JERICHO EIA ISSUES
JERICHO MINE SITE
NUNAVUT

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Tahera Corporation
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Submitted by:

AMEC Earth & Environmental Limited
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1.0 INTRODUCTION

Reviews of the Jericho Mine EIS identified issues related to the impact of the mining operation on surface water quality that were not addressed or not completely addressed to the satisfaction of the reviewers. These issues were related to the effect of the mining operation on:

- The potential for increased concentrations of copper, nickel, cadmium, chromium, molybdenum and the effects of these increases on the aquatic biota;
- The potential for trophic transfer of metals and trace elements in the aquatic food chains;
- The effects of increased total dissolved solids concentrations on the aquatic biota of the area;
- The effects of increased nitrates due to blasting activities; and
- The methylation of mercury in the waters influenced by the mining operation.

These issues were addressed to the extent possible in the given time frame. In regard to trace elements and metals, some introductory statements are necessary within the mining context.

Disturbance of surface material (e.g. overburden, waste rock, etc.) and the physical reduction of large individual pieces of ore or resource-containing rock to relatively small sized particles increases the surface area exposed to weathering by natural forces by several orders of magnitude. Where water contacts this material, water's powerful dissolution chemistry ensures that at least some of the many elements that make up the ore or rock in question enter the dissolved state. This is the height of "mobilization" in elemental transfer from a static solid phase to a much more mobile liquid phase. In addition to the dissolved fraction, water also can support a certain amount of colloidal and suspended material derived from the extraction and processing of the resource-containing rock.

Just how these new additions to the many substances already dissolved and suspended in the natural waters of the area of the mine interact is a study in geochemistry of that particular ecosystem. How these same new additions interact with the organisms inhabiting the same ecosystem is a study in the biochemistry of that particular ecosystem.

Interactions of trace elements and metals with the organisms inhabiting an aquatic ecosystem take two basic forms:

- Adsorption onto the outer layer of material that forms the outer "covering" of the organism; and
- Absorption into the organism's intercellular and intracellular compartments where direct atomic and molecular chemical reactions take place with the atomic and molecular structures of the cell membranes and/or cell contents.

Adsorption to the outer covering (i.e. scales, exoskeleton, pellicle, cell wall, gelatinous sheath, etc.) does not generally interfere with the physiological processes of that particular organism, but my influence the dietary intake of trace elements and metals by the organisms in the next higher trophic level. Absorption, by contrast, almost ensures that some type of biochemical interaction will occur as the trace elements and metals make they way into the inner reaches of the cellular machinery of living organisms. Such interactions have to potential to cause effects so minor that they cannot be observed over long periods of time. The other end of the spectrum of possible effects is the death of the organism within a short time frame.

The wide variety of environmental conditions and chemical characteristics of trace elements and metals that allow absorption to occur are usually covered by the umbrella phrase “biological availability” or “bioavailability.” One of the major determinants of the extent to which elements are bioavailable is the form in which they exist in dilute aqueous solution (the case for most surface waters). These various chemical forms have been termed “species” by many researchers without regard to the definition of “species” in a biological context (subcategories of the genus level of biological nomenclature). Thus, a chemical species may be a free ion, or the same ion electro-chemically connected to another inorganic element or group of elements or an organic compound.

The amount of a trace element or metal actually available for absorption by a living organism may actually be only a fraction of the total amount of the element determined to be present in the water. Unfortunately, the current routine chemical analyses performed by commercial laboratories cannot distinguish between the bioavailable and non-bioavailable chemical forms of these elements. A common surrogate for the bioavailable fraction is the “dissolved” fraction. Even the distinction between dissolved and particulate is an operation expedient and is not based on true nature of the water in question and its associated dissolved, colloidal and suspended material.

The form of an element in aqueous solution is a direct result of the various conditions imposed upon the water matrix by the physical, chemical and biological forces present within the ecosystem. For example, the electro-chemical complex formed between the dissolved copper ion and dissolved organic compounds can have a dramatic and significant effect on the absorption of copper. The effect can be as dramatic as nullifying a normally lethal copper concentration to produce a totally non-lethal solution. By contrast, inorganic electro-chemical complexes of zinc, cadmium and nickel appear to be more important in reducing the availability for absorption than element-organic complexes.

The conditions under which these various inorganic and organic complexes exist are to a large extent determined by the hydrogen ion activity of the water itself (i.e. expressed as the parameter pH). Other competing ions in the aqueous solution also have the potential to accentuate or diminish the absorption capability of a particular form of trace element or metal. Other ecosystem conditions (e.g. anaerobic, aerobic, hypersaline, hyposaline, eutrophic, oligotrophic, etc.) also interact directly or indirectly with all the other factors (including those related to the organism itself) that ultimately determine the extent to which a particular trace element or metal can be absorbed by a living organism

Prediction of the effect of absorption of the trace elements or metals on the various biochemical processes within the biological species inhabiting an aquatic ecosystem is not as advanced a science as the geochemistry. As will become obvious in the discussion below, laboratory studies have been used as the main attempt to address the question of adverse biological effects of mobilized trace elements or metals on aquatic organisms. This seems only logical as the “scientific method” prevails as a paradigm for these investigations. Generally, the conditions that produced by these data are devoid of variability with the exception of the concentrations to which the test organism is exposed. In fact, the conditions used in these laboratory tests are generally designed to execute the maximum absorption of the element by the test organism when, in fact, this does not usually mirror conditions in the real world.

In a similar fashion, the protocols used by the Canadian government for the development of water quality guidelines (i.e. permissible concentrations of contaminants allowable in surface waters for various uses) are biased toward the lowest reported concentration (i.e. maximum absorption) causing an adverse effect. This approach is used even if the reported value is produced by a laboratory test that is totally devoid of real world conditions and may be a statistical outlier when compared with other toxicity reports for the same parameter.

In its defense, the government argues that water quality guidelines must be very conservative to protect the vast amount of water in this country as well as the equally vast variability of the water quality. The question raised at this point is when does a conservative value for the protection of aquatic life become over-protection with all of its associated misappropriation of resources? A very significant fact surrounding the development of numerical water quality guidelines is that these numbers do not take into account the presence of other toxic substances and their effects on the toxicity of the trace element or metal in question.

The various issues related to trace elements and other potential water quality problems of the proposed Jericho mine are discussed below within the context of the above.

2.0 THE INCREASED CONCENTRATIONS OF COPPER, CADMIUM, NICKEL, CHROMIUM, MOLYBDENUM AND THE EFFECTS OF THESE INCREASES ON THE AQUATIC BIOTA

2.1 Copper

Copper is one of those elements which is required for normal physiological functions in almost all plants and animals. However, the levels required for growth and biochemical sustainability are very close to those which can also have an adverse effect. Furthermore, it appears that naturally occurring concentrations may reach levels that affect sensitive species in some environments during some seasons. To confound these relationships, adaptation to increased copper levels by both plants and animals has also been documented.

The environmental factors known to affect the speciation and alter the bioavailability of copper to aquatic organism include pH, the presence of dissolved organic matter (especially humic and fulvic acids), the presence of inorganic carbon and phosphorus, exchange reactions between suspended sediments and water and the presence of other metals (especially the alkaline earth elements; group IIA of the periodic table).

In addition to bioavailability, the toxicity of copper may be affected by the physiological responses of the organism to changes in temperature, dissolved oxygen, pH and photoperiod. Also important is the metabolic condition or stage of the organism (life history stage, life cycle change, size, activity) and any previous exposure to copper or other metals.

The toxicity of copper to primary producers is usually not expressed in terms of an LC50 as is the case for aquatic animals. Instead, the concentration of copper that is effective in changing the process under examination are normally reported.

