Supporting Document N1

Tier 2 Risk Assessment, Giant Mine Remediation Plan (SENES, 2006)

FINAL REPORT

TIER 2 RISK ASSESSMENT
GIANT MINE REMEDIATION PLAN

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1.0 INTRODUCTION

1.1 PREVIOUS RISK ASSESSMENTS

Two previous rounds of risk assessment of the Giant Mine site have been completed and provide starting points for the work described herein. The 2001-2002 ecological and human health risk assessments were carried out in support of the evaluation of alternatives for management of the underground arsenic trioxide dust. These assessments examined the potential adverse effects on human health and aquatic and terrestrial species in the vicinity of the Giant Mine site based on different arsenic discharge scenarios. The assessments focused primarily on potential effects of arsenic exposure on the aquatic environment including Baker Creek, which flows through the Giant Mine site, and Back Bay and Yellowknife Bay, which receive the outflow from Baker Creek. Arsenic in the terrestrial environment was also incorporated into the risk assessments; however, the effects of surface remediation activities were not addressed.

The results of the 2001-2002 risk assessments suggested that a number of aquatic and terrestrial species were potentially at risk in Baker Creek for several of the arsenic release scenarios. Of the terrestrial animals, mink, muskrat and scaup duck were found to be at the greatest potential risk. In Back Bay and Yellowknife Bay, none of the aquatic or terrestrial species considered in the assessment were found to be at risk for any of the arsenic release scenarios.

With respect to humans, the 2001-2002 risk assessment found that the arsenic doses from the consumption of drinking water, fish and animals taken from the study area fell within the normal range of exposure levels for most Canadians for those scenarios involving low-end arsenic release rates. For the more pessimistic scenarios, the estimated arsenic doses exceeded the upper bound for typical Canadian adults and children. The analysis showed that the primary pathways of exposure were related to consumption of water, fish and duck. The results demonstrated that management scenarios that will limit the total arsenic release rate to less than 2,000 kg/y would result in doses that did not exceed the range for typical Canadian adults and children.

The results of the 2001-2002 risk assessments assisted the project team in the selection of the preferred alternative for dealing with the underground arsenic trioxide. They also showed that it would be important to consider aquatic pathways and receptors in any further assessment of site remediation plans.

In 2003, a screening level risk assessment of the Giant Mine site was completed as part of the funding process for the Federal Contaminated Sites Action Plan (FCSAP) program (SENES 2003). The screening level assessment examined existing surface conditions at the Giant Mine site, and used conservative assumptions and literature transfer factors. The 2003 results indicated that in addition to arsenic, antimony, lead, and nickel also present risks to humans.

However, a subsequent review of the 2003 findings, using site-specific transfer factors, resulted in hazard quotient values below 0.5 for lead and nickel. Hazard quotient values of less than 1 are considered to be acceptable; therefore, lead and nickel are not contaminants of concern at the Giant Mine site.

For antimony, the use of site-specific transfer factors resulted in hazard quotient values to 1.2 for an adult and 1.9 for a child. The primary source of antimony is contaminated soils, and the proposed clean-up of arsenic-contaminated soils will also result in a substantial reduction in the antimony levels in the soils left on surface at the site. It has been estimated that after remediation of the arsenic-contaminated soils, antimony concentrations will be below the Canadian Council of Ministers of the Environment (CCME) guideline of 20 mg/kg. (Appendix E provides a more in-depth discussion on the antimony assessment.) Additionally, there are hydrocarbon contaminated soils at the site. The hydrocarbon-contaminated soils are associated with the arsenic-contaminated soils; therefore, clean-up of these soils will result in a clean-up of the hydrocarbon contaminated soils.

Thus, it was determined that only arsenic would be carried through the Tier 2 risk assessment presented herein.

1.2 CURRENT RISK ASSESSMENT

The current risk assessment involved the evaluation of proposed remedial options for arsenic contamination on the surface of the Giant Mine site, as well as possible future releases of arsenic from the underground workings. Therefore, this assessment considered exposure via water pathways of several aquatic and terrestrial species as well as exposure of several of the terrestrial species to contaminated soils, sediments and terrestrial vegetation. Similarly, the contributions of terrestrial pathways to arsenic transfer from soil-to-humans, from soil-to-vegetation-to-humans and from soil-to-vegetation-to-animals-to-humans were also considered. The risk assessment included a complete review of available data on arsenic levels in various media, recent biological studies on fish and muskrat in Baker Creek and benthic invertebrate studies in Baker Creek and Back Bay, prediction of arsenic intakes by ecological and human "receptors", and a comparison of the predicted intakes to toxicological reference values. The risk assessment was undertaken within a probabilistic modelling framework.

Besides assessing the risks of the preferred remediation options for the surface and underground facilities at the Giant Mine site, the risks associated with a *"do-nothing"* scenario were evaluated. Appendix F provides an assessment of the *"walk-away"* or *"do-nothing"* scenario at the Giant Mine site. In this scenario, arsenic surface contamination would be left on site and the mine would be allowed to flood. The worst-case arsenic loadings associated with this scenario were used in the assessment.

1.3 ECOLOGICAL RISK ASSESSMENT FRAMEWORK

The Canadian Council of Ministers of the Environment (CCME 1996, 1997) has provided general guidance concerning their views on what constitutes an ecological risk assessment (ERA). The recommended framework is similar to that proposed by Environment Canada (Environment Canada 1997). The CCME recommends three levels of investigation:

1) Screening Level Assessment (SLA or Tier 1): essentially a qualitative assessment of potential risks to important ecological receptors.
2) Preliminary Quantitative Risk Assessment (PQRA or Tier 2): focuses on filling gaps identified at the screening level.
3) Detailed Quantitative Risk Assessment (DQRA or Tier 3): includes more detailed data and modelling.

Each level of the assessment includes the following elements:

- Receptor Characterization: At this phase of the assessment the potential receptors are identified and the pathways of exposure defined.
- Exposure Assessment: The purpose of this stage is to quantify the contact between the receptor and the contaminant of concern.
- Hazard Assessment: This phase of the ERA examines the potential effects of a contaminant to a receptor.
- Risk Characterization: The risk characterization stage combines the information collected in the exposure assessment and the hazard assessment, and the potential for adverse ecological effects is estimated.

The rigour of the risk assessment adopted for a particular situation should be commensurate with the degree and extent of potential harm and may progress to a more stringent level (i.e. from Tier 1 to Tier 2 or from Tier 2 to Tier 3) depending on the findings at each level. Each level in this tiered approach has the same structure and builds upon the data, information, knowledge and decisions generated from the preceding level. Thus, each level is progressively more rigorous and complex.

As was done previously for the arsenic trioxide in the underground vaults, the current assessment is based on the principles of a Tier 2 preliminary quantitative risk assessment. Included in the Tier 2 assessment were aquatic receptors from various trophic levels (e.g. aquatic plants, phytoplankton, zooplankton, benthic invertebrates, pelagic and benthic fish). The aquatic ecosystem assessment considered exposure to arsenic present in the Baker Creek drainage, as well as in Back Bay and Yellowknife Bay. In addition, detailed pathways modelling was conducted for several terrestrial receptors to estimate their potential exposure to arsenic present

in these water bodies and the study area soils and terrestrial vegetation. The terrestrial receptors identified for inclusion in the assessment included three duck species (mallard, merganser and scaup), bear, hare, spruce grouse, mink, moose, muskrat and wolf. Caribou were also included in the assessment as they comprise a significant portion of the local diet and are known to winter in the study area. It should be noted that site-specific measured data on various media and ecological species were used where available. An important consideration in an ecological risk assessment is that it is not necessary to consider all species found in an area, but rather to consider selected species with different dietary characteristics, so that the effects of all exposure pathways are considered.

Adverse ecological effects were characterized by the value of a simple screening index (generally considered to be 1). This index is calculated by dividing the expected exposure or dose concentration by the selected toxicity reference value for arsenic for each ecological receptor. The approach to the ecological risk assessment is described in Appendix D.

1.4 HUMAN HEALTH RISK ASSESSMENT APPROACH

A human health risk assessment (HHRA) evaluates the probability of adverse health consequences to humans caused by the presence of chemical contaminants in the environment. Receptor characteristics (e.g. proportion of time spent in the study area, source of drinking water, composition of diet) and exposure pathways (e.g. inhalation and ingestion) are taken into consideration. Unlike the ERA, which is concerned with population effects, the HHRA focuses on the effects on individuals. In this assessment, the HHRA examined the potential impact of effluent discharges on adults and children in different parts of the community. The assessment considered similar receptors as was done in the previous assessment, as well as a receptor living at the Giant Mine Townsite. Dietary intakes, as well as intakes due to soil ingestion, water consumption and dermal contact, were considered. A sensitivity analysis was conducted as part of the assessment to examine the influence of obtaining different dietary components from different locations.

Dietary arsenic consists of both organic and inorganic forms of arsenic. Thus, this assessment apportioned the dietary contributions of arsenic into organic and inorganic arsenic forms. Recent studies suggest that some forms of organic arsenic are generally as toxic as inorganic arsenic. Therefore, the toxicity reference value that was based on studies with inorganic arsenic compounds was also applied to the toxic organic fraction of the diet. The toxicity reference values used for the assessment were obtained from Health Canada.

Total estimated arsenic intakes were compared to intakes based on typical Canadian exposures. Inorganic and toxic organic arsenic exposures were used to estimate both carcinogenic and non-carcinogenic risk. In this study, the incremental risk of carcinogenic effects was compared to a 1

in a hundred thousand (1×10^{-5}) reference lifetime risk level. The one in a hundred thousand chance of incurring cancer is considered to be an insignificant risk level by regulatory agencies such as Health Canada.

1.5 REPORT STRUCTURE

The report has been structured into several sections, each of which describes specific aspects of the risk assessment. These aspects include:

Section 2 – Site Characterization: Provides a summary of the most pertinent information from recent surveys of surface water quality, sediment quality, fish communities, soil surveys and garden produce, berry and wildlife surveys to establish current (baseline) conditions. Information on the quality of water contained in Baker Creek, Back Bay and Yellowknife Bay is also summarized, as it is required for prediction of future changes in receiving water quality.

Section 3 – Receptor Characterization: Identifies the aquatic and terrestrial species selected for inclusion in the risk assessment, as well as the human receptors (i.e. adults and children) who live in the study area.

Section 4 – Exposure Assessment: Describes the pathways model used to predict the fate of arsenic in the environment, including its uptake by aquatic and terrestrial species and removal to lake sediments. The site-specific transfer factors used in the pathways model, as well as site-specific arsenic accessibility are also detailed as are issues related to arsenic accessibility. Finally, the pathways of exposure of human receptors and their respective dietary characteristics are described.

Section 5 – Hazard Assessment: Details the toxicity reference values for arsenic used in the assessment to characterize the risks of potential effects on the health of ecological species and humans.

Section 6 – Risk Characterization: Presents the results of the pathways modelling and risk assessments.

Section 7 – Summary and Conclusions: Provides a synopsis of the basis used for the ERA and HHRA and the findings of these assessments.

