbon	mg/L	1.8	1.6	0.6 - 3.5	22	1.9	2.0	0.8 - 2.55	5

[a] data from 2005 to 2007, Appendix Tables B.1 and B.2

[b] data for stations V8 and X14

6.0 POTENTIAL SSWQO DEVELOPMENT

The applicability of the different procedures for SSWQO development depends on: a) natural background concentrations relative to the generic water quality guideline, b) the composition and contaminant-sensitivity of resident aquatic species, and c) the degree to which contaminant toxicity is likely modified by other site-specific water quality factors (Table 6.1). Below, these considerations are evaluated for each of the contaminants considered to be of potential future concern at Faro (COCs).

The conclusions and recommendations provided herein should be considered draft until the CWQGs have been updated for metals. Draft guidelines for cadmium and zinc are expected to be released for public comment by mid 2008.

6.1 Cadmium

6.1.1 Summary of Relevant Information

The aquatic biota found or potentially present near the Faro Mine complex include species that are among the most sensitive to zinc, or that are closely related to sensitive species reported in the scientific literature. Therefore, it seems unlikely that the **Recalculation Procedure** will yield a substantially different SSWQO value than the generic guideline. Similarly, for the **Resident Species Procedure**. the data do not suggest that the resident species will be more sensitive or tolerant of cadmium than sensitive species identified in the literature. Furthermore, the number of tests that would need to be conducted and the requirement to use resident species that may not be readily transported to or currently cultured in commercial laboratories makes this latter procedure very time consuming and expensive (i.e., last resort).

The influence of water hardness on cadmium toxicity is taken into account in the generic CWQG and is expected to be incorporated in the updated guideline. Therefore, the usefulness of the **Water Effect Ratio Procedure** (WERP) or **Biotic Ligand Model** (BLM) for developing a SSWQO for cadmium would be largely dependent on the presence of other water quality factors that might substantially ameliorate cadmium toxicity. The scientific literature generally suggests that the effects of factors such as pH and DOC on cadmium toxicity are relatively modest compared to the influence of calcium and hardness (Table 6.2; Section 4.1.2). Also, ameliorating effects from pH and DOC would be expected at lower and higher levels, respectively, than those typically observed at Faro.

Table 6.1. Summary of approaches for SSWQO development (adapted from CCME 2003).

Procedure for SSWQO Development	Approach	Most Appropriate Applications
Background Concentration Procedure	Replace generic guideline with background concentration	Background concentrations exceed the generic guideline. Pristine, highly-valued waters and/or waters with threatened or endangered species.
Recaculation Procedure	Use the same standard procedures as for deriving generic guidelines after removing data for non-resident species	Sensitivity range of resident species differs from that of complete toxicological data set used to derive the generic guideline
Water Effect Ratio Procedure and Biotic Ligand Model	Conduct toxicity tests using both site water and laboratory water into which the contaminant of interest has been added. The ratio of test results is used to adjust the generic guideline.	Factors are present that are expected to influence the bioavailability of contaminants.
Resident Species Procedure	Conduct toxicity tests with resident species and site water and use the results to derive the SSWQO following the same procedures as used for generic guideline development	Resident species have a unique range of sensitivity to contaminant of interest, possibly partially due to site-specific factors influencing bioavailability.

Table 6.2: Summary of factors that have been shown to have modifying effect on toxicity of metal COCs.

Metal	Effect on Metal Toxicity						
	↑ Ca	↑ Mg	↑ Hardness	↑ H+/↓pH		↑ Na	↑ Dissolved Organic Matter
				on total metal[a]	on free ion		
Cadmium	↓↓		↓↓		↓	↓	↓[b]
Zinc	↓↓	↓[f]	↓↓	↑	↓	↓	↓[b]
Copper	↓	↓	↓	↑	↓	↓	↓↓
Iron[c]				↑			↓[d]
Lead	↓	↓	↓	↑		↓	↓↓[e]

↓↓ strong modifying effect
↓ small, if any, modifying effect

[a] Reduced pH (more H+) causes a greater proportion of metal to be present in free ionic form.

[b] Stronger effect in soft than hard water.

[c] Oxidation-reduction potential affects speciation with reducing conditions favouring the more toxic ferrous (Fe^{2+}) ion and oxidizing conditions favouring the ferric (Fe3+) ion.

[d] While direct toxicity may be reduced by inceased pH or DOM, increased indirect effects may occur as a result of iron complexes precipitating on gills or eggs.

[e] Trend may be reversed among some lower aquatic organisms (e.g., bacteria) exposed to lead.

[f] Relationship may be stronger for invertebrates and algae than fish.