The concentration reported to cause the incipient inhibition of algal growth in the standard bioassay alga, *Selenastrum capricornutum*, is 50 µg/L. Complete growth inhibition was reported for the same species under the same conditions at 900 µg/L. The vast amount of algal toxicity data available span several orders of magnitude and emphasize the fact that copper toxicity is dependent on the chemical form of copper in the water and the resulting bioavailability. The special importance of algal copper toxicity rests in the fact that algae are the base of the aquatic food chain. Direct effects within the aquatic ecosystem result when copper toxicity reduces the amount of algae present (i.e. lethal effects). Indirect effects result when algal survival results in higher concentrations within living algal cells and the resulting consumption of those cells by higher trophic levels.

Data available for copper toxicity to freshwater insecta, mollusca and crustacean range from LC50 values of 39 to 2600 µg/L. Early life stages are more sensitive, generally, than the adult stage. For crustaceans, the group with the largest copper toxicity data base, LC50 values range from <10 to 9,000 µg/L.

In fish, copper toxicity is not only dependent upon the prevailing environmental conditions, but also on the particular species, stage of development and sex. Generally, the adverse effect of copper decreases from small to large fish, from young to old fish and from male to female. Among the various groups of fish, salmonids are generally considered the most sensitive to copper toxicity. Even in this group, however, lethal levels range from a 96 hr LC50 of 10 µg/L (in acid mine wastewater) to a 72 hr LC50 of 4,700 µg/L (water containing 10 mg/L glycine).

One over-riding fact of aquatic copper toxicity seems to prevail in all toxicity testing done to date. The most toxic copper form is the divalent ionic, uncomplexed form. As environmental conditions cause greater dissolved, complexed (both inorganic and organic) forms to appear, toxicity generally decreases. As the presence of antagonistic ions (e.g. calcium) increases, copper toxicity generally decreases. As the adsorption of copper to suspended or colloidal particulates increases, copper toxicity generally decreases.

Table 1 is a summary of selected published literature concerning chronic copper toxicity in the freshwater environment. Data from publications concerning the chronic toxicity of copper to freshwater fish were limited to salmonid species or species routinely used for toxicity testing. In addition to the information provided in the table below, some general attributes of low level copper concentrations in freshwater environments became evident during the review. These attributes can be characterized as follows

- Chronic copper toxicity varies with season in lentic environments; phytoplankton & zooplankton populations are more sensitive in the spring than in the summer and fall. This is probably related to the increased production of organic matter in the time frame from spring to fall and the subsequent binding of ionic copper to the dissolved or colloidal organic matter produced by the planktonic, photosynthetic organisms (Winner et al 1990).

- Chronic copper toxicity can vary significantly among various invertebrate species. The same concentration will allow one species to expand its population (perhaps a hormetic response), but will exterminate a closely related species in the same environment (More and Winner 1989). Some invertebrates can regulate the uptake and whole body concentration of copper (Borgmann and Norwood 1995).

- Chronic copper toxicity is greater at lower pH values compared to higher pH values. This is assumed to be related to the increased abundance of free copper ions and reduced organic an inorganic complex formation at lower pH values (Suedel et al. 1996)

- Chronic copper toxicity is influenced by the invertebrate food supply. Well-fed invertebrates are less susceptible to chronic copper toxicity than starved invertebrates (Koivisto et al. 1992).

TABLE 1: AQUATIC NO-EFFECT AND SUBLETHAL EFFECT COPPER CONCENTRATIONS REPORTED IN THE LITERATURE FOR FRESHWATER FISH* AND INVERTEBRATES

COPPER CONCENTRATION	DEFINED RESPONSE	ENDPOINT	ORGANISM	ENVIRONMENTAL CONDITIONS	REFERENCE
			FISH		
0.9 mg.L	Maximunm acceptable toxicant concentration	Survival growth & reproduction for 9 months	Fathead minnow	Calcium: 40-100 mg/L pH: 7.6-8.6 TDS: =189-641mg/L	Brungs et al. 1976
0.0094-0.0045 mg/L	No adverse effects concentration	Survival, growth & reproduction for 2 years	Brook Trout	Hardness: 40-46 mg/L pH: 6.9-8.8 Temp: 5-18^{o}C	McKim & Benoit 1974
0.33 mg/L	Incipient lethal level for fish not previously exposed to copper	Survival after 3 week exposure	Rainbow Trout	Hardness: 374+/-9.8 pH: 7.75 +/-0.04 Conductivity: 610 +/- 80 uS/cm	Dixon & Sprague 1981
0.0111 mg/L:	Maximum acceptable toxicant concentration	Reproductive Impairment after 9 months	Bluntnose Minnow	Hardness: 172-230 mg/L pH: 7.8-8.31	Horning & Neiheisel 1979
0.032 mg/L	Maximum acceptable toxicant concentration	Reproduction after 6 months of exposure	Fathead minnow	Hardness: 202 mg/L pH: 7.5-8.2	Pickering et al. 1977
0.055 mg/L	chronic sublethal effect	No increase in acid-soluble thiols or cysteine after 28 d exposures	Rainbow Trout	pH:7.8+/-0.1 Calcium: 41.6 mg/L	Lauren & Mcdonald 1987
0.024 mg/L	No effect level for total copper	Survival after 60	Steelhead Trout	pH: 7.0-7.3 Hardness: 18.0-29.2	Mudge et al. 1993

COPPER CONCENTRATION	DEFINED RESPONSE	ENDPOINT	ORGANISM	ENVIRONMENTAL CONDITIONS	REFERENCE
		days exposure			
0.022 mg/L	chronic sublethal effect	Olfactory Impairment after 37 wk exposure	Rainbow Trout	Hardness: 62 – 64 pH: 6.5 – 6.6 Condcutivity:200 uS/cm	Saucier et al 1991
0.0046 mg/L	chronic sublethal effect	reduced growth & elevated body Cu conc.after60 d exposure	Rainbow Trout	Hardness: 24.5 – 25.0 mg/L pH: 7.41 – 7.49	Marr et al. 1996
0.022 mg/L	No-effect level	Survival & growth over 78 d	Rainbow Trout	Hardness: 120 mg/L pH: 7.4-7.9	Seim et al.1984
INVERTEBRATES					
0.040 mg/L	Highest no-observed effect concentration	Survival and reproduction over a 72 hr period	*Daphnia magna*	Hardness: 130-160 mg/L pH: 8.2-9.5	Winner and Farrell 1976
0.025 mg/L	Lowest observed effect concentration	Growth after 10 days exposure	*Chironomus riparius*	Hardness: 151.2 mg/L pH: 6.8-7.2 Conductivity: 322.7 uS/cm	Taylor et al. 1991
0.021 mg/L	Maximum acceptable toxicant concentration	Growth after 10 days exposure	*Chironomus riparius*	Hardness: 151.2 mg/L pH: 6.8-7.2 Conductivity: 322.7 uS/cm	Taylor et al. 1991
0.017	No observed effect concentration	Growth after 10 days exposure	*Chironomus riparius*	Hardness: 151.2 mg/L pH: 6.8-7.2 Conductivity: 322.7 uS/cm	Taylor et al. 1991
0.030 mg/L	Minor delay in maturation	Growth & reproduction over 15 d exposures	*Daphnia pulex*	pH: 6.9-7.1 Alkalinity: 9.5 mg/L	Koivisto & Ketola 1995
0.004 mg/L	No observed effect concentration	Struct. & funct. over 32 wk exposure	Lab. microcosms derived from pond	Hardness: 200 mg/L	Hedtke 1984
0.020 – 0.043 mg/L	Lowest observed effect concentration	Microbial community sturct over 35 d exposure	Artificial substrate microcosms	pH: 8.1-9.5	Balczon & Pratt 1994
0.063 mg/L	No observed effect concentration	reproduction during 7 d exposure	*Ceriodaphnia dubia*	Hardness:94.1 mg/L pH: 8.2 Conductivity: 115.7 uS/cm	Belanger et al. 1989

* Note: References related to fish species were confined to salmonid species and species used for standardized toxicity

The information in Table 1 indicates that the no-effect and maximum acceptable toxicant concentration values for copper range over two orders of magnitude. There is an obvious negative correlation with the water hardness, therefore this fact should be taken into account if these numbers are used for establishing aquatic impacts in the Jericho surface waters. There are very probably other copper reactants present in the surface waters of the Jericho project which were not present in the various tests that produced the above data. Using these numbers therefore requires the assumption that these other reactants are not significant in terms of their effect on copper toxicity. The validity of this assumption is arguable.