2.0 SITE CHARACTERIZATION

This section describes the study area, presents a summary of arsenic levels measured in environmental media in the area and outlines considerations included in the assessment of environmental effects and risks of arsenic releases from remediation actions at the Giant Mine site. Details of the remediation option under investigation and the results of engineering and hydrogeological investigations are presented in other project documents.

2.1 STUDY AREA

The study area comprises the Giant Mine site, located immediately north of the City of Yellowknife, and surrounding environs including a segment of the northern arm of Great Slave Lake known as Yellowknife Bay (see Figure 2.1-1). Yellowknife Bay receives drainage from the Yellowknife River at its north end and extends about 18 km before opening into Great Slave Lake. In this study, Yellowknife Bay was broken into three different segments, as shown in Figure 2.1-2. The first segment, encompassing Back Bay, a small bay separated from the main part of Yellowknife Bay by Latham Island, and the majority of the shoreline along the Giant Mine site, was chosen to represent the highest impacted area. Segment 2 encompasses input from the Yellowknife River and extends to the tip of Latham Island and is referred to in this assessment as North Yellowknife Bay. The third segment encompasses the stretch from the City of Yellowknife to the Dettah community and is referred to herein as South Yellowknife Bay. The physical characteristics of the bay segments are summarized in Table 2.1-1.

TABLE 2.1-1
BACK BAY AND YELLOWKNIFE BAY PHYSICAL ATTRIBUTES

Characteristics	Units	Back Bay Segment 1	Yellowknife Bay	
			Segment 2	Segment 3
Surface Area	m^2	2.51×10^6	5.30×10^6	12.73×10^6
Volume	m^3	17.33×10^6	44.03×10^6	143.02×10^6
Mean Depth	m	6.9	8.3	11.2

Surface water modelling was conducted to assess the movement of arsenic from the Giant Mine site into the aquatic environment of Back Bay via surface run-off in Baker Creek. A lake dispersion model (referred to herein as LAKEVIEW and described in Appendix B) was used to determine the concentration of arsenic in surface water and sediments in Back Bay and Yellowknife Bay for the remediation case loading scenario. Historical information was used to calibrate the model, as discussed in Appendix B.

FIGURE 2.1-1

REGIONAL SETTING

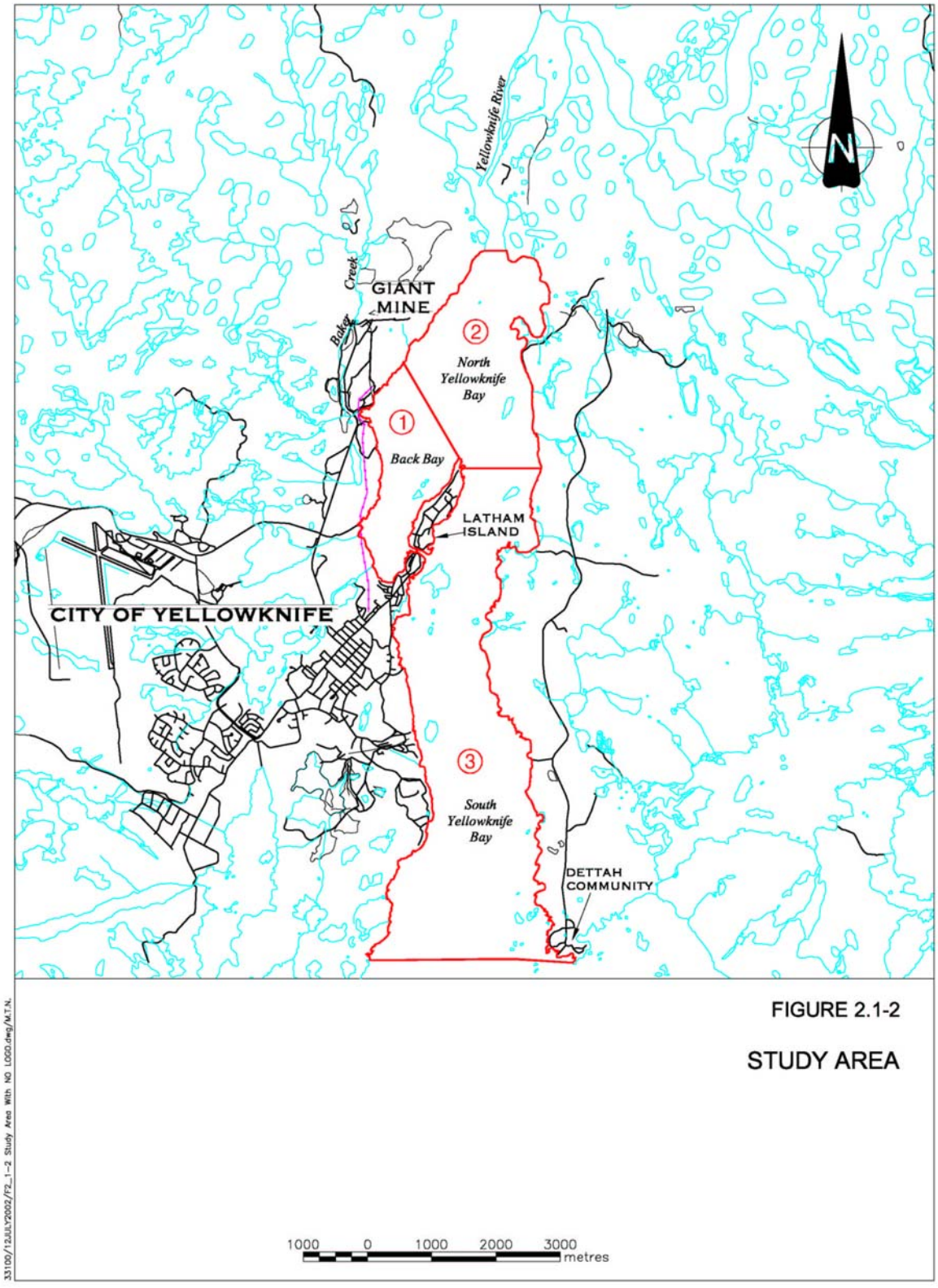


FIGURE 2.1-2

STUDY AREA

Besides the aquatic environment, the risk assessment also took into account arsenic levels present in soil and vegetation at the Giant Mine property and in the surrounding communities. These include the City of Yellowknife, Latham Island and the community of Dettah on the east shore of Yellowknife Bay.

2.2 SOURCE CHARACTERISTICS

2.2.1 Potential Future Releases Associated with Remediation Case

The modifications proposed in the Remediation Plan discussed in the main report are expected to decrease the arsenic discharges from surface sources within the mine area. After those measures are implemented, it is reasonable to expect that background and mine area arsenic releases will be reduced. In addition, some of the arsenic load from the mine site will go directly into Back Bay and some will still enter Baker Creek. The risk assessment examined one "Remediation Case" that was selected to cover these likely arsenic releases.

In the Remediation Case, 290 kg/yr from background sources (220 kg/yr upstream of the mine and 70 kg/yr from tributaries) and 190 kg/yr in surface runoff from the mine site is assumed to go directly into Baker Creek, for a total of 480 kg/yr. In addition, it was assumed that 140 kg/yr of arsenic from the treatment plant and 70 kg/yr from surface run-off would enter Back Bay. In total for the Remediation Case, it was assumed that 690 kg/yr arsenic was emitted to the aquatic environment. In addition, Baker Lake sediments will be removed and some sections of Baker Creek will be realigned; however, portions of Baker Creek will still have arsenic concentrations of up to 2,200 mg/kg. No surface ponds will be present on site with the exception of the treated water storage pond. The arsenic concentration in the pond is expected to average approximately 0.38 mg/L, but the pond will be fenced. Therefore, it will be inaccessible.

2.3 ARSENIC IN ENVIRONMENTAL MEDIA

The following sections summarize raw data in a number of media that were considered in the assessment. In the aquatic environment, measured data in water, aquatic vegetation, benthic invertebrates and fish were used to derive transfer factors for us in the pathways model. In the terrestrial environment, measured concentrations of soil, garden vegetables and berries were used. Site-specific transfer factors were derived from measured soil and vegetation concentrations for predicting concentrations of arsenic in browse and forage.

2.3.1 Surface Water Quality

A detailed summary of the data collected on arsenic levels in Baker Creek, Back Bay and Yellowknife Bay waters is provided in Appendix A. A number of literature sources were

reviewed for the compilation of the data, including: Dillon (2002a, 2002b, 2004), Falk *et al.* (1973), HydroQual (1989), Jackson *et al*. (1996), Jackson *et al.* (1998), Lorax (1999), Mace (1998), Moore *et al.* (1978) and Ollson (1999). Table 2.3-1 presents a statistical summary of the data for each area of concern. It should be noted that high concentrations of arsenic in Baker Creek occurred prior to the water treatment process being installed.

TABLE 2.3-1
MEASURED ARSENIC LEVELS IN WATER (1973-2004) (µg/L)

	Baker Creek	Back Bay	Yellowknife River	North Yellowknife Bay	South Yellowknife Bay
# of Samples	53	49	8	26	35
Arithmetic Mean	998	31	0.26	11	17
Standard Deviation	2332	108	0.07	20	89
Geometric Mean	170	4.7	0.25	3.9	1.3
Geometric Std Dev	8.9	6.0	1.4	4.2	3.8
Minimum	0.3	0.3	<0.3	0.3	0.3
Maximum	12,600	740	0.3	83	350

Note: for the purposes of the summary, values measured as < the detection limit were considered as ½ the detection limit.

2.3.2 Sediment Quality

A large number of studies on sediments in Yellowknife Bay have been carried out and a detailed analysis of the sediment data considered for this assessment is presented in Appendix B. Literature sources included: Golder (2004), Dillon (2002a, 2002b, 2004), HydroQual (1989), Jackson *et al.* (1996), Jackson *et al.* (1998), Mace (1998), Moore *et al.* (1979) and Mudroch *et al.* (1989). Table 2.3-2 presents a statistical summary of the data.

TABLE 2.3-2
MEASURED ARSENIC LEVELS IN SEDIMENT (mg/kg (dw))

	Baker Creek	Back Bay	North Yellowknife Bay	South Yellowknife Bay
# of Samples	29	37	76	11
Arithmetic Mean	1,545	875	269	32
Standard Deviation	837	707	347	22
Geometric Mean	1,202	612	110	26
Geometric Std Dev	2.5	2.7	4.2	1.9
Minimum	69.7	59.2	6.9	10
Maximum	3,757	3,140	1,870	90

Note: Surficial sediments (0 - 5 cm) only were considered.

2.3.3 Fish Surveys

A detailed summary of data collected on arsenic levels in fish is provided in Appendix A. In preparation of Table 2.3-3 provided in this section, the data on several fish species were combined as it was found that there was little difference in arsenic levels between species. Arsenic concentrations in long nose sucker, northern pike and lake whitefish were available for Baker Creek and Yellowknife Bay (Falk *et al.* 1973). Northern pike and lake whitefish are included for Yellowknife Bay. For Resolution Bay, burbot, inconnu, lake trout, northern pike and walleye were analyzed by Evans *et al.* (2001) and Boucher *et al.* (1997).