Therefore, it is likely that the generic CWQG could be raised by only a small factor (likely <4) if at all, based on water quality considerations. This is substantiated by other studies which have

have investigated SSWQO development for cadmium. Toxicity tests conducted using river water with pH of 6.6 to 7.5, moderate hardness (68-108 mg/L), and low to moderate DOC (1.2-7.9 mg/L), yielded a Water Effect Ratio (WER) for cadmium of <2 (i.e., the factor by which the generic guideline could be raised) (Diamond et al. 1997). A cadmium WER of 3.9 was demonstrated by Jop et al. (1995) for a river with neutral pH and low hardness (<30 mg/L; DOC not reported). A mean WER of 3.9 (geometric mean of results for three species) was developed for the St. Louis River, which has slightly alkaline pH (7.2-7.8), hardness of approximately 65 mg/L, and relatively high DOC (14-23 mg/L). While each WER is site-specific, these examples identify the range of values that can be expected for waters with varying concentrations of modifying factors.

Development of the “biotic ligand model” (BLM - formerly the “gill model”) in recent years has attempted to better account for the bioavailability of metals to aquatic life. The BLM, which quantifies the capacity of metals to bind to the gills of aquatic organisms, has been proposed as a reliable method for estimating the bioavailable portion of dissolved metals in the water column based on site-specific water quality parameters such as alkalinity, pH and dissolved organic carbon (McGeer et al. 2000, Meyer et al. 1999, Pagenkopf 1983, Di Toro et al. 2001, Paquin et al.2002; Santore et al. 2002, USEPA 2007). Thus, a BLM allows for desk-top prediction of WERs under different water quality scenarios (Di Toro et al. 2001). However, the BLM for cadmium is still in development and reliable models for predicting the site-specific bioavailability and toxicity of cadmium may not be available for several more years (Niyogi and Wood 2004, Croteau and Luoma 2007).

6.1.2 Recommendation and Rationale

Cadmium concentrations downstream of the Faro Mine complex do not appear to be of concern at the present time (Minnow 2007). Water quality predictions (presented by Senes 2006) did not identify when cadmium concentrations will rise at Faro, but an updated water quality model is expected to be completed in mid 2008 that will predict temporal changes. If elevated cadmium concentrations are not expected for a decade or more, the concern is long-term, rather than immediate. In the interim, new information (e.g., a BLM for cadmium or updated requirements for SSWQO development) may negate the validity of a SSWQO developed in the short-term.

Also, it is expected that a revised CWQG will be released in draft by mid-2008 for public review and will likely be formally approved and published (possibly after revisions) within 6-12 months thereafter. Early indications are that the guideline may rise by approximately one order of magnitude relative to the current guideline (at any given water hardness), which would further diminish any immediate concern regarding potential effects of cadmium releases from Faro. In addition, the documentation associated with the updated guideline will include a species sensitivity distribution model, which may be useful in augmenting the information presented herein with respect to assessing potential future impacts associated with cadmium.

Lastly, data presented in the scientific literature suggest that any relief that may be afforded by development of a SSWQO for cadmium at Faro is likely to be minor, at best (e.g., ≤4-fold and, more likely, a <2-fold increase).

Therefore, it is not recommended that a SSWQO be developed for cadmium at the present time. This conclusion should be revisited once the updated water quality model and CWQG are available in case either alters the expectations stated above. Regardless, the toxicity information summarized in this document and the additional data that will be forthcoming in the supporting rationale document for the revised CWQG for cadmium will be adequate to assess potential impacts (or establish that none will be expected) associated with cadmium in the Environmental Risk Assessment and overall Environmental Assessment process for mine closure.

6.2 Zinc

6.2.1 Summary of Relevant Information

As identified for cadmium, the aquatic biota found or potentially present near Faro include species that are among the most sensitive to zinc, or that are closely related to sensitive species reported in the scientific literature. Therefore, it seems unlikely that the **Recalculation Procedure** will yield a substantially higher SSWQO value than the generic guideline. This was the conclusion of an earlier attempt to apply this procedure for development of a SSWQO for zinc in the South McQuesten River, Yukon (CCME 2003). Similarly, the **Resident Species** Procedure, is unlikely to result in a beneficial SSWQO because the data do not suggest that the resident species will be dramatically more tolerant of zinc than those for which zinc toxicity data exist in the literature.

Relative to the current CWQG for zinc (30 ug/L), the utility of the **WERP** would likely be greatest in environments with high hardness, high DOC, and neutral to slightly basic pH (e.g., pH 7-8). While insufficient data were available to account for water hardness at the time the CWQG for Zn was first developed, the effects of hardness (and its components-calcium and magnesium) on ameliorating zinc toxicity to aquatic biota are now well documented (Table 6.2). Therefore, it is expected that the updated CWQG, which is likely to be drafted in 2008, will largely take the influence of water hardness near Faro into account.

High DOC and slightly alkaline pH are factors that would cause much of the Zn present to be complexed to organic matter, and some of the inorganic zinc to be in less toxic (relative to the free ion) hydroxide and/or carbonate forms. Neutral to slightly alkaline pH would also minimize the formation of potentially ingestible suspended or colloidal zinc forms that occur at higher pH (i.e., pH>8). Indeed the pH at Faro is slightly alkaline (Table 6.1) and thus may afford some protection against zinc toxicity relative to tests conducted at other pH values and presented in the literature. However, DOC is quite low in surface waters near Faro. Sodium is the only other water quality factor that has been shown to modify zinc toxicity, but its influence was relatively modest (Table 6.2).