2.2 Cadmium

It has been recognized for years that water hardness has a strong antagonistic effect on cadmium toxicity in freshwater environments. From the most recent literature regarding this phenomenon, it is apparent that calcium is the main cause of this antagonism. Cadmium is hypothesized to interfere with calcium metabolism by competitive inhibitory processes. Magnesium, the second major ion contributing to water hardness, does not reduce cadmium toxicity with the same effectiveness as calcium. While dissolved organics generally reduce cadmium toxicity, the uptake of an organic cadmium complex is extremely dependent on the exact type of organic compound forming the complex.

Cadmium uptake in freshwater fish occurs mainly via the gills. The rate of uptake is determined by the concentration and permeability of the various chemical forms of cadmium. Cadmium forms both soluble and insoluble complexes with the dominating anions in alkaline waters (i.e. OH^-, HCO_3^-, CO_3^{-2} and Cl^-). These inorganic complexes appear to be less available to fish.

Acclimation to increased concentrations of cadmium has been reported in the published literature (Alabaster and Lloyd 1982) for brown trout, coho salmon and rainbow trout previously exposed to sublethal levels cadmium at an earlier life stage. Acclimation of fish can be accomplished at the embryonic stage by exposure to levels of 1 to 4 µg/L for periods ranging up to 50 days. Acclimation was also demonstrated for rainbow trout at the alevin stage with a seven-day exposure to 10 µg/L resulting in the survivors' ability to withstand concentrations of 100 µg/L for seven days, compared to two days for the non-acclimated populations.

Table 2, below, presents selected chronic and sublethal effects data on cadmium exposures to freshwater fish and invertebrates.

TABLE 2: AQUATIC CHRONIC AND SUBLETHAL EFFECTS OF CADMIUM CONCENTRATIONS REPORTED IN THE LITERATURE FOR FRESHWATER FISH* AND INVERTEBRATES

CADMIUM CONCENTRATION	DEFINED RESPONSE	ENDPOINT	ORGANISM	ENVIRONMENTAL CONDITIONS	REFERENCE
FISH					
0.00018 mg/L	No-observed effect concentration	Survival & growth of early life stages after 60 d exposure	Atlantic salmon	Hardness: 13 mg/L	Sprague 1987
0.0018 mg/L	Lowest observed effect concentration	Survival & growth of early life stages after 60 d exposure	Atlantic salmon	Hardness: 13 mg/L	Sprague 1987
0.0036 mg/L	No-observed effect concentration	Survival & growth of early life stages after 60 d exposure	Rainbow trout	Hardness: 100 mg/L	Sprague 1987
0.005 mg/L	Lowest observed effect concentration	Survival & growth of early life stages after 60 d exposure	Rainbow trout	Hardness: 100 mg/L	Sprague 1987
0.0007 mg/L	No-observed effect concentration	Life cycle exposure	Rainbow trout	Hardness: 31 mg/L	Sprague 1987
0.0135 mg/L	No-observed effect concentration	Life cycle exposure	Rainbow trout	Hardness: 326 mg/L	Sprague 1987
0.001 mg/L	No-observed effect concentration	Survival & growth of early life stages after 60 d exposure	Brook trout	Hardness: 37 mg/L	Sprague 1987
0.003 mg/L	Lowest observed effect concentration	Survival & growth of early life stages after 60 d exposure	Brook trout	Hardness: 37 mg/L	Sprague 1987
0.0017 mg/L	No-observed effect concentration	Life cycle exposure	Brook trout	Hardness: 44 mg/L	Sprague 1987
0.0038	No-observed effect concentration	Growth of early life stages after 31 d exposure	Brook trout	Hardness: 45 mg/L	Sprague 1987
0.0038	No-observed effect concentration	Growth of early life stages after 33 d exposure	Brown trout	Hardness: 45 mg/L	Sprague 1987
0.0044	No-observed effect concentration	Growth of early life stages after 31 d exposure	Brook trout	Hardness: 45 mg/L	Sprague 1987
0.0002 mg/L	Reduced survival	Mortality after 18 mo exposure	Rainbow trout (juveniles)	Hardness: 112 mg/L	EPA 1985
0.140 mg/L	Death & deformity in 50% of test population	Mortality after 28 d exposure	Rainbow trout (embryo-larvae)	Hardness: 104 mg/L	EPA 1985

CADMIUM CONCENTRATION	DEFINED RESPONSE	ENDPOINT	ORGANISM	ENVIRONMENTAL CONDITIONS	REFERENCE
0.010 mg/L	Physiological effects	4 mo exposure	Rainbow trout (fingerlings)	Hardness: 320 mg/L	EPA 1985
0.10 mg/L	Reduced growth & survival	47 d exposure	Rainbow trout (fingerlings)	Hardness: 98.6 mg/L	EPA 1985
<0.005	Reduced survival	Mortality after 62 d exposure	Rainbow trout (embryo-larvae)	Hardness: 100 mg/L	EPA 1985
0.70 mg/L	50% mortality;	Mortality after 7 d exposure	Rainbow trout (larvae)	Hardness: 107 mg/L	EPA 1985
INVERTEBRATES					
0.005 – 0.010 mg/L	Maximum acceptable toxicant concentration	Mortality after 21 d exposure	*Daphnia pulex*	Hardness: 110 mg/L	EPA 1985
0.0007-0.001 mg/L	Reproduction	Impairment after 21 d exposure	*Daphnia magna*	Hardness: 45 – 82 mg/L	Wong 1987
0.00078 mg/L	Reproduction	impairment 50% after 20 d exposure	*Moina macrocopa*	Hardness: 45 – 82 mg/L	Wong 1987
0.055 mg/L	Mean life expectancy	Reduced by 50% after 157 d exposure	*Daphnia galeata mendotae*	Hardness: 45 – 82 mg/L	Wong 1987
0.0035 mg/L	Relative abundance of crustaceans in zooplankton	50% reduction after 24-31 d exposure	*Zooplankton population*	Hardness: 45 – 82 mg/L	Wong 1987

* Note: References related to fish species were confined to salmonid species and species used for standardized toxicity testing

As in the previous discussion of copper, the numbers in Table 2 reflect only the conditions of the tests conducted to produce the data. The validity of their use in determining acceptable cadmium concentrations for the surface waters of the Jericho Project is open to question. Although the presence of a copper-cadmium mixture may cause increased toxic action over and above the toxic reactions of each individual element, this is only one possibility. The other possibility is that the non-lethal copper “loading” of aquatic organisms might actually have a beneficial effect in terms of reduced cadmium toxicity (Brown 1975).

This antagonistic behaviour between cadmium and copper hypothesized in 1975 was concluded to exist by Finlayson and Verrue (1982). Their work used juvenile Chinook salmon in continual-flow acute toxicity tests of copper, zinc and cadmium both as individual toxicants and as mixtures. Copper:cadmium ratios of 1:0.028 and 1:0.038 demonstrated additive toxicity. Tests with the two metal mixtures, including zinc and copper as well as zinc and cadmium, indicated that changes in the amount of copper relative to the concentrations of both zinc and cadmium would change the type of interaction in mixtures of the three metals. This was subsequently supported by their tests. Copper-zinc-cadmium ratios of 1:12:0.08 and 1:3:0.02 exhibited an antagonistic response with the lower copper ration producing greater antagonistic behaviour.