TABLE 2.3-3
MEASURED ARSENIC LEVELS IN FISH MUSCLE (mg/kg (ww))

	Baker Creek	Yellowknife Bay	Resolution Bay	Overall
# of Samples	9	408	14	431
Arithmetic Mean	0.29	0.20	0.16	0.20
Standard Deviation	0.06	0.15	0.07	0.15
Geometric Mean	0.28	0.20	0.15	0.19
Geometric Std Dev	1.24	2.08	1.48	1.66
Minimum	0.21	0.02	0.08	0.02
Maximum	0.37	1.11	0.32	1.11

Note: for the purposes of the summary, values measured as < the detection limit were considered as ½ the detection limit.

Dillon (2002c) carried out some recent surveys of fish in Baker Creek. In these surveys they captured 3 arctic grayling and 2 northern pike upstream of the mine. They also captured 7 longnose sucker and 3 northern pike downstream near the mouth of Baker Creek. The average

concentration of arsenic in the fish captured upstream of the mine was 3 mg/kg wet weight and the average captured downstream was 1 mg/kg wet weight. Two of the samples from the upstream location and 7 of the samples from the downstream location were below the detection limit. The authors could not determine why there was a difference in the arsenic concentrations in these two different locations. The overall arsenic concentration in fish tissue was 1.6 mg/kg. These concentrations were significantly elevated compared to those of previous studies which were conducted in 1973.

2.3.4 Aquatic Vegetation Concentrations

The complete data on measured arsenic concentrations in aquatic vegetation are provided in Appendix A. A summary of the vegetation data is presented in Table 2.3-4 for the Yellowknife Area for the combined dataset of all plant species. Vegetation considered in the summary include: bur reed, cattail, duckweed, Eurasian water milfoil, pondweed, horsetail and sedge. Data included in the summary statistics were obtained from Koch *et al*. (2000) and Dillon (2002a).

TABLE 2.3-4
MEASURED ARSENIC LEVELS IN
AQUATIC VEGETATION (mg/kg (dw))

	Yellowknife Area
# of Samples	23
Arithmetic Mean	58
Standard Deviation	55
Geometric Mean	33
Geometric Std Dev	3.9
Minimum	0.52
Maximum	260

2.4 TERRESTRIAL ENVIRONMENT

2.4.1 Soil Concentrations

Measured arsenic concentrations in soil were available for a number of areas around Yellowknife, including the Giant Mine property, the Giant Mine Townsite, Latham Island, the Dettah community and the City of Yellowknife. The complete dataset is provided in Appendix A and a summary of the data is presented in Table 2.4-1.

TABLE 2.4-1
MEASURED ARSENIC LEVELS IN SOIL (mg/kg (dw))

	Yellowknife City	Latham Island	Dettah Community	Giant Mine Site	Old Giant Mine Townsite
# of Samples	282	24	8	226	41
Arithmetic Mean	97	193	46.6	2497	1630
Standard Deviation	157	202	53.1	6313	3145
Geometric Mean	42	106	26.3	988	353
Geometric Std Dev	3.8	3.6	3.1	4.4	6.7
Minimum	2.5	7	7.2	5.2	19
Maximum	1190	780	144	87000	16600

2.4.2 Terrestrial Vegetation Concentrations

The complete data on measured arsenic concentrations in vegetation are provided in Appendix A. A summary of the vegetation arsenic measurements is presented in Table 2.4-2 for moss, lichen and mushrooms. Data for other types of vegetation are provided in Appendix A.

TABLE 2.4-2
MEASURED ARSENIC LEVELS IN
TERRESTRIAL VEGETATION (mg/kg (dw))

Terrestrial Vegetation	# of Samples	Minimum	Maximum	Geometric Mean	Average	Standard Deviation
Moss	7	490	1900	1018.8	1100	452
Lichen	9	6.4	2300	55.7	336.4	754
Mushroom	5	8.3	1010	70	295.6	434

Arsenic concentrations in berries were studied by Davey *et al.* (1998) and associated papers. Areas of interest included the City of Yellowknife, Giant Mine, Joliffe Island (south of Latham Island) and Dettah Road. Berries analyzed included: raspberry, gooseberry, cranberry, rose hip and blueberry. A summary of the data is presented in Table 2.4-3.

TABLE 2.4-3
MEASURED ARSENIC LEVELS IN BERRIES (mg/kg (ww))

	Yellowknife	Giant Mine	Joliffe Island	Dettah Road	Overall
# of Samples	7	6	3	2	18
Arithmetic Mean	0.10	0.52	0.12	0.04	0.24
Standard Deviation	0.06	0.70	0.04	0.001	0.43
Geometric Mean	0.08	0.28	0.12	0.04	0.12
Geometric Std Dev	2.0	3.3	1.3	1.0	2.8
Minimum	0.02	0.05	0.08	0.04	0.02
Maximum	0.20	1.91	0.12	0.04	1.91

2.4.3 Medicinal Tea Concentrations

A review of information from the Medicinal Plants Study Report 2002-2003 (Chan 2003) provided a summary of arsenic concentrations in teas obtained from various medicinal plants within the study area. Table 2.4-4 provides a summary of the arsenic concentrations in medicinal teas obtained from the Giant Mine site, as well as in the vicinity of the Dettah Community.

TABLE 2.4-4
MEASURED ARSENIC LEVELS IN MEDICINAL TEAS (μg/L)

Location	Summary of Measured Data					
	# of Samples	Minimum	Maximum	Arithmetic Mean	Geo Mean	Geo Std. Dev.
Giant Mine	9	0.1	527.4	77.8	32.3	3.8
Dettah	29	0.05	170.4	29.6	14.5	3.3

2.4.4 Terrestrial Animal Concentrations

Information exists on contaminant levels in wildlife from the Northwest Territories in general. Although these data are from areas other than Yellowknife and the receptor locations considered in this assessment, the data from the literature are presented here to provide a reference level and comparison for predicted levels. Recently, a survey was conducted on muskrats living on Baker Creek; these results are also presented below.

Moose and Caribou

Six moose and two woodland caribou were sampled by Sahtu Dene Council (1998). Arsenic levels in all kidney, muscle and liver samples for moose and caribou were below the detection limit of 0.2 mg/kg (dw). Comments on the results included the small number of samples analyzed for the large geographically diverse area.

Similar results were presented by Elkin *et al.* (1998) in a study of 20 caribou from the Bluenose caribou herd. Arsenic levels measured in liver, kidney, spleen and bone were below the detection limit of 0.2 mg/kg (dw) for all samples. The results of a comparable study of 20 caribou from the Lake Harbour herd (Elkin *et al.* 1999) showed arsenic levels in liver and kidney below the detection limit. Liver and kidney of 20 caribou from the Beverly herd also had average arsenic levels below the detection limit (Elkin and MacDonald 2000). Of the 20 samples, two liver samples had detectable arsenic levels (maximum 0.2 mg/kg (dw)) and three kidney samples had detectable arsenic levels (maximum 0.4 mg/kg (dw)).

As part of a monitoring project of contaminants in Yukon wildlife, hunters submitted tissue samples for wildlife including bison, moose, caribou and mule deer (Gamberg and Palmer 1998). Results of arsenic levels in muscle were available for six bison samples. The average arsenic concentration was 0.06 mg/kg (dw) with a standard deviation of 0.02 mg/kg (dw). Reported contaminant levels were considered to be baseline.

Birds

A summary of arsenic levels in harvested avian species in the Canadian Arctic from 1988 to 1994 was provided by Braune *et al.* (1997). Birds were grouped by trophic level:

- browser - ground dwellers such as grouse and ptarmigan that feed mainly on terrestrial vegetation;
- grazers - geese that graze mainly on aquatic and terrestrial vegetation;
- omnivores - surface-feeding ducks with a varied diet consisting of mainly aquatic vegetation;
- molluscivores - diving ducks feeding mainly on invertebrates; and
- piscivores-diving - ducks feeding mainly on fish.

Results from Braune *et al.* (1997) are shown in Table 2.4-5.

TABLE 2.4-5
MEASURED ARSENIC LEVELS IN WATERFOWL AND GAME BIRDS (µg/kg(ww))

Trophic Level	Number of samples	Arsenic Concentration Range
Browser	91	< 70
Grazers	59	< 100
Omnivores	149	< 30 – 110
Molluscivores	178	< 30 – 426
Piscivores	5	< 30 – 232

Muskrat

A study by Kennedy *et al.* (1998) completed on beaver and muskrat in the area of the Slave River Delta did not detect (detection limit of 5 mg/kg (dw)) arsenic in the liver or muscle of beaver (n = 12) and muskrat (n = 10 for liver, n = 9 for muscle).

Recent biological studies were carried out that specifically targeted the muskrat population on Baker Creek. Twelve active burrows were discovered (downstream of the mine workings) that supported an estimated population of between 66 and 197 animals (Jacques Whitford 2003). Subsequent to this study, a number of muskrat were trapped both upstream and downstream of the mine workings and arsenic analyses were carried out on the muscle, organs (liver and kidney) and tail of the muskrats (Golder 2004). Table 2.4-6 provides a summary (mean and maximum) of the measured data. The analyses showed that the kidney had the highest concentration of arsenic and the muscle had the lowest arsenic concentration (Golder 2004). This is not surprising as studies in other animals generally support the notion that organs have higher concentrations of metals than the muscle. As seen from the table, the mine workings have an effect on the concentration of arsenic measured in the muskrats. Muskrats downstream have, on average, approximately two times higher concentrations than the ones collected upstream.

TABLE 2.4-6
SUMMARY OF MEASURED ARSENIC LEVELS IN TISSUES OF MUSKRATS FROM BAKER CREEK

	Measured Arsenic Concentrations (mg/kg ww)					
	Upstream			Downstream		
	Liver	Kidney	Muscle	Liver	Kidney	Muscle
Mean	0.66	0.7	0.24	1.39	2.64	0.51
Maximum	1.18	1.4	0.5	1.76	7.18	0.63

Source: Golder (2004).

3.0 RECEPTOR CHARACTERIZATION

The receptor characterization phase involves selection of ecological and human receptors for inclusion in the risk assessment and identification of their pathways of exposure to the contaminants of concern. As it is not practical to assess risks to all ecological species, it is common practice to select representative species based on level of potential exposure, importance as a food source for other species and/or humans, importance for cultural reasons, or because they are endangered or rare species. Factors generally considered in the selection of human receptors include proximity of residence to source, reliance on local vegetation and game as a food source, use of the study area, etc.

3.1 ECOLOGICAL RECEPTORS

The assessment for the Remediation Case at the Giant Mine site considered the potential adverse effects on the ecology in the area, as well as the water and sediment quality. The health of ecological receptors depends on the quality of the food (i.e., aquatic and terrestrial vegetation and biota), soil or sediment and water they consume, therefore, the sources of these dietary items are important aspects to consider when assessing the implications of the Remediation Case at the Giant Mine site.