The **WERP** has been used at other sites for derivation of SSWQO for zinc (Diamond et al. 1997, Vigneault et al. 2004; Deforest et al. 2004). Toxicity tests conducted using river water with low to moderate DOC (1.2-7.9 mg/L) and pH of 6.6 to 7.5 yielded a WER of 2.0, indicating zinc was only slightly less toxic in site water than laboratory water (Diamond et al. 1997). At a mine site where the receiving water had high DOC and was naturally soft, effluent liming resulted in higher downstream hardness (Vigneault et al. 2004). Acute and sublethal toxicity tests yielded a maximum WER for zinc of 5 in downstream waters. DeForest et al. (2004) used biotic ligand models to predict WERs for zinc in stormwater discharges at an airport of between 2 and 5.

The **BLM** approach for zinc is relatively less advanced in comparison to copper and silver (Niyogi and Wood 2004), although there have been numerous recent advances (De Schamphelaere and Janssen 2004, De Schamphelaere et al. 2005, Hiejerick et al. 2005b, Muyssen et al. 2006, Wilde et al. 2006). It seems likely that a model version that adequately predicts chronic zinc toxicity to a variety of biota will soon be available.

6.2.2 Recommendation and Rationale

An updated water quality guideline is expected to be completed in draft form by mid-2008 that may alter any conclusions that would be made now regarding the risk of future aquatic impacts associated with zinc. The documentation is expected to include a species sensitivity distribution model that may be useful in augmenting the information presented herein with respect to assessing potential future impacts associated with zinc. It is also expected that the new guideline will likely take into account the relative effects of water hardness (and perhaps other water quality factors if a BLM is used to derive the guideline). It is possible that factors other than hardness, such as DOC and sodium, that are present in site water at Faro may not be taken into account by the new guideline and may slightly reduce the toxicity of zinc in site water compared to typical laboratory water (used in much of the published literature related to zinc toxicity), but effects are likely to be modest (e.g., less than 5-fold).

Also, as indicated for cadmium, zinc concentrations downstream of the Faro Mine complex do not appear to be of concern at the present time (Minnow 2007). An updated water quality model for Faro should be available in mid 2008 that will predict when peak concentrations of zinc are likely to occur. As stated for cadmium, if elevated zinc concentrations are not expected for a decade or more, new information (e.g., a BLM for cadmium or updated requirements for SSWQO development) generated in the meantime may negate the validity of a SSWQO developed in the short-term.

Therefore, it is not recommended that a SSWQO be developed for zinc at the present time. This conclusion should be revisited once the updated water quality model and CWQG are available. Regardless, the toxicity information summarized in this document and the additional data that will be forthcoming in the supporting rationale document for the revised CWQG for zinc will be adequate to assess potential impacts (or establish that none will be expected) associated with zinc in the Environmental Risk Assessment and overall Environmental Assessment process for mine closure.

6.3 Copper, Iron and Lead

Copper, iron, and lead are not presently exceeding CWQGs in surface waters downstream of Faro (Minnow 2007). These substances are also of questionable future concern because it is expected that the updated water quality model for Faro will indicate that future concentrations will be less than were originally predicted (in Senes 2006) (D. Hockley, SRK Consulting, November 2007, pers. comm.). Therefore, it is recommended that potential development of SSWQO for these substances be deferred until the updated water quality model is complete and can be used to whether or not these substances are likely to be of concern in the future.

6.4 Summary

The Phase I SSWQO project involved:

- summarizing existing and predicted contaminant concentrations downstream of the Faro Mine complex relative to water quality guidelines for protection of aquatic life to identify contaminants of current or future concern (COCs);

- reviewing and summarizing information regarding the aquatic chemistry of each COC;
- summarizing data describing the toxicity of each COC to sensitive organisms;
- identifying the water quality factors that modify the toxicity of each COC;
- summarizing data for potential modifying factors in surface waters near the Faro Mine complex;
- indicating, of the species identified as being sensitive to each COC, which ones are known to be or are potentially present in surface waters at or near Faro; and
- making recommendations based on the above as to which, if any, COCs may be amenable to SSWQO development, and which of the available procedures for SSWQO development is/are most appropriate.