Because the tests of Finlayson and Verrue (1982) indicated antagonistic behaviour for the three metal mixtures, safe concentrations for the chinook salmon that are estimated from tests of the individual metals should be suitable for mixtures as well. The safe concentration for dissolved cadmium using single metal toxicity tests had been previously identified as lying between 0.0004 and 0.0008 mg/L (Davis et al. 1979) for relatively soft waters (maximum 50 mg/L hardness).

2.3 Nickel

The number of reports of nickel toxicity in the scientific literature is small compared to elements such as copper and cadmium. This in itself is an indication of the frequency in the environment where nickel is considered a problem. Aquatic organisms are exposed to low-level natural background concentrations of nickel in freshwater and marine environments in the range 0.0054 to 0.086 mg/L.

In the aqueous environment, nickel commonly exists as the free, divalent nickel ion (Ni^{2+}) as a result of dissolution of a nickel salt or degradation of one of the many organo-nickel compounds or complexes. Nickel is an essential element for some fungi, but has not been found to be essential for higher plants or animals. Nickel compounds are generally very soluble in water with solubility decreasing with increasing $CaCO_3$ content .

In the water column, nickel is concentrated in the colloidal portions of the dissolved solids. Compared to the larger particulates, the colloidal fraction can “carry” a substantial portion of nickel in water. In sediments, nickel is thought to adsorbed onto particulates in a form which is neither exchangeable nor readily soluble in alkaline waters and thus not biologically available.

Table 3 below summarizes selected toxicity data for laboratory studies of nickel. In general, concentration ranges can be established at which certain effects are observed in the laboratory setting. However, differences among species or life stage sensitivity and environmental conditions may result in an overlap of some ranges. The concentration ranges are:

<u>Less then or equal to 1 mg/L</u>: Values in this range reduced hatching of eggs per spawning incident in fathead minnows exposed to .073 mg/L, but did not affect growth and survival of adults. Reduced growth in four algal species was reported at 0.1 to 0.7mg/L. Background levels of nickel in many freshwater environments are also found in this range.

<u>1.0 – 10 mg/L</u>: This range includes the threshold of acutely toxic effects in soft water for several species including freshwater zooplankton, fathead minnows and several aquatic insects.

<u>10 – 100 mg/L</u>: This range includes acutely toxic values for many freshwater organisms including Atlantic silverside, winter flounder, channel catfish, killifish, trout, sunfish, copepods and aquatic insects.

100 – 1000 mg/L: This range includes values which are lethal at 96 hours exposure in several marine organisms due to the higher concentration of antagonistic ions (e.g. calcium) in this type of environment.

TABLE 3: CONCENTRATIONS REPORTED IN THE LITERATURE FOR NICKEL TOXICITY TO FRESHWATER FISH* AND INVERTEBRATES

NICKEL CONCENTRATION	ACUTE/CHRONIC	ENDPOINT	ORGANISM	ENVIRONMENTAL CONDITIONS	REFERENCE
FISH					
0.73 mg/L	chronic	Reproductive impairment	Fathead minnow	Hardness: 210 mg/L Temp: 25°C	Pickering 1974
0.325 mg/L	chronic	Mortality	Rainbow trout	Hardness: 50 mg/L	EPA 1980
0.134 mg/L	chronic	No-effect level after 38 d exposure	Rainbow trout (pre-swimup larvae)	Hardness: 27-39 mg/L	Nebeker et al. 1985
0.7 mg/L	chronic	Significant adverse effects on growth, 38 d exposure	Rainbow trout (pre-swimup larvae)	Hardness: 27-39 mg/L	Nebeker et al. 1985
0.003 mg/L	chronic	Lethal to 1% of the population; 32 d exposure	Rainbow trout (embryo & larvae)	Hardness: 195 mg/L	Birge 1978
160 mg/L	acute	50% Mortality	Rainbow trout	Hardness: 42 mg/L Temp: 12°C	Willford 1966
270 mg/L	acute	50% Mortality	Brook trout	Hardness: 42 mg/L Temp: 12°C	Willford 1966
5.9 – 7.9 mg/L	Acute	50% Morftality	Fathead minnow	Soft water Temp: 25°C	Pickering & Henderson 1966
51.6 – 59.3 mg/L	Acute	50% Morftality	Fathead minnow	Hard water Temp: 25°C	Pickering & Henderson 1966
75 mg/L	Acute (48 hr)	50% Mortality	Lake trout	Not provided	-EPA 1981
INVERTEBRATES					
0.130 mg/L	Chronic (3 wk)	50% Mortality	*Daphnia magna*	Lake Superior water	NRCC1981
0.095 mg/L	Chronic(3 wk)	50% Reproductive impairment	*Daphnia magna*	Lake Superior water	NRCC1981
0.51 mg/L	Acute (48 hr)	50% Mortality	*Daphnia magna*	Lake Superior water Organisms unfed	NRCC1981
1.12 mg/L	Acute (48 hr)	50% Mortality	*Daphnia magna*	Lake Superior water Organisms fed	NRCC1981

* Note: References related to fish species were confined to salmonid species and species used for standardized toxicity testing

In consideration of all nickel toxicity data reviewed by Birge and Black (1980), it appears that reproduction of most aquatic species would not be appreciably impaired by dissolved nickel concentration of 0.010 mg/L.

2.4 Chromium

Chromium is a relatively common element with an average concentration of 100 mg/lkg in the earth's crust. It is in the 21st position of the ranking of the most commonly occurring elements in the earth's crust. Although there are relatively few natural ore deposits, many rocks and sediments contain 70-90 mg/kg of chromium with a total range of 5 to `1500 mg/kg (Bowen 1979). The element exists in all oxidation states from –2 to +6, but only the trivalent and hexavalent compounds are of practical and environmental importance (Gauglhofer 1991).

In natural water, the trivalent Cr^{+3} ion forms very stable complexes with negatively charged organic or inorganic substances to the point that significant amounts of the Cr^{+3} ion alone are not available in normally occurring natural water containing dissolved or particular matter of an organic or inorganic nature. Even if anions are not present, chromium can react with water to form colloidal hydrous oxides. Below pH 5, these colloids takes the form of a stable, negatively charged hexaquo complex and above pH 9 they are soluble negatively charged hydroxides. Most trivalent chromium compounds are soluble only in low pH water (i.e. <pH 5).

In natural water, hexavalent (Cr^{+6}) chromium exists almost entirely in the form of oxo anions (i.e. CrO_4^{-2} & $Cr_2O_7^{-2}$), thus its aqueous environmental chemistry is dramatically different from that of the trivalent form. In dilute aqueous solution (i.e. <1 mg/L), the predominant form of hexavalent chromium is dissolved CrO_4^{-2} which does not complex with other anions. This anionic form is a strong oxidizing agent, especially under acidic conditions and has a strong tendency to react with oxidizable organic substances resulting in the formation of Cr^{+3}. If, however, the concentration of oxidizable substances in low, then the hexavalent oxo anion may persist for substantial periods of time.

Concentrations of chromium in natural waters that have not been affected by waste disposal are commonly less than 10 µg/L. Natural waters contain only the trivalent form of chromium. The presence of the hexavalent form is almost always due to anthropogenic activity. The median value for public water supplies in the U.S. was reported to be 0.43 µg/L (Dufor and Becker 1964). North American rivers were reported to have an average concentration of 1.4 µg/L (Kharkar et al. 1968). Concentrations in lake water have been reported to range from 0.1 to 1.7 µg/L (Richard and Bourg 1991). Concentrations of dissolved chromium in uncontaminated groundwater have been reported as great as 200 µg/L in U.S. desert southwest due to the dissolution and oxidation of chromium-bearing minerals by oxygen bearing recharge water (Robertson 1975).