3.1.1 Aquatic Receptors

Aquatic receptors selected for the assessment were based on the species chosen in previous assessments at the Giant Mine site. Figure 3.1-1 provides a schematic representation of the selected ecological receptors for the aquatic environment. The aquatic species chosen represent a typical food chain that would be found in aquatic systems and are known to be found in Baker Creek, Back Bay and Yellowknife Bay (see Appendix D).

3.1.2 Terrestrial Receptors

Table 3.1-1 provides a list of the receptors assessed from the terrestrial environment. These receptors were chosen as representative species found in the area (see Appendix D for a detailed discussion). In this assessment, it was assumed that all the terrestrial receptors would be found along Baker Creek. For Back Bay and Yellowknife Bay, it was assumed that ducks were the only terrestrial receptors that would spend a large portion of their time, while in the study area, in contact with these open water bodies.

FIGURE 3.1-1
AQUATIC RECEPTORS CONSIDERED IN THE ASSESSMENT

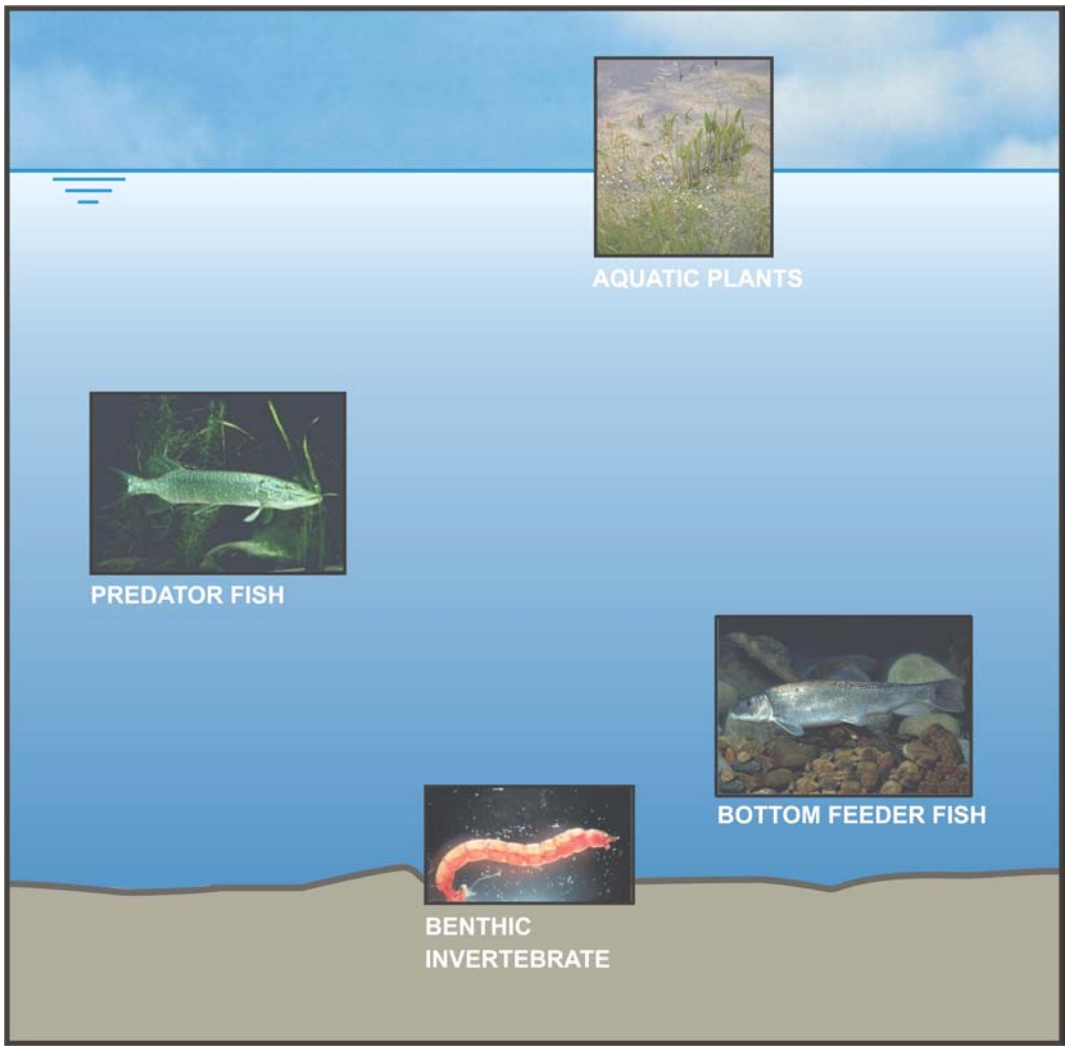


TABLE 3.1-1
SUMMARY OF THE ECOLOGICAL RECEPTORS SELECTED FROM THE TERRESTRIAL ENVIRONMENT

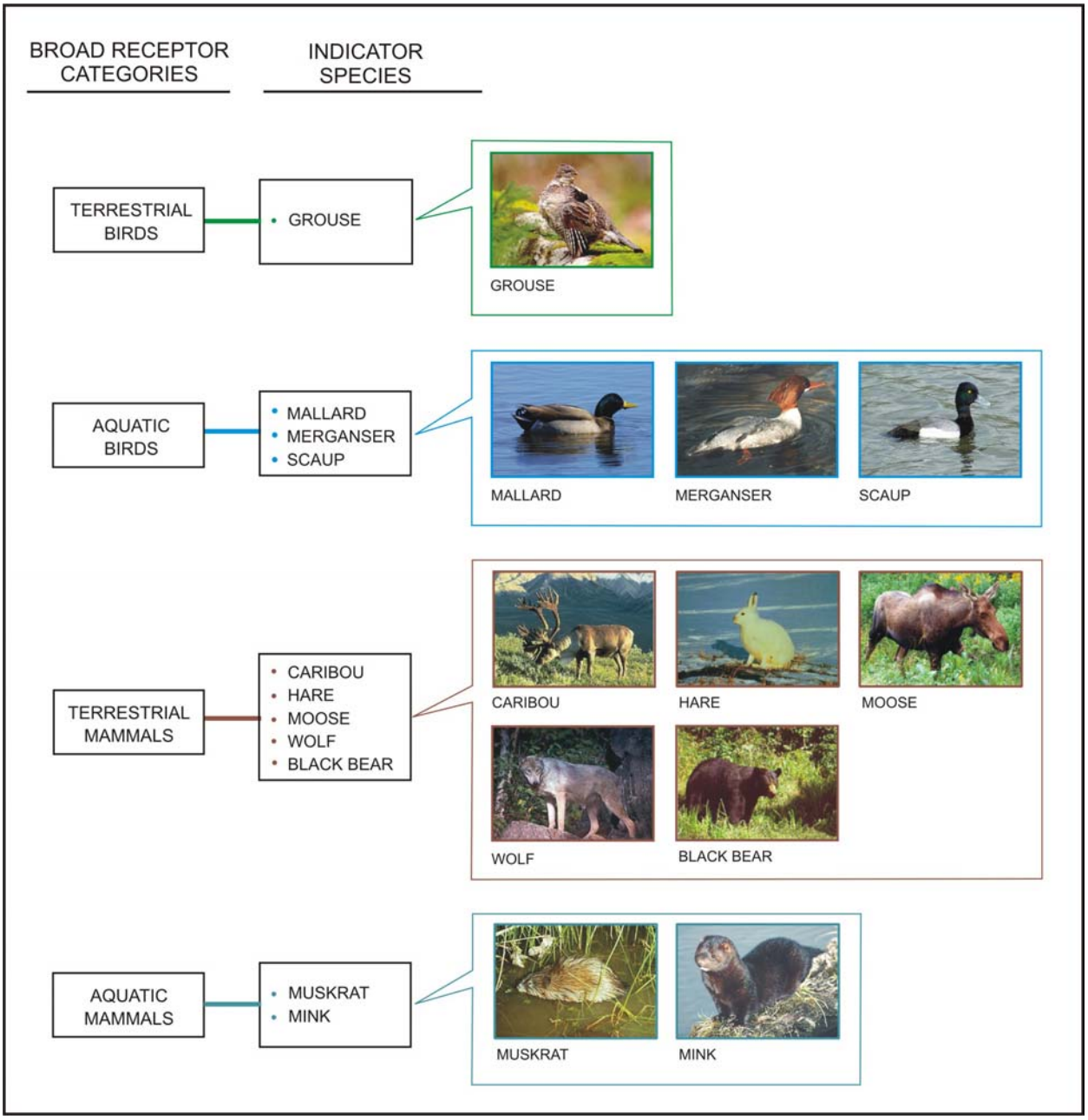
HERBIVORE • Barren Ground Caribou • Moose • Muskrat • Snowshoe hare • Spruce grouse
OMNIVORE • Black Bear • Ducks - Mallard - Merganser - Scaup
CARNIVORE • Mink • Wolf

Five different herbivore species were chosen as being representative of species in the area: Muskrat which consumes primarily aquatic vegetation; moose which consume aquatic and terrestrial vegetation; and barren ground caribou, spruce grouse and hare which consume only terrestrial vegetation. Ptarmigan was not explicitly considered in the assessment as its diet is very similar to that of spruce grouse which was chosen for this assessment.

Black bear and ducks were selected as representative of the omnivore species. Ducks were chosen as representative of aquatic birds and were subdivided into mallards (which consume a mixture of plankton from the water column and benthic invertebrates from the lake sediment), mergansers (which consume primarily fish) and scaup (which consume mainly aquatic snails from lake sediment). Bear was chosen since a large portion of its diet consists of berries.

Finally, two different carnivores were chosen: wolf, which consume mainly terrestrial animals, and mink, which consume mainly fish from the aquatic environment. The wolf was chosen since it is important culturally to the First Nations People. While fox are known to be found in the vicinity of the Giant Mine site, they have not been explicitly considered since their exposure is captured within the range of exposures of the mink and the muskrat. Similarly, these two receptors also encompass the exposure of a beaver. All terrestrial receptors were assumed to consume soil or sediment depending on where they obtained their food. Figure 3.1-2 provides a schematic of the receptors selected for this assessment. The following section discusses the pathways that have been considered in this Tier 2 assessment.

FIGURE 3.1-2
TERRESTRIAL RECEPTORS CONSIDERED IN THE ASSESSMENT



3.2 ECOLOGICAL PATHWAYS

Several different pathways were considered in the ecological assessment. These pathways are linked to either the aquatic environment of Baker Creek, Back Bay and Yellowknife Bay or the terrestrial environment including the Giant Mine site and lower Baker Creek watershed.

Figures 3.2-1a to 3.2-1d provide schematic representations of the potential pathways of exposure for the terrestrial receptors considered in this assessment. The figures also provide the ingestion rates for all the pathways considered for each of the terrestrial receptors.