Cadmium, copper, iron, lead, and zinc were identified as COCs based on predicted water quality downstream of the Faro Mine complex. However, it was concluded that it would be premature to develop SSWQO for COCs at the Faro Mine complex at this time. Current water quality at the mine does not appear to be adversely impacting biota downstream of the site (Rose and Vangorda Creeks) and degraded water quality may not occur for many years into the future. In the meantime, Environment Canada has indicated that Canadian water quality guidelines for most metals will be updated over the coming few years, and that draft guidelines for cadmium and zinc, in particular, will likely be released for public comment in 2008. In addition, water quality predictions for the Faro Mine complex are also being updated and are expected to be available in mid 2008. Potential development of SSWQO should be re-evaluated when this additional information (updated water quality guidelines and predictions) becomes available. It is noteworthy that that the mine complex causes elevations above background levels of substances that tend to modify the toxicity of the identified COCs (e.g., pH, Ca, Mg, hardness, sodium, and alkalinity). While not a natural condition, this will ameliorate the toxicity that may otherwise result from mine-related loadings of these substances. However, it will be important to determine how concentrations of such modifying factors are expected to vary over time, relative to changes in concentrations of the COCs (presumably through updates to the water quality predictions).

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APPENDIX A

APPROACHES FOR DEVELOPING SSWQO FOR PROTECTION OF AQUATIC LIFE

APPENDIX A: APPROACHES FOR DEVELOPING SSWQO FOR PROTECTION OF AQUATIC LIFE

There are a number of accepted options for the development of SSWQOs (USEPA 1983, 1984; CCME 2003). These include the:

- Background Concentration Procedure (BCP);
- Recalculation Procedure (RCP);
- Water Effect Ratio Procedure (WERP); and
- Resident Species Procedure (RSP).

Two of the above procedures do not require toxicity testing (the BCP and RCP) and can be undertaken as a paper exercise if adequate data are available. Although not yet recognized in guidance documents, application of Biotic Ligand Models in SSWQO development is becoming more widespread. Each of these procedures is briefly described below.

A.1 Background Concentration Procedure

The Background Concentration Procedure (BCP) recognizes that it is not possible to meet generic water quality guidelines in surface waters that have natural concentrations of substances that exceed the guidelines. Therefore, the concentration representing the upper limit of regional background concentrations can be established as a SSWQO. Statistical procedures for defining the upper limit of background have included the mean + $t_{\alpha=0.05(1)}$*standard deviation (approximates a 95th percentile) or the 90th or 95th percentile, among others (BCMELP 1997; Hill et al. 2006, Roe et al. 2006).

A.2 Recalculation Procedure

The Recalculation Procedure (RCP) requires knowledge of the aquatic species that occur (or have historically occurred) in the watershed of interest. The RCP involves the removal of data for species that are not resident in the watershed (or surrogates) from the toxicity data set that was initially used to develop the CWQG. Alternatively, data for resident species, or surrogate species, that are not reflected in the guideline data set may be added (e.g., threatened or endangered species). A SSWQO is derived using the same methodology that was originally used to derive the generic guideline, but only considers the reduced data set. If it can be demonstrated that the sensitive species that influenced the derivation of the generic water quality criterion do not occur locally, the recalculation will likely result in a new, higher SSWQO. Conversely, if the most sensitive species are resident in the watershed, then this method would not be expected to result in a change in the water quality criterion.

A.3 Water Effect Ratio Procedure

The Water Effect Ratio Procedure (WERP) can be beneficial under certain physical and chemical conditions. It is based on the fact that toxicity of metals in certain waters (e.g., waters containing chelating substances or high hardness) may be less than was predicted by the laboratory toxicity tests which were used to derive the generic CWQG. Substances whose toxicity is greatly reduced in the presence of such materials, such as copper, are generally amenable to application of the WERP for SSWQO development.

In this method, a ratio is determined between the toxic concentration of a substance in site water versus the toxic concentration in laboratory water. If this ratio is > 1, the data indicate the substance is less toxic in site water. The SSWQO is established by applying this ratio to the generic guideline. For example, if toxicity in site water occurs at concentrations at least three times the concentration in laboratory water, a WER of three is used and the SSWQO would be three times the generic guideline.

A.5 Resident Species Procedure

The Resident Species Procedure (RSP) involves the development of a new objective from scratch, or *de novo*, using site water and species that are found locally. Development of an SSWQO from scratch is expensive because toxicity tests must be conducted with enough organisms to meet minimum data set requirements (typically several fish species, several invertebrate species and one plant or algal species). However, it also provides a highly site-specific water quality objective because both site water and site organisms are used. Due to the expense, it is typically only considered as a last option.

A.6 Biotic Ligand Model

Biotic ligand models simulate site-specific water quality influences on metal bioavailability by accounting for the interaction of a metal with organic and inorganic ligands in the water (speciation), and the competition of other cations at the surface of the receptor organism (the biotic ligand), to predict toxicity. Therefore, a BLM could be used for SSWQO development as an alternative to, or in conjunction with one of the traditional approaches described above (Cruz et al. 2006; Jarvis et al. 2006). Considerable international research has been and continues to be undertaken to develop and validate BLMs to predict the toxicity of various metals in aquatic systems (Allan 2003; HydroQual 2005; Chowdhury et al. 2006, Kozlova et al. 2006, Ranville et al. 2006, Smith et al. 2006, Wisniewski et. al. 2006). However, there are still significant data gaps limiting the application of BLM for some metals. For example, BLMs have not yet been developed for some metals because toxicity data are lacking for a sufficient variety of aquatic species, exposure durations (e.g., acute versus chronic), or water quality conditions. A distinct advantage of such a model would be its to ability to predict whether aquatic toxicity could be expected under future scenarios of water quality at Faro.