The dynamics of chromium in lakes varies with the chemical changes which accompany the seasons and the trophic regime of the lake. Lakes that have anoxic waters for at least part of the year due to their increased productivity (e.g. eutrophic lakes) generally show that Cr^{+6} is removed from the water column by reduction or adsorption. Reduced Cr^{+3} is present in the colloidal form in these lakes and is not oxidized to Cr^{+6}.

However, the presence of chromium in oligotrophic lakes may have the potential for oxidation of Cr^{+3} to Cr^{+6}. A study of chromium in a Swiss alpine lake at an altitude of 2,496 m which was not influenced by anthropogenic activities found concentrations of dissolved chromium reaching 2.5 µg/L with over 90% in the form of Cr^{+6}. The lake was located in a serpentinite complex and received water from rainfall, snowmelt and as drainage from ultramafic rock known for its high chromium content (Johnson and Sigg 1992). The similarity between lakes located at high elevations and lakes located at high latitudes is such that the normal chromium cycling in temperate, low latitude, eutrophic lakes may not be relevant to the cycling of chromium in the high latitude oligotrophic lakes of the Jericho project. The rates of chromium oxidation and reduction under various environmental conditions have yet to be detailed and explained (Richard and Bourg 1991). This seems especially true for oligotrophic environments. Collection of water samples utilizi1ng ultra-clean techniques indicate that Canadian pristine lakes and rivers contain from <1 to 5 µg/L total chromium (CCME 1999).

The aquatic toxicity of Cr^{+3} and Cr^{+6} are dramatically different as is evident by their respective federal Canadian guideline values. The freshwater aquatic guideline for Cr^{+3} is 8.9 µg/L while the guideline for the more toxic Cr^{+6} form is 1.0 µg/L. These guideline values were derived from chronic exposure laboratory tests conducted under conditions that maximized the bioavailability of the chromium in solution. In natural settings, a measurement of total chromium (i.e. both Cr^{+3} and Cr^{+6}) in an unfiltered water sample may produce values that exceed the guidelines without adverse effect to the ecosystem biota (CCME 1999).

The maximum observed concentration of chromium in kimberlite ore samples was reported to be 1,500 mg/kg and the average concentration in simulated fine processed kimberlite was reported to be 995 mg/kg in comments from the review of the Jericho Environmental Impact Statement by the provincial Department of Sustainable Development. These values were characterized as being at the extreme high end of typical ranges in background soils and freshwater sediments. However, what is not known is how these concentrations translate into water concentrations and the particular form taken in the aqueous environments at the Jericho project site.

The kimberlites are a group of volatile (CO_2) rich, potassic ultrabasic rocks with variable composition megacrysts in a fine grained groundmass. The megacrysts can contain several minerals, including chromite ($FeCr_2O_4$) in which the chromium is in the trivalent or reduced state. Thus, the potential for chromium to exceed its current concentration in the surface water of Lake D10 (<0.5 µg/L) depends largely on the solubility of chromite as kimberlites are processed. The 71st edition of the CRC Handbook of Chemistry and Physics reports chromite to be "insoluble" in both cold and hot water. If, however, minute amounts of Cr^{+3} were released as a result of the kimberlite processing, the strong affinity for negatively-charged anions and colloids make it relatively immobile and non-toxic (i.e. not biologically available). One of these negatively-charged ions for which Cr^{+3} has a strong affinity are oxygen-binding groups, such as hydroxyl ions with which it forms nearly insoluble compounds such as $Cr(OH)_3$. Soluble Cr^{+3} concentrations in equilibrium with $Cr(OH)_3$ are <0.05 µg/L in water of pH 6 and $<5 \times 10^{-12}$ µg/L at pH 8. It solubility increases significantly at pH values less than 5.5 and slightly at pH values greater than 8 (James 2002).

With relation to the Jericho mine, the question to be addressed is: What environmental conditions need to be present in order for the very low levels of Cr^{+3} to be converted to the much more mobile, more bioavailable and more toxic Cr^{+6}?

To convert Cr^{+3} to Cr^{+6} requires an oxidizing environment. The extent of Cr^{+3} oxidation the water and sediments of lakes similar to Lake D10 would be strongly dependent upon four factors:

- The original form of Cr^{+3};
- The oxidizing potential for Cr^{+3};
- The reduction potential for Cr^{+6}; and
- The pH of the environment as the master variable.

These four factors can be ranked and then added to obtain a potential chromium oxidizing score (PCOS). The dynamic, undulating nature of chromium oxidation-reduction equilibria in Lake D10 will be dominated by pH. Higher pH values will promote the oxidation of Cr^{+3}; lower pH values will promote the reduction of Cr^{+6}. The presence of manganese hydroxides will promote oxidation of Cr^{+3}, whereas the presence of organic compounds will promote the reduction of Cr^{+6} (James 2002). The Lake D10 environment (including water, sediments and the biological community) will be subject to a number of changes as a result of its conversion to the PKCA in both a spatial and temporal time frames that cannot be accurately predicted at this time. The best management system for controlling any chromium-related issue resulting from the mining activity at the Jericho site would be to promote those en0vironmental conditions favouring chromium reduction and discouraging those environmental conditions favouring chromium oxidation. Much is unknown about chromium cycling in lakes and each lake environment has the potential for its own particular idiosyncrasies. This is especially true if major changes are introduced in the chemistry and biology of the lake via the conversion to a waste treatment facility. However, if a chromium management system is put in place, the chromium issue should become insignificant.

The current prognosis for the water of the PKCA is for a pH between 8 and 9 and due to chemical reagents used. Any chromium management system should attempt to control the pH of the PKCA water to a maximum of pH 8. A key element of this strategy would be the introduction to the PKCA of organic material in the form of hay or straw. The introduction of hay or straw over the water surface will act to reduce suspended solids concentrations by microbial alteration of the zeta potential as well as increase the organic matter in the lake. As the combination of suspended solids and degrading hay sink to the lake bottom, the sediment-water interface will become a habitat suitable for sulfate reducing bacteria (SRB). The generation of sulfide by this microbial community will ensure the availability of negatively charged ions and a reducing environment so that the equilibrium kinetics of chromium cycle in the PKCA is forced toward the formation and maintenance of sediment-seeking, trivalent chromium compounds. Additions of phosphorus may also be necessary for the maintenance of the SRB community and for the continued input of organic matter to the PKCA via photosynthesis by algal cells or emergent aquatic macrophytes.

If these measures fail to eliminate hexavalent chromium from the surface waters of the PKCA, then the toxicity of hexavalent chromium becomes an issue. Sauter et al (1976) attempted to assess the “no effect” level of waterborne hexavalent chromium to rainbow trout eggs and fry in soft water (hardness 30-46 mg/L) at pH values from 6.7 to 7.4 and 60 d exposures. The maximum acceptable toxicant concentration under these conditions was stated to be between 0.051 and 0.105 mg/L.

2.5 Molybdenum

Molybdenum is a rather rare element with an occurrence, on average, of about 1.2% in igneous rocks and 5.5% in sedimentary rocks. Its oxidation states range from +3 to +6, but the most common solid and aqueous forms of the element have oxidation states of +4 and +6. Under oxidizing conditions, the predominant state is Mo^{+6} that forms molybdate ions in water. Below pH 2, the predominant molybdate form is the dissolved, undissociated H_2MoO_4 (aqueous). The $HMoO_4^{-1}$ ion is predominant between about pH 2 and pH 5. Above pH 5, the dominant form is MoO_4^{-2}. Many of the metallic elements have molybdates of low solubility. Molybdate ions tend to polymerize and an extensive series of such forms can exist, depending on pH and total molybdenum concentration. The polymers do not occur as major components of molybdenum in water unless the solution exceeds 10 mg/L.