It has been assumed that all species with the exception of ducks drink water from Baker Creek while in the study area. These species also obtain all their food from along Baker Creek or the Giant Mine site. Several different ducks were assessed in different areas such as Baker Creek, Back Bay and Yellowknife Bay. These ducks were assumed to obtain water from the location where they were assessed. Details of receptor dietary characteristics and other model parameters are provided in Appendix D.

FIGURE 3.2-1 a
POTENTIAL PATHWAYS OF EXPOSURE FOR GROUSE, CARIBOU AND HARE

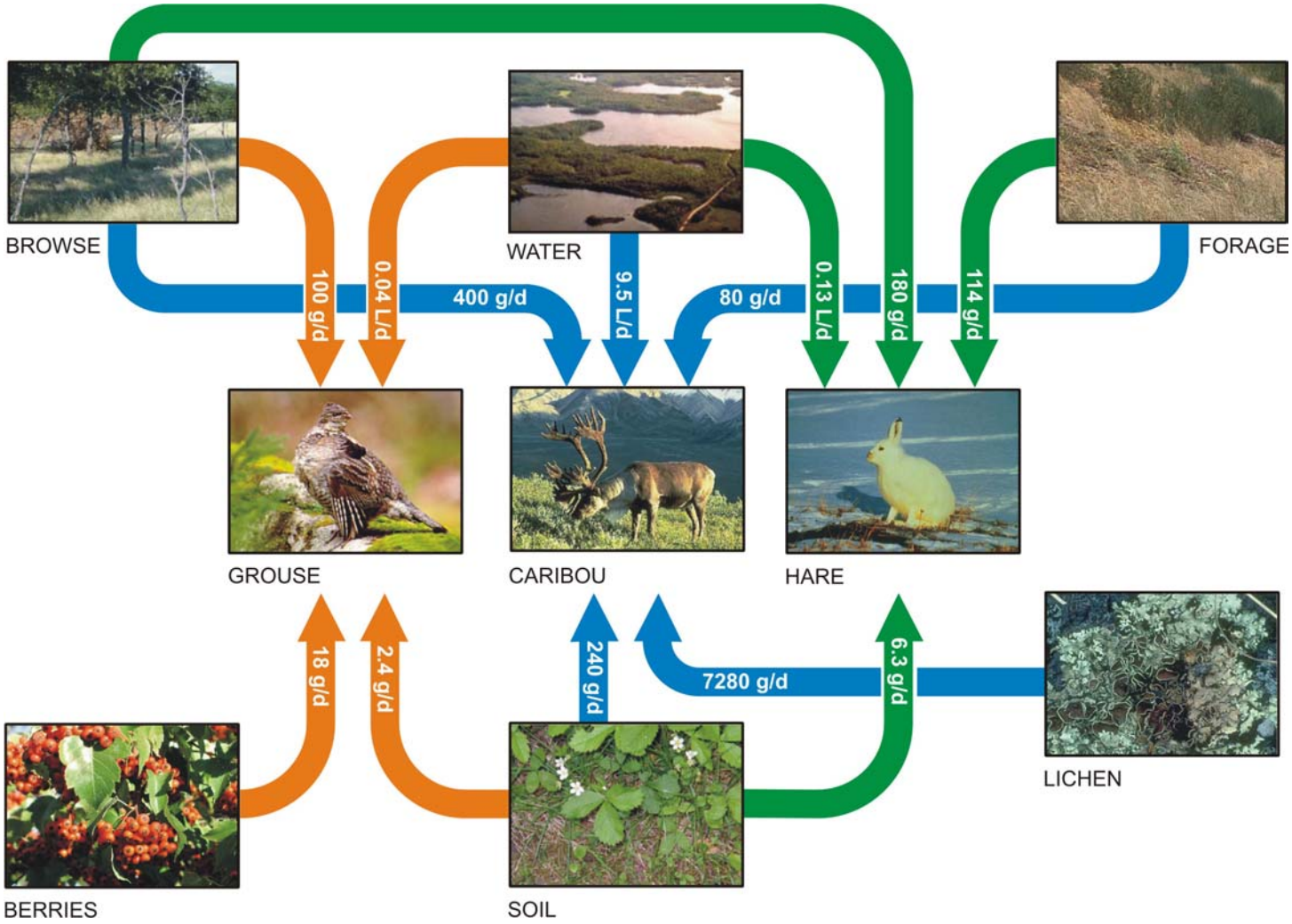


FIGURE 3.2-1 b
POTENTIAL PATHWAYS OF EXPOSURE FOR WOLF AND BLACK BEAR

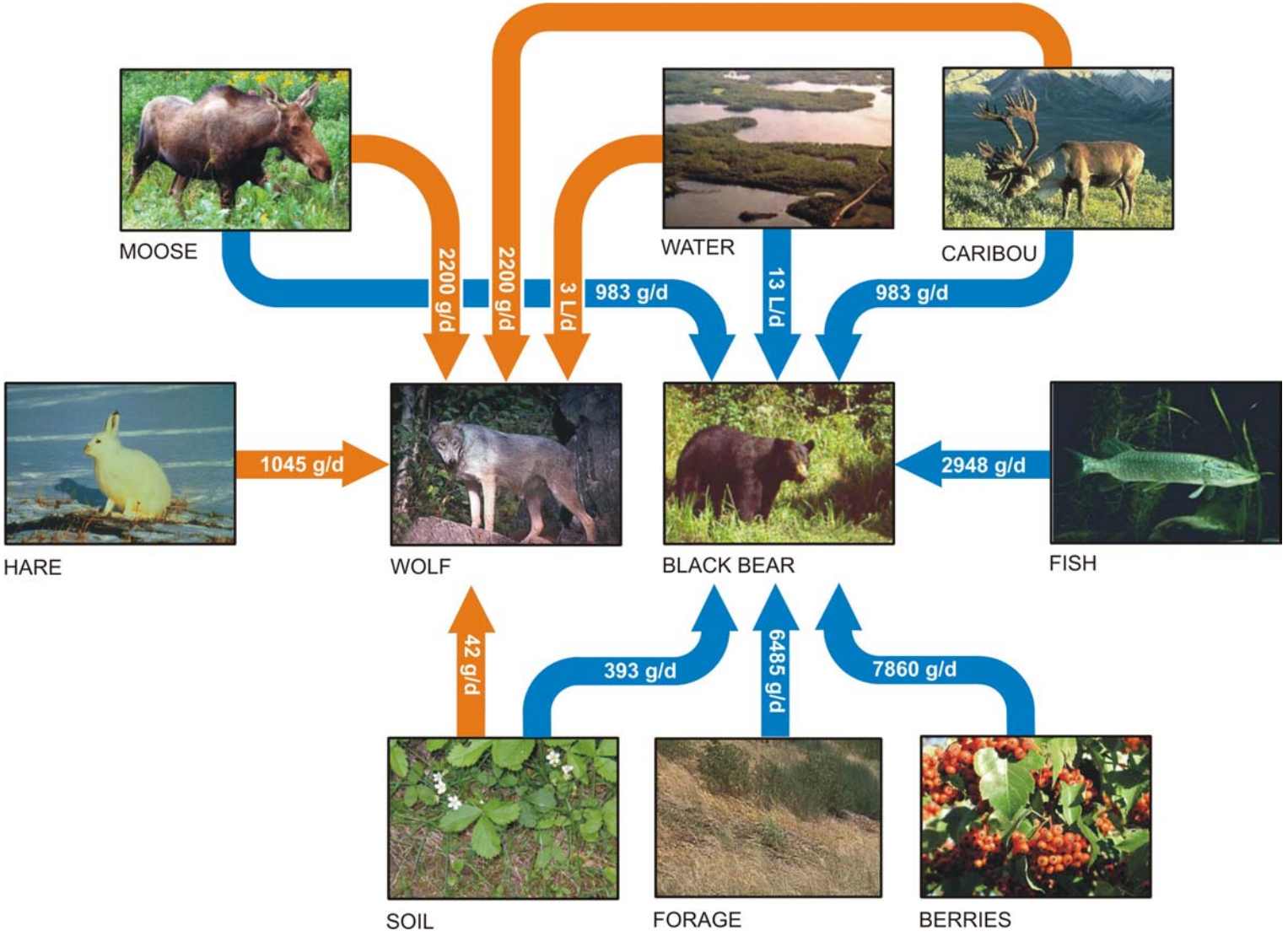


FIGURE 3.2-1 c
POTENTIAL PATHWAYS OF EXPOSURE FOR MINK, MOOSE AND MUSKRAT

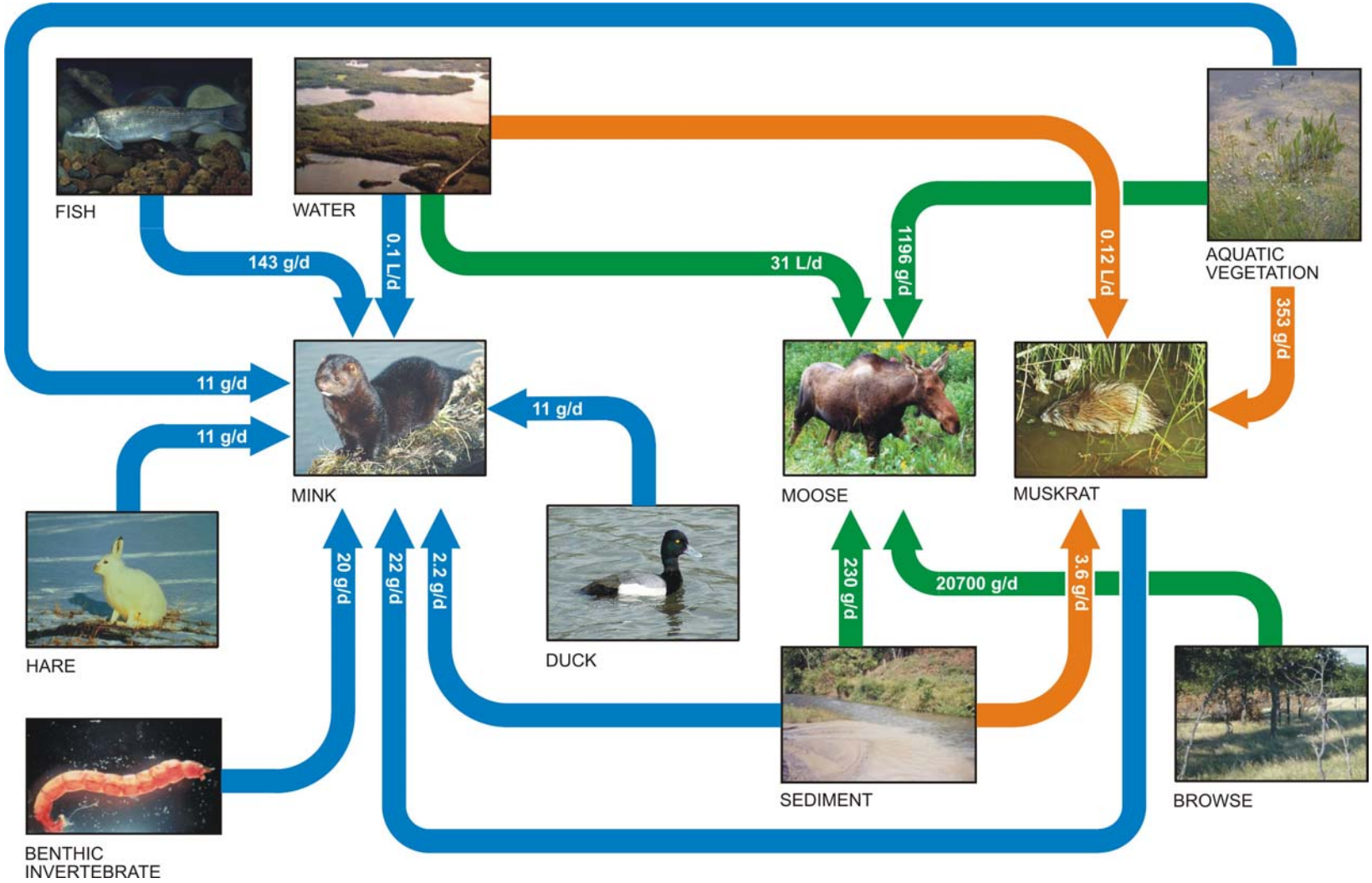
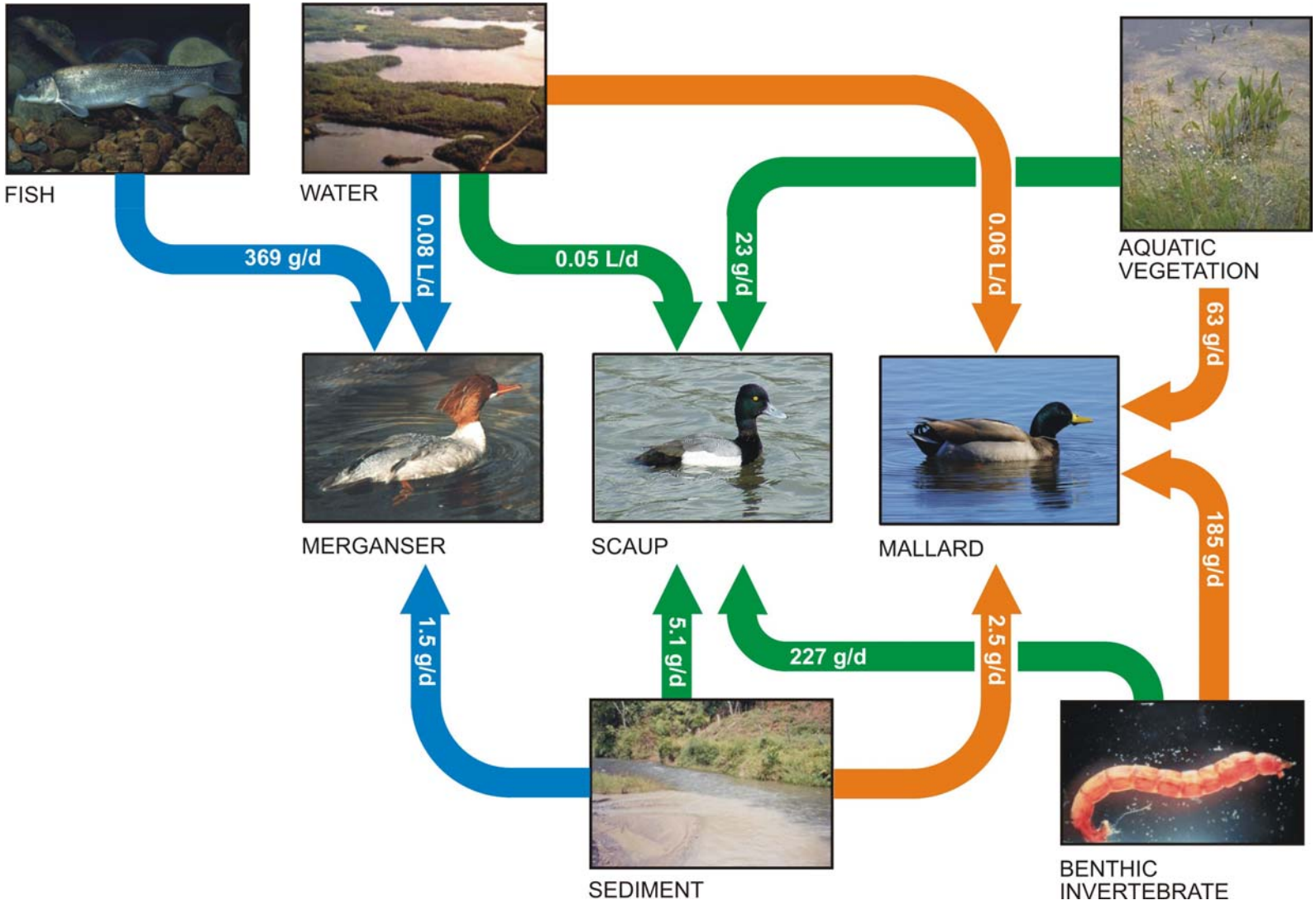


FIGURE 3.2-1 d
POTENTIAL PATHWAYS OF EXPOSURE FOR MALLARD, MERGANSER AND SCAUP



3.3 HUMAN RECEPTORS

The human receptors considered in the assessment were designed to represent a wide range of locations and activities. Four different locations were considered: the Giant Mine Townsite, Latham Island, the City of Yellowknife and in the Dettah community. These locations are shown on Figure 3.3-1. At each location, both adult and child receptors were considered. The designations assigned to the human receptors carried through the risk assessment were as follows:

- Receptor 1a and c – an adult and child at Giant Mine Townsite.
- Receptor 2a and c – an adult and child on Latham Island.
- Receptor 3a and c – an adult and child in the City of Yellowknife.
- Receptor 4a and c – an adult and child in the Dettah community.