APPENDIX B
WATER QUALITY DATA

Table B.1: Summary statistics for background concentrations of Cd, Cu, Fe, Pb and Zn (2005 - 2007).

Cadmium										
Statistic	V1	VR	R7	FDU	W10	R6	Next Creek	Rose Cr Trib U/S K8	SF Rose Ck U/S Haul Rd	Combined
n	17	6	37	11	8	10	1	1	2	93
median	0.0001	0.000016	0.0001	0.0001	0.0001	0.0001	-	-	0.0000075	0.0001
mean	0.000070	0.000019	0.000106	0.000059	0.000081	0.0001	0.00001	0.00001	0.0000075	0.0001
standard deviation	0.000043	0.000010	0.000052	0.000047	0.000035	0.0000	-	-	0.0000035	0.0001
minimum	0.000007/<0.00001	0.00001	<0.0002	0.000009/<0.0002	0.000018/<0.0002	0.00001/<0.0002	-	-	<0.00001	<0.00001
maximum	0.0001	0.000036	0.0004	0.0001	0.0001	0.0001	-	-	0.00001	0.0004
# <detection limit (DL)	13	0	35	6	6	6	0	0	1	67
% <DL	76	-	95	55	75	60	-	-	50	72
# > guideline	12	1	36	6	6	6	0	0	0	67
% > guideline	71	17	97	55	75	60	-	-	-	72
# <DL & DL> guideline	11	0	35	6	6	6	-	-	0	64
% <DL & DL> guideline	65	-	95	55	75	60	-	-	-	69
maximum DL	0.0002	-	0.0002	0.0002	0.0002	0.0002	-	-	0.00001	0.0002
95th pecentile	0.0001	0.0000335	0.0001	0.0001	0.0001	0.0001	-	-	0.00000975	0.0001
95th percentile, detected values or <DL where DL < guideline	0.000033	"	0.0003805	0.0000124	0.0000294	0.00001985	-	-		0.00004
n, detected values or <DL where DL < guideline	6	6	2	5	2	4	1	1		29

Copper										
Statistic	V1	VR	R7	FDU	W10	R6	Next Creek	Rose Cr Trib U/S K8	SF Rose Ck U/S Haul Rd	Combined
n	17	6	37	11	8	10	1	1	2	93
median	0.0005	0.0007	0.0005	0.0005	0.0010	0.0005	-	-	0.0004	0.0005
mean	0.0005	0.0007	0.0007	0.0008	0.0012	0.0007	0.0006	0.0004	0.0004	0.0007
standard deviation	0.0004	0.0002	0.0006	0.0008	0.0008	0.0008	-	-	0.0000	0.0006
minimum	0.00029/<0.001	0.00045	<0.001	0.00039/<0.001	<0.001	0.00035/<0.001	0.0006	0.0004	0.0004	0.00029/<0.001
maximum	0.002	0.00104	0.003	0.003	0.003	0.003	-	-	0.0004	0.003
# <detection limit (DL)	10	0	30	3	2	5	0	0	0	50
% <DL	59	-	81	27	25	50	-	-	-	54
# > guideline	0	0	1	1	1	1	0	0	0	4
% > guideline	-	-	3	9	13	10	-	-	-	4
# <DL & DL> guideline	0	0	0	0	0	0	0	0	0	0
% <DL & DL> guideline	-	-	-	0	-	-	-	-	-	-
maximum DL	0.001	-	0.001	0.001	0.001	0.001	-	-	-	0.001
95th pecentile	0.00080	0.00100	0.002	0.002	0.0027	0.0020	-	-	0.004	0.002

Iron										
Statistic	V1	VR	R7	FDU	W10	R6	Next Creek	Rose Cr Trib U/S K8	SF Rose Ck U/S Haul Rd	Combined
n	17	6	37	11	8	10	1	1	2	93
median	0.025	0.0295	0.12	0.05	0.025	0.115	-	-	0.117	0.07
mean	0.0376	0.0305	0.198108108	0.056727273	0.04175	0.1132	0.014	0.02	0.117	0.113
standard deviation	0.0361	0.006156298	0.262428168	0.035544594	0.041268286	0.048759728	-	-	0.066468	0.182
minimum	0.005/<0.05	0.024	<0.05	0.024/<0.05	0.015/<0.05	<0.05	0.014	0.02	0.07	0.005/<0.05
maximum	0.14	0.042	1.41	0.14	0.14	0.18	-	-	0.164	1.41
# <detection limit (DL)	7	0	4	2	5	1	0	0	0	19
% <DL	41	-	11	18	63	10	-	-	-	20
# > guideline	0	0	3	0	0	0	0	0	0	3
% > guideline	-	-	8	0	-	-	-	-	-	3
# <DL & DL> guideline	0	0	0	0	0	0	0	0	0	0
% <DL & DL> guideline	-	-	-	0	-	-	-	-	-	-
maximum DL	0.05	-	0.05	0.05	0.05	0.05	-	-	-	0.05
95th pecentile	0.1000	0.03925	0.744	0.11	0.1099	0.171	-	-	0.1593	0.25