Molybdenum has a relatively high geochemical mobility as it has a tendency to enter into water solution under conditions normally present at the surface of the earth (Chappell 1975). The amount of molybdenum in water solution has been reported to be controlled by a number of factors, including precipitation reactions with common metals as metal molybdates. For example, ferrous molybdate solubility can limit molybdate concentrations to less than 100 µg/L in relatively iron-rich waters at a pH less than five. This type of solubility control is less effective in oxidizing waters of higher pH due to the lower iron solubility under such conditions. Studies of molybdenum in streams draining the Climax area of Colorado attributed downstream attenuation of molybdenum in water to adsorption by amorphous ferric hydroxide. This has been verified by laboratory studies (LeGendre and Runnells 1975).

The solubility product of calcium molybdate is $10^{-8.7}$(Mannheim 1978), which suggests that surface waters containing substantial concentrations of dissolved calcium should not contain large concentrations of dissolved molybdenum. This has not as yet been proven and conflicting reports on the aqueous forms and behaviour of molybdenum appear in the published literature (Smith et al. 1977).

The environmental mobility and availability of molybdenum is strongly influenced by pH and oxidation-reduction conditions with increased mobility and availability in oxic systems compared to anoxic systems. Molybdenum is greatly enriched in sediments with high organic content in anoxic conditions and in sediments that have overlying water concentrations with low dissolved oxygen concentrations. Observations such as these suggest an efficient transfer mechanism for molybdenum from the dissolved phase to the particulate phase in anoxic environments. Decreases in dissolved molybdenum concentrations in the presence of hydrogen sulfide generally have been attributed to precipitation of molybdenum sulfide or co-precipitation of molybdenum with iron sulfide (Smith 1997).

Extensive compilations of water quality data generally show that river and lakes not extensively influenced by pollution sources have less than 1 µg/L elemental molybdenum. Higher concentrations, however, are not uncommon. Surface water concentrations in the molybdenum mining areas of Colorado can be as great as 3,800 µg/L (Mannheim 1978). The median value for public water supplies has been reported to be 1.4 µg/L (Durfor and Becker 1964) although molybdenum concentrations exceeding 100 µg/L have been reported for public drinking water reservoirs serving the Denver, Colorado area (Barnett et al. 1969). About one third of the 1500 samples of surface water from various locations in the continental United States had detectable molybdenum concentrations with a maximum concentration of 1,100 µg/L and a mean concentration of 68 µg/L (Kopp and Kroner 1968).

The aquatic toxicity of dissolved molybdenum can be generally characterized as "moderate" as most acute toxicity data falls in the range 10^5 – 10^6 µg/L. Chronic toxicity has been observed in the range 10^3 – 10^5 µg/L (CCME 1999). The Canadian federal (interim) guideline is 73 µg/L based on one non-standard 28-day static water (with 24 hr replacement) toxicity test. The British Columbia guidelines are 1,000 µg/L (average value of 30 days) and 2,000 µg/L (maximum value not to be exceeded) (L.G. Swain, 1986). The British Columbia guidelines are based on standard acute toxicity tests and are more environmentally realistic than the federal guideline.

In the context of the Jericho mine, releases of molybdenum would probably be attenuated only by dilution with few other mitigating factors available (i.e. sorption, precipitation, biological uptake etc.). Because of the very dilute nature of the area waters, the antagonistic effects of water hardness against molybdenum toxicity will be nil. Certainly, concentrations of molybdenum exceeding 1,000 µg/L entering receiving waters should be avoided. Predicted concentrations of molybdenum in the PKCA effluent are about 6.6 µg/L compared to a background concentrations of <0.05 to < 0.1 µg/L. According to the available toxicity data, a molybdenum concentration of 6.6 ug/L will not be detrimental to the receiving waters or the aquatic life therein. By contrast, the possibility exists that since molybdenum is an essential requirement for the functioning of the enzyme nitrogen reductase (required for biological fixation of atmospheric nitrogen), the increase in the amount and availability of molybdenum may result in a shift in the algal community of Lake D10 with nitrogen fixing blue-green algae becoming more prominent in the PKCA water. This same effect, although on a much smaller scale, may also be observed in Lake C3. These changes are contingent upon the assumption that molybdenum is an element limiting growth in the lakes.

Biological uptake of molybdenum in the PKCA (Lake D10) via the phytoplankton and zooplankton communities of the ecosystem could well be a significant mitigating activity if sufficient nutrients (nitrogen and phosphorus) were made available in order for these two ecosystem components to flourish. However, their current biomass as described in the Environmental Impact Assessment for the Jericho Project would not appear to allow them to be a significant factor in removing molybdenum from the water column.

2.6 Acclimation of Fish to Metals

Although not an issue raised by the reviewers of the EIA, the phenomenon of acclimation has not been discussed in relation to the potential for surface water metal contamination at the Jericho site. This is a significant omission when issues of aquatic metal contamination are brought forward. Acclimation to metals is usually operationally defined as an increase in resistance or tolerance to a lethal challenge after a period of pre-exposure to sublethal levels.

Fish have considerable ability to acclimate to a variety of environmental stressors such as temperature and dissolved oxygen. Such acclimation, in strict physiological terms, involves adjustment of processes such as enzymatic reaction rates. Acclimation to metals may also occur by other mechanisms, such as exclusion or excretion of metals and by adjustment of membrane permeability to metals and/or the production of the protein metallothioein. Significant acclimation to cadmium has been demonstrated for the rainbow trout and fathead minnows to environmental mixtures of copper, cadmium and zinc (Spry and Wiener 1991). If the levels of the most pronounced metals entering the surface waters of the Jericho mine area were slowly raised, acclimation could be a major factor in the survival of the impacted fish populations.

3.0 THE POTENTIAL FOR TROPHIC TRANSFER OF METALS AND TRACE ELEMENTS IN THE AQUATIC FOOD CHAINS.

There are literally mountains of published papers on this general topic. The very first thing which must be accomplished prior to any discussion of this topic is to come to an agreement concerning the terminology used in the published literature as it is not uniform. The following definitions of terms apply to most of the published research.

Bioconcentration: the uptake of metals/trace elements (M/TR) elements by aquatic organisms where the sole contaminant source is water;

Bioaccumulation: the uptake of M/TR from both dietary and water sources;

Biomagnification: the process of both bioconcentration and bioaccumulation that result in increased tissue contentrations of M/TR as it passes through two or more trophic levels;

Trophic transfer: the transport of M/TR between two trophic levels; and

Trophic transfer coefficient (TTC): defined by Suedel et al. (1994) as the concentration of a M/TR in a consumer tissue divided by the concentration of the M/TR in food (i.e. the preceeding trophic level.

Given these definitions, Biomagnification occurs when tissue concentrations of M/TR increase between two or more trophic levels. Generally, animals and plants living in M/TR-contaminated environments will generally have higher concentrations of the particular M/TR than animals and plants living in uncontaminated environments (i.e. bioconcentration and bioaccumulation). However, because of the manner in which the M/TR is bound to the tissue of the prey organism (or plant) and also due to the internal environment of the animal digestive system, the M/TE may not be efficiently absorbed by the animal or not absorbed to the extent of providing sufficient amounts of M/TR to the next trophic level. Thus, M/TR can be transferred across one trophic level, but not two levels.

Suedel et al (1994) gives the following status of M/TR in aquatic environments

Arsenic: Biomaginification of arsenic in aquatic food chains is dependent on trophic level with TTC values ranging from 0.1 to 20.1. These data suggest that arsenic does not biomagnify in either producer or primary and secondary consumer-level organisms (forage fish), but may biomagnify in tertiary consumers(predatory fish).

Chromium: Chromium does not magnify in aquatic systems.

Copper: Copper usually does not biomagnify in aquatic systems, however there are conflicting studies. The phenomenon of “luxury uptake” of copper among some marine organisms may be mistaken for trophic transfer.

Cadmium: There is little evidence to suggest that cadmium biomagnifies in the aquatic environment. Cadmium has been found to bioaccumulate in freshwater detritovores, but there is no apparent relationship between tissue concenrtration and trophic level.