Risk assessments generally rely on making inferences, assumptions and the use of models, which lead to uncertainties in the estimates. Some of the assumptions are related to how much an individual eats, where they get their food from and their life stage. In conducting this risk assessment, assumptions were made that are more likely than not to result in overestimations of exposure. However, to ensure that different types of exposures related to the geographical source of the dietary components, the amount of food consumed and the exposures of sensitive life stages such as toddlers were captured, a sensitivity analysis was conducted for the individual receptors as described in the following sections.

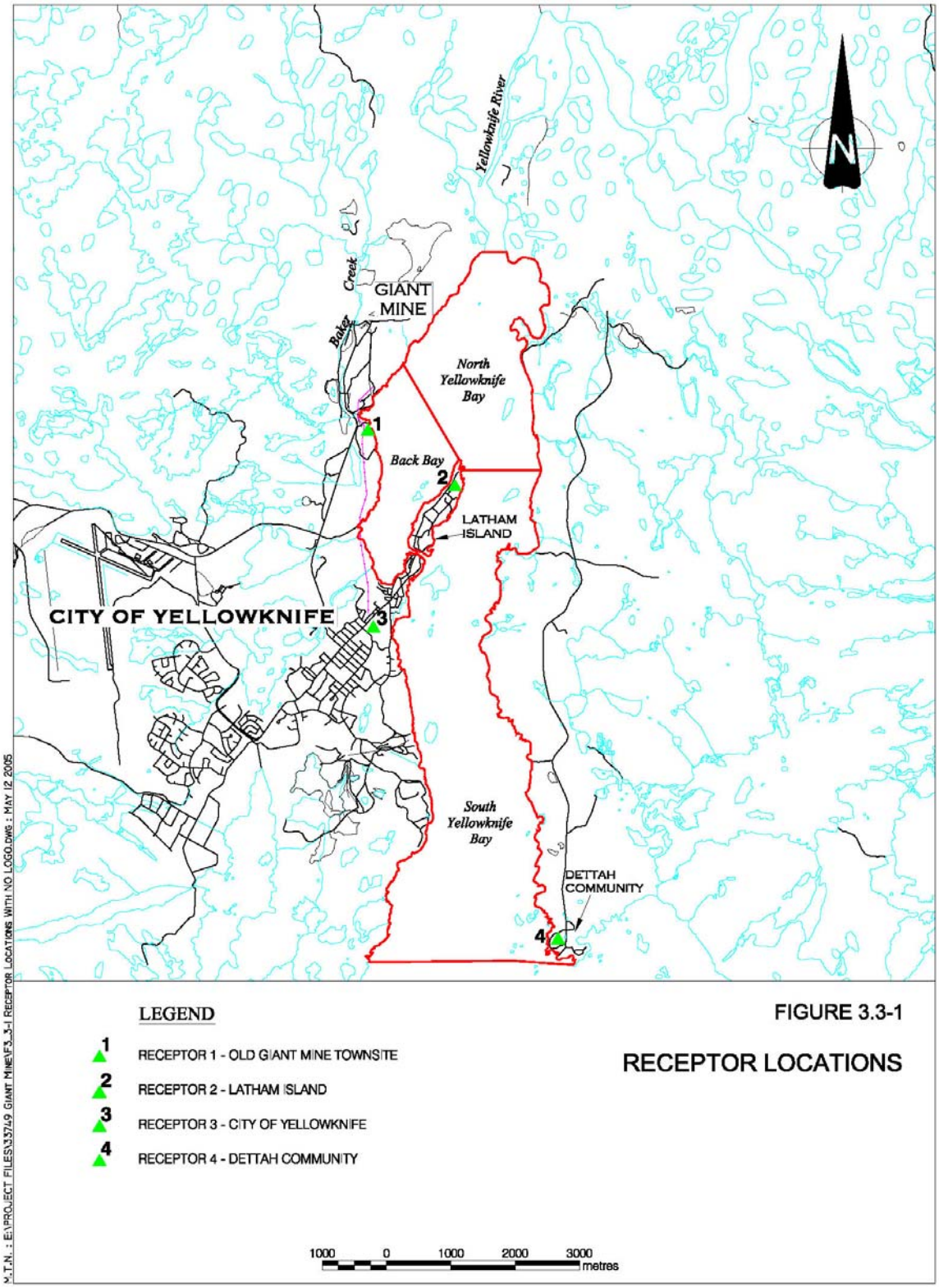


FIGURE 3.3-1

RECEPTOR LOCATIONS

3.4 HUMAN EXPOSURE PATHWAYS

3.4.1 Pathways Considered

The specific pathways of arsenic exposure to humans are shown on Figure 3.4-1 and include:

- ingestion of drinking water;
- uptake by fish of arsenic from the aquatic environment and consumption of contaminated fish flesh by human receptors;
- uptake by moose of arsenic from aquatic plants, browse, sediment and water and consumption of contaminated moose flesh and organs by the human receptors;
- uptake by spruce grouse and snowshoe hare of arsenic from browse (grouse), forage (hare), soil and water and consumption of contaminated grouse and hare flesh by the human receptors;
- uptake by mallard of arsenic from aquatic plants, sediments and water and consumption of contaminated mallard flesh by the human receptors;
- uptake by caribou of arsenic from lichen, soil and water and consumption of contaminated caribou flesh and organs by the human receptors;
- inhalation of arsenic present in the atmospheric environment by all human receptors;
- ingestion of medicinal teas by some of the receptors;
- ingestion of berries and garden produce grown in the study area by all human receptors; and,
- inadvertent ingestion of soil and associated arsenic present at the respective human receptor locations.