Lead										
Statistic	V1	VR	R7	FDU	W10	R6	Next Creek	Rose Cr Trib U/S K8	SF Rose Ck U/S Haul Rd	Combined
n	17	6	37	11	8	10	1	1	2	93
median	0.0005	0.00018	0.0005	0.0005	0.0005	0.0005	-	-	0.0001	0.0005
mean	0.0004	0.000207667	0.000489189	0.000387636	0.0004305	0.0003342	0.00005	0.0004	0.0001	0.0004
standard deviation	0.0002	0.000106543	6.57596E-05	0.000187399	0.000137016	0.000219299	-	-	0	0.0002
minimum	0.000008/<0.001	0.00007	<0.0002	0.000088/<0.001	.000134/<0.0(	0.000017/<0.001	0.00005	0.0004	<0.0002/0.000	0.000008/<0.001
maximum	0.0005	0.000347	<0.001	0.0006	).00031/<0.00	0.0002/<0.001	-	-	0.0001	0.0006
# <detection limit (DL)	11	0	37	6	6	6	0	0	1	67
% <DL	65	-	100	55	75	60	-	-	50	72
# > guideline	0	0	0	0	0	0	0	0	0	0
% > guideline	-	-	-	0	-	-	-	-	-	-
# <DL & DL> guideline	0	0	0	0	0	0	0	0	0	0
% <DL & DL> guideline	-	-	-	0	-	-	-	-	-	-
maximum DL	0.001	-	0.001	0.001	0.001	0.001	-	-	0.0002	0.001
95th pecentile	0.0005	0.00034	0.0005	0.00055	0.0005	0.0005	-	-	0.0001	0.0005

Zinc										
Statistic	V1	VR	R7	FDU	W10	R6	Next Creek	Rose Cr Trib U/S K8	SF Rose Ck U/S Haul Rd	Combined
n	17	6	37	11	8	10	1	1	2	93
median	0.0025	0.00225	0.0025	0.0025	0.00375	0.0025	-	-	0.0012	0.0025
mean	0.005041176	0.0024	0.005905405	0.003863636	0.006425	0.00289	0.0019	0.001	0.0012	0.0048
standard deviation	0.008876096	0.001485934	0.007149046	0.002947973	0.007049164	0.002103146	-	-	0.00099	0.0064
minimum	0.0006/<0.001	0.0009	<0.001	0.0015/<0.005	0.0017/<0.005	0.0004/<0.005	0.0019	0.001	<0.001	0.0004/<0.001
maximum	0.038	0.0049	0.035	0.011	0.023	0.006	-	-	0.0019	0.038
# <detection limit (DL)	8	0	25	4	3	4	0	0	1	45
% <DL	47	-	68	36	38	40	-	-	50	48
# > guideline	1	0	1	0	0	0	0	0	0	2
% > guideline	6	-	3	0	-	-	-	-	-	2
# <DL & DL> guideline	0	0	0	0	0	0	0	0	0	0
% <DL & DL> guideline	-	-	-	0	-	-	-	-	-	-
maximum DL	0.005	-	0.005	0.005	0.005	0.005	-	-	0.001	0.005
95th pecentile	0.0164	0.00445	0.0186	0.009	0.01775	0.006	-	-	0.0018	0.0164

Notes:

Data set includes extra March and September to December sample data.

VR = Upper West Fork Vangorda

new FC data were added to FDU data set.

In cases where lowest detected concentration was less than lowest DL value, both have been reported under "minimum".

Background benchmarks based on combined reference station data are highlighted in green.

In cases where some values are <DL and DL is > guideline, computed 95th percentile may not yield good estimate of background (e.g., cadmium). Therefore, new 95th percentiles were computed (pale yellow) using only detectable values or <DL values provided DL was < guideline.

Table B.2: Summary statistics for potential toxicity modifying factors at mine-exposed stations (2005 - 2007).

Station V8							
Statistic	**pH (pH units)**	**Hardness (mg/L)**	**Calcium (mg/L)**	**Magnesium (mg/L)**	**Sodium (mg/L)**	**Alkalinity (mg/L)**	**DOC (mg/L)**
n	36	37	38	38	38	38	1
median	8.1	293	67.4	27.6	3.8	180	2.55
mean	8.1	280	66.1	27.8	3.7	172	-
standard deviation	0.25	99	23.0	10.2	1.1	55	-
minimum	7.3	112	26.9	10.6	1.5	77	-
maximum	8.5	449	120	44.3	5.4	255	-
Station X14							
Statistic	**pH (pH units)**	**Hardness (mg/L)**	**Calcium (mg/L)**	**Magnesium (mg/L)**	**Sodium (mg/L)**	**Alkalinity (mg/L)**	**DOC (mg/L)**
n	88	93	93	93	93	66	4
median	7.8	175	52.0	11.5	4.7	104	1.85
mean	7.8	216	63.5	13.9	6.1	111	1.73
standard deviation	0.26	146	43.4	9.08	4.9	45	0.68
minimum	7.2	54	16.1	3.24	1.4	33	0.80
maximum	8.4	812	243	49.8	27.7	240	2.40

Table B.3: Summary statistics for potential toxicity modifying factors at background stations (2005 - 2007).