Lead: A review of aquatic lead uptake studies demonstrated little or no evidence for lead biomagnification in aquatic food webs.

Mercury: Mercury has the potential to biomagnify in aquatic systems due to the relative ease with it becomes associated with an organic ligand.

Zinc: Laboratory and field studies indicate that zinc has the potential to transfer, but not biomagnify, though aquatic food webs.

Other M/TR: Based on limited evidence, barium, boron, beryllium, cobalt, manganese, molybdenum and nickel do not biomagnify in aquatic food webs.

The biomagnification of mercury is associated with its organic forms (e.g. methylmercury, dimethymercury, phenylmercury, etc.) Likewise, where arsenic is present in an organic form, it too has much greater potential to pass from lower to higher trophic levels.

4.0 EFFECTS OF INCREASED DISSOLVLED SOLIDS ON AQUATIC BIOTA

It is a fact that an increase in the total dissolved solids (TDS) concentration of the water in the PKCA will occur. The only unknowns associated with this fact are the magnitude of the increase (which can be estimated) and the effect of the increase on the aquatic ecosystems of the PKCA and the water body (Lake C3) receiving the PKCA effluent.

The report by Golder Associates on the effect of increased TDS on the aquatic communities of another sub-arctic lake (Snap Lake) gives a good description of the various aquatic toxic concentrations of the major ions making up TDS (especially calcium and chloride). An important corollary issue not discussed in the Snap Lake report is the effect of the change in relative amounts of the major ions (i.e. the ratios of the major ions) and the effect of such changes on the resident biota of Snape Lake. Wetzel (1975) gives examples of how changes in the ratio of monovalent to divalent ions can produce significant changes within the composition of the phytoplankton community.

Of the various communities that are included in the PKCA and Lake C3 ecosystems (e.g. phytoplankton, periphyton, zooplankton, benthos, fish), the very first organisms to be influenced by the increases in TDS will be the algal cells within the phytoplankton community. This due to the fact that reproduction (especially via mitotic cell division) is much faster in these unicellular organisms then in the multi-cellular organisms which make up the most obvious members of the other communities. Changes in the phytoplankton community in terms of species and morphology tend to produce a "ripple effect" in the higher trophic levels of the lake ecosystem. Generally, the greater the change in phytoplankton species and morphology is directly related to a greater ripple effect. Such a ripple effect may be due to the change in morphology of the dominant algal species from a small, individual coccoid shape to a multicellular coneobium. This may disrupt the feeding of the zooplankton to the point where dramatic decreases occur in the density of the zooplankton community. Of course, the opposite is also a possibility,

Calcium, for example, has been implicated in the growth and population dynamics of freshwater algae. The calcium requirement of higher plants has been identified and the implication is strong that calcium also positively affects the growth of green algae (phylum Chlorophyta). The phytoplankton community of Lake C3 has been reported to consist of 9.9% Chlorophyta and 27% blue-green algae (phylum Cyanophyta). The rather large percentage of blue-green algae in this environment could possibly be due to the ability of blue-green algal species, in general, to take up phosphorus over and above that required for normal physiological functions and store it within the cell. In addition, the dense, gelatinous matrix within which the cells exist (especially the genus *Aphanocapsa*) may permit the fixation of diatomic nitrogen thus allowing it a nitrogen source (along with the stored phosphorus) for growth and reproduction that is unavailable to other algal groups such as the Chlorophyta. The introduction of greater calcium concentrations may stimulate the growth of the Chlorophyta and, in some fashion, reduce the physiological advantages possessed by the Cyanophyta. If the resulting Chlorophyta species are more "edible" by the zooplankton community, significant changes in structure of the next highest trophic level could be expected.

While the future concentrations of major ions in the PKCA can be estimated mathematically, there is no ecologically equivalent calculation that will predict the composition of the phytoplankton community or the ripple effects to be experienced by the higher trophic levels. There are simply too many factors that contribute to the dynamics of communities within ecosystems for any exact forecast to be sufficiently descriptive of the changes that will take place.

Without an much better understanding of structure and function of the PKCA and Lake C3 ecosystems, it is impossible to describe the exact nature of the effects resulting from mining activity. The only predictions that can be made are:

- Changes in the amounts and relative portions of major ions that constitute the total dissolved solids component of the PKCA and Lake C3 surface waters will result in changes in their respective ecosystems;

- Whether these changes are interpreted as "positive" or "negative" depends on human interpretation; and

- Once the inputs from mining activity have ceased, the ecosystems will eventually return to a condition similar or equivalent to their pre-mining condition.

5.0 THE EFFECTS OF INCREASED NITRATES DUE TO BLASTING ACTIVITIES

The increase in the concentration of nitrate in surface waters associated with the mining activity at the Jericho Project is concern from a aquatic toxicity and human health (i.e. drinking water) perspective. It is a fact that a nitrogen cycle currently exists in Lake D10, but the particular dynamics of this nitrogen cycle are not known. That is, the rates at which nitrate is converted to other compounds (e.g. organic nitrogen, gaseous inorganic oxides of nitrogen and to diatomic nitrogen) are unknown. Thus, the input of more nitrate from mining activities may have a dramatic effect on the existing equilibrium dynamics of nitrogen in what will become the PKCA. The impact of the increased nitrate inputs to the existing nitrogen cycle in Lake D10 was not addressed in the EIS. If the PKCA is intended as a treatment facility, then preparations should be made to enhance those particular facets of the currently existing nitrogen cycle in Lake D10 which convert nitrate to other less toxic compounds.

Conversion to organic nitrogen is probably the simplest pathway and can be accomplished by creating optimal conditions for algal growth. Phosphorus additions may be necessary to maximize this pathway. Pathways for the conversion of nitrate to gaseous inorganic oxides (N_2O and N_2) can also be enhanced by the initial addition of organic substances such as straw or hay. Increased primary production as a result of enhancing the conversion of nitrate to organic nitrogen will also assist in the creation of conditions for biological denitrification. Strictly chemical denitrification methods are also commercially available from a number of sources.

6.0 THE METHYLATION OF MERCURY IN THE WATERS INFLUENCED BY MINING ACTIVITY

Methylmercury in aquatic ecosystems and its accumulation in aquatic biota, especially fish, is obviously correlated with methylmercury production in the same ecosystem. The question related to the Jericho Mine is whether changes in the water quality of the lakes impacted by the mining activity will have increased methylmercury production in excess of the production that occurs naturally.

To begin with, a number of water quality variables have been correlated with the amounts of methlymercury in northern lakes. Dissolved organic carbon (DOC), dissolved iron and dissolved manganese were found to be positively correlated with the concentration of methylmercury in the water (i.e. the more dissolved organic carbon, iron or manganese, the more methylmercury). The pH of the lake water and its dissolved oxygen concentration were negatively correlated with the amount of methylmecury in the water (i.e. the lower the pH and dissolved oxygen, the higher the amount of methylmercury (Watras et al. 1995).

Initially, it would seem that any impacts of mining activity or remedial activities that would raise the water pH and dissolved oxygen concentration and also decrease dissolved iron dissolved manganese and dissolved organic carbon would be beneficial in reducing the potential for methylmercury production. While these measures would certainly have the potential to reduce methylmercury production in a given lake, the mercury cycle in lakes is much more complex. Mercury methylation in lakes is counter-balanced by mercury demethylation with numerous environmental factors influencing both rates of methylmercury production and methylmercury degradation concurrently.

Surface sediments, with their much higher microbial populations compared to the overlying water, are the main locations for methylmercury production and degradation in almost all lakes (Gilmour et al. 1998, Marvin-DiPasquale et al. 1998). Mercury methylation can be accomplished by a number of microbes and can even occur as the result of abiotic chemical reactions unaided by biochemical systems. One of the major groups of organisms responsible for mercury methylation are the sulfur reducing bacteria (SRB) which normally reside in their greatest numbers at the sediment-water interface. Since the activity and density of the SRB are strongly influenced by the amount of sulfur present, there is a strong tie between methylmercury production and the lake sulfur cycle. However, the exact impact of sulfur on the mercury cycle varies with the form of sulfur and its concentration. From the studies published thus far, several points stand out relative to the production of methymercury in a northern lake sediment.