Based on discussions with the Giant Mine Remediation Project Team and members of the community, it was determined that most of the residents in the Yellowknife area are on municipally supplied water. Therefore, in the core analysis it was assumed that receptors obtain all their drinking water from the municipal supply.

In order to capture a range of potential exposures for the receptors, the geographical source of several of the dietary components was varied between the receptors. The receptors were assumed to derive many dietary components from sources in their immediate vicinity, while other components were obtained from the Giant Mine site. For example, large and small game in many cases were assumed to be captured on the Giant Mine site along the Baker Creek watershed. A summary of the geographical sources assumed for each dietary component for the different receptors is provided in Table 3.4-1.

FIGURE 3.4-1
POTENTIAL PATHWAYS OF EXPOSURES FOR HUMAN RECEPTORS

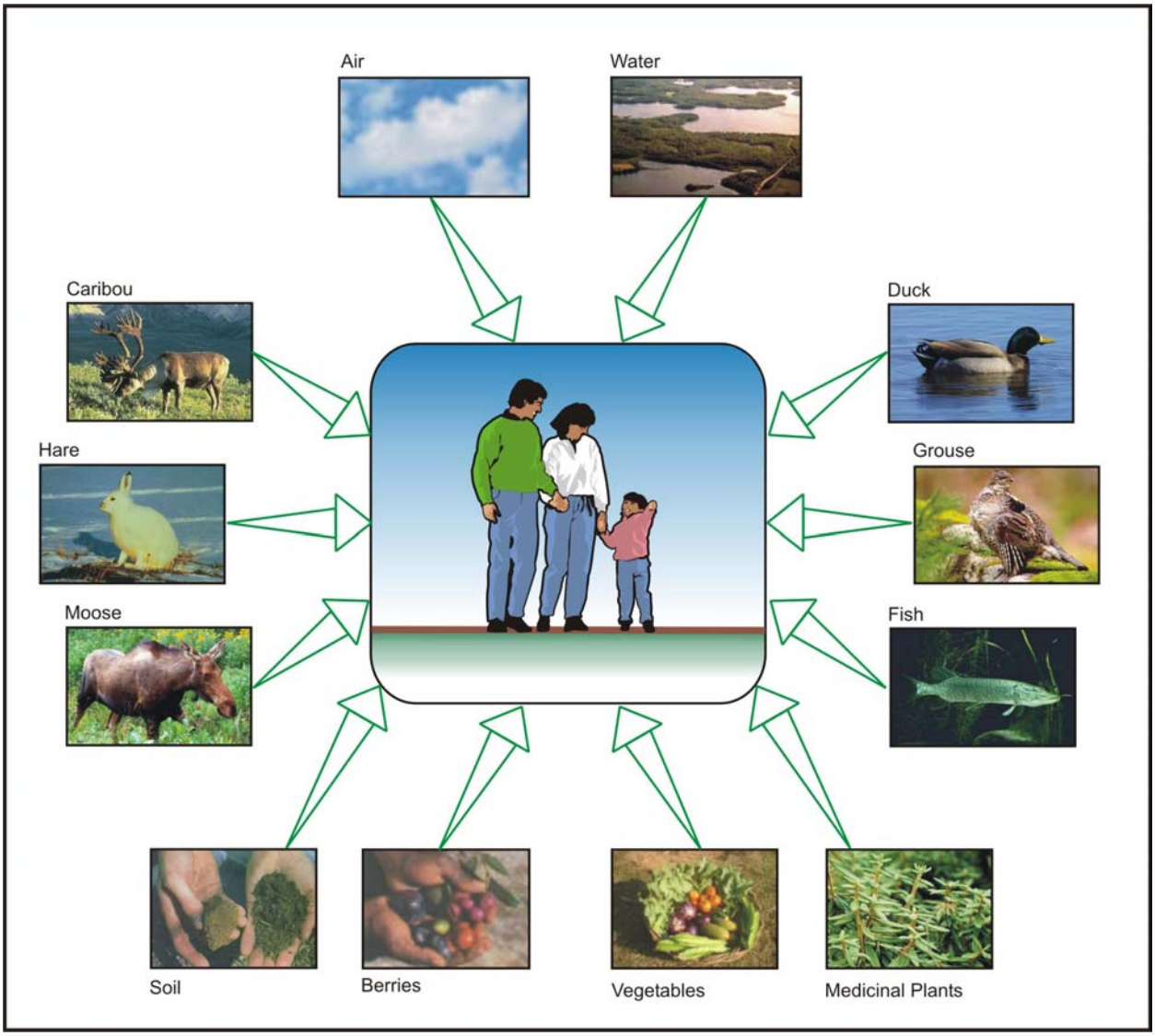


TABLE 3.4-1
PRIMARY GEOGRAPHICAL SOURCES OF DIETARY COMPONENTS FOR HUMAN RECEPTORS

Dietary Component	Receptor 1a and c Giant Mine Townsite	Receptor 2a and c Latham Island	Receptor 3a and c City of Yellowknife	Receptor 4a and c Dettah Community
Drinking Water	Municipal Supply	Municipal Supply	Municipal Supply	Municipal Supply
Soil	Giant Townsite	Latham Island	City of Yellowknife	Dettah Community
Garden Produce	Giant Townsite	Latham Island	City of Yellowknife	Dettah Community
Berries	Giant Mine Site	Latham Island	City of Yellowknife	Dettah Community
Large Game	Baker Creek	Baker Creek	Baker Creek	Dettah Community
Small Game	Baker Creek	Baker Creek	Baker Creek	Dettah Community
Ducks	Baker Creek/Back Bay	Back Bay	North Yellowknife Bay	South Yellowknife Bay
Fish	Back Bay	Back Bay	North Yellowknife Bay	South Yellowknife Bay
Medicinal Teas	-	Giant Mine Site	-	Dettah Community
Supermarket Foods	Imported	Imported	Imported	Imported

To account for the fact that some individuals may obtain one or more dietary components from alternative sources, sensitivity analyses were carried out for each receptor. In each sensitivity case, the geographical source of one of the dietary components was changed. The final sensitivity case examined the cumulative effects of alternative sources for the most exposed scenario.

For the receptors assumed to live on the Giant Mine Townsite, the assumptions related to source of backyard garden produce and the location of where fish were caught (Table 3.4-2) were changed in the sensitivity analyses. The sensitivity analyses considered the exclusion of backyard garden produce, and the consumption of fish from Baker Creek. Lastly, the sensitivity analysis also examined the impact of exposure to a toddler at this location. Thus, the assessment considers three life stages (toddler, child and adult). Exposures for infants and teens are captured within the range of receptors considered in this assessment.

TABLE 3.4-2
GEOGRAPHICAL SOURCES OF DIETARY COMPONENTS FOR THE SENSITIVITY ANALYSIS FOR GIANT MINE TOWNSITE RECEPTOR (1a and 1c)

Dietary Component	Dietary Component Sources		
	Primary Source	**Sensitivity Case 1**	**Sensitivity Case 2**
Drinking Water	Municipal Supply	Municipal Supply	Municipal Supply
Soil	Giant Town Site	Giant Town Site	Giant Town Site
Garden Produce	Giant Town Site	Giant Town Site	***Not Considered***
Berries	Giant Mine Site	Giant Mine Site	Giant Mine Site
Large Game	Baker Creek	Baker Creek	Baker Creek
Small Game	Baker Creek	Baker Creek	Baker Creek
Ducks	Baker Creek/Back Bay	Baker Creek/Back Bay	Baker Creek/Back Bay
Fish	Back Bay	***Baker Creek***	Back Bay
Supermarket Food	Imported	Imported	Imported

For the Latham Island receptors, the sensitivity analyses examined changing the source of drinking water from the municipal supply to Back Bay and the source of fish from Back Bay to Baker Creek (see Table 3.4-3).

TABLE 3.4-3
GEOGRAPHICAL SOURCES OF DIETARY COMPONENTS FOR THE SENSITIVITY ANALYSIS FOR LATHAM ISLAND RECEPTORS (2a and 2c)

Dietary Component	Dietary Component Sources			
	Primary Source	**Sensitivity Case 1**	**Sensitivity Case 2**	**Sensitivity Case 3**
Drinking Water	Municipal Supply	***Back Bay***	Municipal Supply	***Back Bay***
Soil	Latham Island	Latham Island	Latham Island	Latham Island
Garden Produce	Latham Island	Latham Island	Latham Island	Latham Island
Berries	Latham Island	Latham Island	Latham Island	Latham Island
Large Game	Baker Creek	Baker Creek	Baker Creek	Baker Creek
Small Game	Baker Creek	Baker Creek	Baker Creek	Baker Creek
Ducks	Back Bay	Back Bay	Back Bay	Back Bay
Fish	Back Bay	Back Bay	***Baker Creek***	***Baker Creek***
Medicinal Teas	Giant Mine Site	Giant Mine Site	Giant Mine Site	Giant Mine Site
Supermarket Foods	Imported	Imported	Imported	Imported

For the City of Yellowknife receptors, the sensitivity analyses examined the effects of obtaining berries from the mine area, fish from Baker Creek and drinking water from Back Bay (see Table 3.4-4). It was also assumed that some individuals may obtain their water from Back Bay.

TABLE 3.4-4
GEOGRAPHICAL SOURCES OF DIETARY COMPONENTS FOR THE SENSITIVITY ANALYSIS FOR CITY OF YELLOWKNIFE RECEPTORS (3a and 3c)

Dietary Component	Primary Source	Sensitivity Case 1	Sensitivity Case 2	Sensitivity Case 3	Sensitivity Case 4
Drinking Water	Municipal Supply	Municipal Supply	Municipal Supply	***Back Bay***	***Back Bay***
Soil	City of Yellowknife	City of Yellowknife	City of Yellowknife	City of Yellowknife	City of Yellowknife
Garden Produce	City of Yellowknife	City of Yellowknife	City of Yellowknife	City of Yellowknife	City of Yellowknife
Berries	City of Yellowknife	***Giant Mine Site***	City of Yellowknife	City of Yellowknife	***Giant Mine Site***
Large Game	Baker Creek	Baker Creek	Baker Creek	Baker Creek	Baker Creek
Small Game	Baker Creek	Baker Creek	Baker Creek	Baker Creek	Baker Creek
Ducks	North Yellowknife Bay	North Yellowknife Bay	North Yellowknife Bay	North Yellowknife Bay	North Yellowknife Bay
Fish	North Yellowknife Bay	North Yellowknife Bay	***Baker Creek***	North Yellowknife Bay	***Baker Creek***
Supermarket Foods	Imported	Imported	Imported	Imported	Imported

For the Dettah receptors, the sensitivity analyses examined the effects of obtaining game from the mine area and drinking water from South Yellowknife Bay (see Table 3.4.5).