Station V1							
Statistic	**pH (pH units)**	**Hardness (mg/L)**	**Calcium (mg/L)**	**Magnesium (mg/L)**	**Sodium (mg/L)**	**Alkalinity (mg/L)**	**DOC (mg/L)**
n	15	16	17	17	17	15	5
median	7.7	33	10.6	1.45	1.9	26	1.30
mean	7.7	37	11.6	1.78	1.9	31	1.18
standard deviation	0.40	14	4.36	0.78	0.41	11	0.29
minimum	7.0	13	4.31	0.64	0.87	15	0.70
maximum	8.3	59	18.7	3.07	2.7	54	1.4
VR (UWFV)							
Statistic	**pH (pH units)**	**Hardness (mg/L)**	**Calcium (mg/L)**	**Magnesium (mg/L)**	**Sodium (mg/L)**	**Alkalinity (mg/L)**	**DOC (mg/L)**
n	5.0	5	6	6	6	6	5
median	7.3	35	11.0	1.89	1.9	31	2.10
mean	7.5	36	11.1	1.98	1.8	33	2.06
standard deviation	0.63	6	1.63	0.35	0.31	6.5	0.78
minimum	6.9	29	8.85	1.66	1.4	26	1.20
maximum	8.4	44	13.4	2.62	2.23	45	3.10
R7							
Statistic	**pH (pH units)**	**Hardness (mg/L)**	**Calcium (mg/L)**	**Magnesium (mg/L)**	**Sodium (mg/L)**	**Alkalinity (mg/L)**	**DOC (mg/L)**
n	36	37	37	37	37	1	
median	7.8	103	30.6	6.48	2.4		
mean	7.8	100	29.9	6.18	2.4	157	
standard deviation	0.20	34	10.4	2.04	0.84		
minimum	7.3	28	8.41	1.62	0.77		
maximum	8.2	147	43.9	9.19	4.0		
FDU							
Statistic	**pH (pH units)**	**Hardness (mg/L)**	**Calcium (mg/L)**	**Magnesium (mg/L)**	**Sodium (mg/L)**	**Alkalinity (mg/L)**	**DOC (mg/L)**
n	11	10	11	11	11	5	4
median	7.4	13	3.69	0.76	1.8	15	1.95
mean	7.3	13	3.76	0.76	1.8	17	2.00
standard deviation	0.49	4	1.15	0.26	0.49	7.0	0.71
minimum	6.5	7	2.07	0.40	1.1	12	1.20
maximum	8.0	21	6.15	1.38	2.89	29	2.90
W10							
Statistic	**pH (pH units)**	**Hardness (mg/L)**	**Calcium (mg/L)**	**Magnesium (mg/L)**	**Sodium (mg/L)**	**Alkalinity (mg/L)**	**DOC (mg/L)**
n	8	7	8	8	8	2	2
median	7.6	50	16.4	2.12	1.8	49.8	3.15
mean	7.6	49	15.1	1.92	1.7	50	3.15
standard deviation	0.26	11	4.48	0.54	0.27	3.7	0.49
minimum	7.2	27	7.67	1.04	1.3	47	2.80
maximum	7.9	60	20.0	2.50	2.0	52	3.50

Table B.3: Summary statistics for potential toxicity modifying factors at background stations (2005 - 2007).

R6							
Statistic	**pH (pH units)**	**Hardness (mg/L)**	**Calcium (mg/L)**	**Magnesium (mg/L)**	**Sodium (mg/L)**	**Alkalinity (mg/L)**	**DOC (mg/L)**
n	10	10	10	10	10	10	4
median	8.1	158	45.1	10.8	2.1	140	1.00
mean	8.0	165	47.2	11.4	2.3	138	1.28
standard deviation	0.22	36	11.6	1.68	1.2	12	0.90
minimum	7.5	132	35.3	10.3	1.4	120	0.60
maximum	8.3	261	77.8	16	5.5	159	2.50
Next Creek							
Statistic	**pH (pH units)**	**Hardness (mg/L)**	**Calcium (mg/L)**	**Magnesium (mg/L)**	**Sodium (mg/L)**	**Alkalinity (mg/L)**	**DOC (mg/L)**
n			1	1	1	1	1
median							
mean			16.5	2.47	2.4	47	2.70
standard deviation							
minimum							
maximum							
Rose Cr Trib U/S K8							
Statistic	**pH (pH units)**	**Hardness (mg/L)**	**Calcium (mg/L)**	**Magnesium (mg/L)**	**Sodium (mg/L)**	**Alkalinity (mg/L)**	**DOC (mg/L)**
n	1	1	1	1	1	1	
median							
mean	7.8	77	25.0	3.57	2.9	78	
standard deviation							
minimum							
maximum							
SF Rose Ck U/S Haul Rd (USFR)							
Statistic	**pH (pH units)**	**Hardness (mg/L)**	**Calcium (mg/L)**	**Magnesium (mg/L)**	**Sodium (mg/L)**	**Alkalinity (mg/L)**	**DOC (mg/L)**
n	1	1	2	2	2	2	1
median			13.2	2.38	2.4	43	
mean	7.5	58	13.2	2.38	2.4	43	1.90
standard deviation			6.21	1.50	0.68	29	
minimum			8.82	1.32	1.9	22	
maximum			17.6	3.44	2.88	63	