- Methylmercury concentration is inversely correlated with sulfide concentration in more nutrient rich environments (i.e. higher sulfide concentrations mean lower methylmercury concentrations);
- Pore water sulfide can be used to predict methylmercury concentrations in surficial sediments;
- While increasing sulfide concentratons tend to inhibit mercury methylation, increasing sulfate concentrations tend to stimulate mercury methylation;

In order to enter the food web, the methylmercury must leave the sediment when it was produced. This flux of methylmercury out of the sediment is thought to be the result of a variety of mechanisms including:

- The normal activities of benthic organisms inhabiting the sediment-water interface and subsequently mixing the sediment pore water with water immediately above the sediment-water interface;
- The normal migration of benthic invertebrates into the water column during the night;
- The nocturnal efflux of methylmercury from the sediments is also assisted by the suboxic conditions in the sediment-water interface that typically occur at night;
- The feeding of pelagic or profundal organisms in the sediment-water interface.

Mercury loading along with the sulfur loading are probably the two most important direct factors influencing the mercury methylation/demethylation portion of mercury cycle in lakes. However, this portion of the mercury cycle is probably influenced by a number of indirect, less well understood, factors. Nitrogen and phosphorus, for example, are instrumental in the control of organic matter production and the relative organic matter content of the sediments that will, in turn, influence the microbial and invertebrate activity in this environment. Eutrophic or nutrient-rich environments tend to have more dissolved organic matter associated with them than oligotrophic environments. Dissolved organic matter, while it may be positively correlated with methylmercury concentrations in the water, may also retard the uptake of methylmercury in the lake biota.

The result of the various interacting factors influencing mercury methylation within the wide range of aquatic environment results in the great variation of biotic mercury levels. In general, however, the biota of oligotrophic lakes often show higher levels of methymercury than those in eutrophic lakes. Eutrophic lakes exhibit a high rate of production and thus a large biomass in which methylmercury will be diluted (compared to the relative small biomass in oligotrpohic lakes). The high production is also responsible for the production of a large amount of extracellular products which act as complexing agents with mercury and other metals. The high production of eutrophic lakes also gives rise to a high sedimentation rates. These high sedimentation rates enrich the sediments which are generally reducing in nature and support the formation of HgS which would tend retard methylmercury production.

Oligotrophic lakes and lakes subject to acidification have low productivity, low biomass and thus a lower degree of methylmercury dilution within that biomass. Acid pHs favour the formation of methylmercury and also results in an increased uptake from the atmosphere both of organic and inorganic mercury. The fact that sediments of oligotrophic lakes are generally oxidized and do not contain considerable amounts of complexing material (unless they are very humic) will lead to relatively high levels of bioavailable mercury (Beijer and Jernelov 1979)

The lake to be primarily impacted by the Jericho Mine (Lake D10) is a highly stressed aquatic ecosystems due to the almost total lack of dissolved substances normally associated with viable, self-sustaining and productive aquatic ecosystems. The process wastewater, sanitary and surface water runoff inputs will very probably cause a “fertilization effect” causing a dramatic and visible change in the Lake D10 ecosystem. As this lake is already a stressed environment, the addition of any dissolved elements or compounds will create a new and unique aquatic environment unlike any in the region. This is not necessarily a “bad” or “adverse” situation. Lake D10, as it currently exists, is not unique to the area in terms of its chemistry and biology. Thus, if mining in this area is to be permitted, it is only practical to use Lake D10 for the treatment of wastewater generated by the mine. Wastewater treatment systems of all types using static water generally develop some type of aquatic community within the confines of pH and temperature. The presence of a biotic component to a static wastewater treatment system will usually serve to enhance the treatment of the wastewater. In the case of the Jericho Mine, the biotic components of the lake to be used for wastewater treatment are so stressed that it would provide little enhancement if nutrients such a calcium, magnesium, sodium, potassium, phosphorus, nitrogen were not added. As previously stated, the aquatic ecosystem resulting from the input wastewater from the mine, process facility and surface runoff will be unique. Even though the concentrations of the various minerals and compounds entering Lake D10 can be predicted, the details surrounding the dynamics of the ecosystem cannot be defined. At the vary least, there will very probably be increased organic production which may result in much more organic sediments. The input of sulfur to the sediments along with the organic matter and its cycling within the lake will be critical to the SRB, which, like the rest of the microbial populations of these lakes currently live in an almost constant state of starvation. The development of the SRB microbial community will be a critical feature of the methylation/demethlyation processes of the mercury cycle.

Many of the numerous interactive factors, both direct and indirect, which act to influence the formation of methylmercury are, in large part, known from recent research. However, taking this information and applying it to a specific situation such as Lake D10 is fraught with pitfalls. Generally, the current weight of evidence implies that nutrient rich aquatic systems seem to have a lower net production of methylmercury as compared to nutrient poor systems even thought this net production is cyclic in nature and changes with the season. As Lake D10 becomes less nutrient poor and more nutrient enriched due to the input of various waste streams and surface runoff water, it should follow that net methylmercury production would decrease (i.e. the rate of methylmercury production should become less than the rate of degradation of methymercury on an annual basis. However, the combination of dissolved substances, their chemical and biochemical interactions and the interaction with an active biotic community resulting from the input of wastewaters and surface water runoff will produce a new and unpredictable ecosystem in Lake D10. The discharge from Lake D10 to Lake C3, may or may not produce smaller changes (compared to those in Lake D10) to the Lake C3 ecosystem, depending on the chemical composition of the Lake D10 discharge

While models of mercury cycling have been used to describe the cycling of mercury in various aquatic environments, the lack of knowledge for some specific or unique aquatic environment and the scarcity of the available data seem to be major obstacles to presenting a properly constructed model (Braga et al. 2000). As such, prediction of a decrease in net methlymercury production in Lake D10 and perhaps even on Lake C3 can only be made in very general terms and without much confidence.

7.0 CONCLUSIONS WITH REGARD TO THE ISSUES DISCUSSED

The previously discussed issues cover a wide range of topics. However, the statement was made during this discussion that the science of geochemistry was more advanced than the science of biogeochemistry. This situation results in a rather well defined prediction of the amounts of some chemical substances that will occur in the surface waters associated with the Jericho project. By contrast, the biological response to the presence of these substances is much less well defined. Many substances entering the surface waters as a result of mining activity are known to be detrimental to aquatic life. However, these substances do not exist as a simple inorganic, dilute solution. Rather, as they are released into these waters, they become part of a dynamic ecosystem in which they both exert an effect, and in turn, are affected by other substances and aquatic organisms from the microbial level to that of fish. Some of the substances introduced by mining activities will have a potentially detriment effect on the resident aquatic life. However, other substances introduced by the mining activities will have the potential to reduce or even eliminate the detrimental effects cause by the other substances. In addition, the biological portion of the ecosystem has the capability of adaptation to the introduced substances.

Unfortunately, there so many geochemical and biochemical reactions occurring simultaneously as well as sequentially (to say nothing of the lack of information on reaction kinetics) that modeling of these reactions at the level required for accurate prediction of the final effect on the entire aquatic ecosystem is currently impossible. All that can be predicted is that a change in the aquatic ecosystems receiving various waste waters and surface runoff water from the mine will occur. Given the well-documented resiliency of ecosystems in general and aquatic ecosystems in particular, it is safe to say that the ecosystems directly impacted by mining will survive, but perhaps with their structure and function altered. The degree of alternation, as with everything else, depends on the amount and type of the substances added and the final interactive product of all the positive and negative attributes of these substances. Generally, the less added, the less alteration can be expected.

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