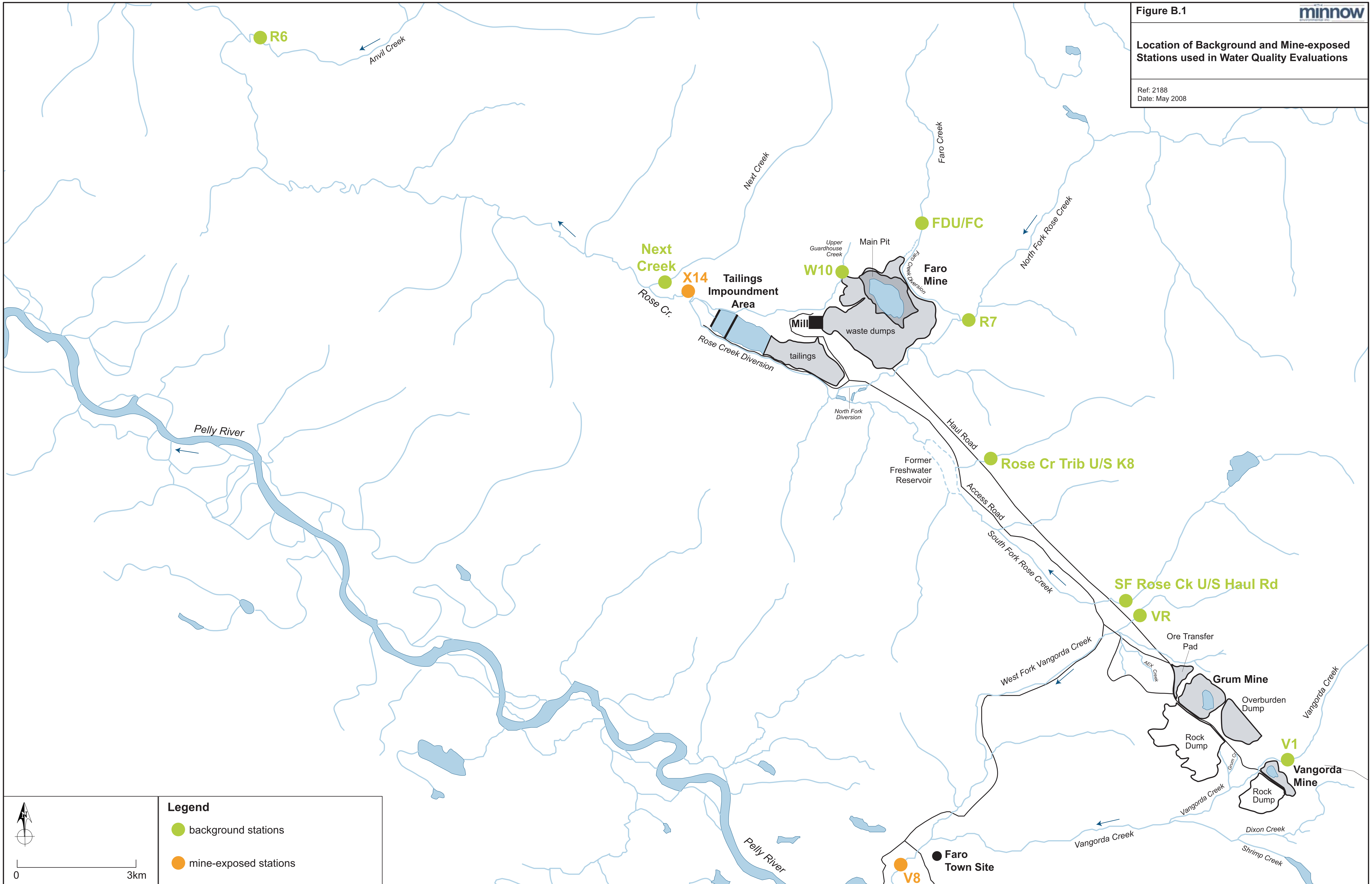

Figure B.1

Location of Background and Mine-exposed Stations used in Water Quality Evaluations

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