TABLE 3.4-5
GEOGRAPHICAL SOURCES OF DIETARY COMPONENTS FOR THE SENSITIVITY ANALYSIS FOR DETTAH COMMUNITY RECEPTORS (4a and 4b)

Dietary Component	Primary Source	Sensitivity Case 1	Sensitivity Case 2	Sensitivity Case 3
Drinking Water	Municipal Supply	***South Yellowknife Bay***	Municipal Supply	***South Yellowknife Bay***
Soil	Dettah Community	Dettah Community	Dettah Community	Dettah Community
Garden Produce	Dettah Community	Dettah Community	Dettah Community	Dettah Community
Berries	Dettah Community	Dettah Community	Dettah Community	Dettah Community
Large Game	Dettah Community	Dettah Community	***Baker Creek***	***Baker Creek***
Small Game	Dettah Community	Dettah Community	***Baker Creek***	***Baker Creek***
Ducks	South Yellowknife Bay	South Yellowknife Bay	South Yellowknife Bay	South Yellowknife Bay
Fish	South Yellowknife Bay	South Yellowknife Bay	South Yellowknife Bay	South Yellowknife Bay
Medicinal Teas	Dettah Community	Dettah Community	Dettah Community	Dettah Community
Supermarket Foods	Imported	Imported	Imported	Imported

Assumed Receptor Characteristics

The adult and child receptors located on Latham Island and in the Dettah Community, were assumed to have a diet mainly consisting of traditional or country food that has been obtained from Baker Creek, Back Bay, South Yellowknife Bay or the area around the Dettah Community. Food intake rates for these receptors (2a,c and 4a,c) were based on data from Receveur *et al.* (1996) and (1998) and are presented in Appendix D.

Receptors 3a and 3c, located in Yellowknife, and Receptors 1a and 1c at the Giant Mine Townsite, were assigned dietary characteristics that are similar to those of the general Canadian population who purchase most of their food items at grocery stores. The average total meat, fish and poultry consumption of a typical Canadian adult is estimated to be in the order of 277 g/d. For these receptors it was assumed that approximately 1/3 of the meat, fish and poultry intake (89.8 g/d) is obtained from country food (local) sources (Receveur *et al*. 1996) whereas the rest of the meat, fish and poultry intake consists of store bought food (see Table 3.4-9). Food intake rates for these receptors were derived from Richardson (1997) and Receveur *et al.* (1996) and are summarized in Appendix D. Further details on the data sources for the intake rates for each exposure pathway are summarized for all receptors at the end of this section (see Table 3.4-10).

Receptor characteristics for the toddler receptor were required in the sensitivity analysis, and were derived using the same methods and assumptions as for the child receptor. These characteristics are also summarized in the following sections.

As discussed in the previous section, the primary source of drinking water for all receptors was assumed to be the municipal supply.

3.4.1.1 Intakes of Meat, Fish, Poultry and Eggs

The intakes of traditional food were obtained from a study of Yellowknife Dene (Receveur *et al.* 1998). This report presents a secondary data analysis of dietary interviews completed in the communities of Dettah and Latham Island between 1993 to 1995 by Mackenzie Regional Health Services. The survey was based on 24-hour dietary recall methods and provides information on the percent of population consuming traditional food. The use of the 24-hour recall method is limited since it does not allow for an accurate estimation of intake on an annual basis. Additionally, this study is also limited in that it does not include information on serving size. To augment the data available for the Yellowknife Dene, serving sizes and yearly frequencies were estimated from the larger Dene/Metis survey (Receveur *et al.* 1996).

The total intake of meat, fish, poultry and eggs for several Dene communities is included in Table 3.4-6.

TABLE 3.4-6
TOTAL MEAT, FISH, POULTRY AND EGG INTAKE BY ADULTS (g/d)

	Sahtu	Dogrib	Deh-Cho	South-Slave
Traditional Sources	456	358	139	269
Market Food	203	150	220	205
Total	659	508	358	474

Source: based on Receveur *et al.* (1996).

From a review of the available information, Receveur *et al.* (1998) concluded that the percentage of the Yellowknife Dene population consuming each traditional food species was generally similar to that reported by Dogrib participants. Therefore, the information on the Dogrib community was used to estimate the total intake by Yellowknife Dene (Receveur *et al.* 1998). The dietary survey for the Dogrib community only focuses on adult intakes. Therefore, for this assessment it was assumed that children consumed the same diet as adults adjusted to account for differences in total intake rates. In a Canada wide survey carried out for Health Canada (1994) it was estimated that the total meat, fish and poultry intake for a child 5 to 11 years of age was approximately 75% of that for an adult. For a toddler 0.5 to 4 years old, this difference is 50%. These ratios were applied to the meat, fish and poultry intakes of children and toddlers in this assessment.

Intakes of Fish

The study on the Yellowknife Dene (Receveur *et al.* 1998) provides an estimate of average daily fish consumption for fish consumers (Table 21 of the report). The report states that the estimates are based on the serving size and frequency of consumption from the Dogrib data in Receveur *et al.* (1996). There is no information in the Receveur *et al.* (1996) report on serving size for several of the fish types included for the Yellowknife Dene which leads to uncertainty in the determination of fish consumption. The total average fish intake presented in Table 21 of Receveur *et al* (1998) is 167 g/d for the Yellowknife Dene (consumers only).

In the Receveur *et al.* (1996) report, the intake of traditional foods is provided for the whole population (Table 13 of the report). This table provides a fish intake of 67 g/d for the Dogrib communities. An estimated average fish consumption of 84 g/d for the Yellowknife Dene community was determined using the information in Receveur *et al.* (1998). For comparison purposes, the average adult intake of fish in the Hatchet Lake study (CanNorth 2000) for First Nations communities in northern Saskatchewan was 55 g/d.

The Tier 2 risk assessment was conducted using the average fish intake rate from the Yellowknife Dene to estimate fish intakes for the Latham Island and Dettah Community

receptors. However, given that there is a large difference between the intake rates for fish consumers only versus the whole population, part of the sensitivity analysis explored the impact the higher fish intake rates would have on total arsenic exposure for Latham Island and Dettah receptors with a high fish diet. In the primary assessment, individuals at the Latham Island (Receptor 2a) and Dettah community (Receptor 4a) were assumed to consume 84 g/d of fish. To account for higher fish intakes, the sensitivity analysis considered individuals who consumed 167 g/d of fish.

Thus, a receptor who consumed a high proportion of fish (167 g/d) was chosen for the sensitivity analysis while the Tier 2 risk assessment receptors were assumed to consume less fish (84 g fish/d). The fish intakes for a child and toddler were assumed to be 75% and 50% of the adult intake respectively, as discussed in the previous section.

In order to account for fish consumption within the probabilistic framework, a lognormal distribution was selected based on work by Richardson (1997), which indicates that the fish intake of First Nations People can be defined using a lognormal distribution.

Intakes of Large Game, Small Game and Ducks

The total meat, fish, poultry and egg intakes by adults provided in Table 3.4-6 for the Dogrib can further be broken down into the various types of food consumed. Given that the Dogrib community has similar characteristics to the native communities in the Yellowknife area, the break down of the sources of traditional food provided in Receveur *et al.* (1996) for the Dogrib community was used in this assessment. In the assignment of intake rates, consideration was also given to the results of other relevant studies (CanNorth 2000). The information contained in the CanNorth (2000) study was particularly interesting as data were collected based on a four-week period. This study was used to augment the information from the Northwest Territories Dene Studies (Receveur *et al.* 1996, 1998), which were based on 24-hour recall methods.

Table 3.4-7 provides the dietary breakdown of the meat and poultry sources.

TABLE 3.4-7
DIETARY BREAKDOWN OF MEAT AND POULTRY SOURCES USED IN THIS ASSESSMENT

Meat/Poultry	Fraction	Average Adult Daily Intake (g/d)
Caribou	0.79	352
Moose	0.01	5
Small mammals	0.01	4.6
Game birds	0.002	1.1
Ducks	0.004	2.0
Total meat and poultry	-	364.7

The total intakes from meat, poultry and fish for Receptors 2 and 4 were assigned the following lognormal distributions, based on the dietary survey discussed above and the standard deviation from information provided in Richardson (1997) for Canadian populations. The data from Richardson (1997) is based on a 24-hr recall survey. The distributions (in g/day) were assigned the following attributes (geometric mean, geometric standard deviation, minimum, maximum):

Toddler – LN (215, 1.5, 69.2, 788)

Child – LN (306, 1.5, 98.2, 1119)

Adult – LN (414, 1.5, 133.1, 1516)

The results of the surveys of the Dogrib and Dene communities suggest that the intake of caribou meat is consistent with a lognormal distribution. In contrast, the intakes of ducks, land birds (grouse) and mammals (hare) are not well characterized by the lognormal distribution. However, because these species comprise a small portion of the diet (< 2%) for an average consumer, they were assigned lognormal distributions for the purposes of this assessment.

Based on the assigned distribution for meat, poultry and fish, the total protein intake can be determined. Using information provided on the U.S. Food and Drug Administration website, the protein content of food items, the mean intake for individuals at Latham Island and the Dettah community can be calculated as 139 g protein/d. At the 95th percentile level, the protein intake equals 249 g/d. This protein intake is higher and covers a broader range than the protein intakes reported for the Yellowknife Dene shown below (Receveur *et al.* 1998, Table 8):

Gender	Age	Protein Intake
Male	20 – 40 yrs	113 ± 13 g/d
Male	41+ yrs	107 ± 11 g/d
Female	20 – 40 yrs	91 ± 7 g/d
Female	41+ yrs	129 ± 14 g/d

As described previously, it was assumed that receptors at the Giant Mine Townsite and the City of Yellowknife obtain ⅓ of their food from local sources. Therefore, the total intakes from meat, poultry and fish for individuals at the Giant Mine Townsite and the City of Yellowknife were assigned the following lognormal distributions (in g/d):

Toddler – LN(46.7, 1.5, 13.8, 158)

Child – LN(66.3, 1.5, 19.7, 223.8)

Adult – LN(89.8, 1.5, 26.6, 303.2)

The child and toddler intake values are based on 75% and 50% of the adult intakes respectively.

3.4.1.2 Vegetation

Berries

It was assumed that in the Yellowknife area, blueberries, cranberries and cloud berries are eaten by the local population. This agrees with the information provided in a survey of other Dogrib communities (Receveur *et al.* 1996).

A total berry intake rate of 3108 g/person/y (8.7 g/d) is recommended in the draft report on arsenic exposure from consumption of berries in the Akaitcho Territory (Akaitcho 2000). The intake of berries for a child was 6.4 g/d, and this value was also used for the toddler. These values were used in the current assessment.

Vegetables

It was assumed that humans in the Yellowknife area consumed vegetables (both above ground and below ground) from backyard plots. The Ontario Ministry of the Environment in their document entitled "*Soil Investigation and Human Health Risk Assessment for the Rodney Street Community, Port Colborne"* (October 2001), derived the percentage of the vegetables intake that an individual would obtain from a backyard garden. From their calculations they determined that 7.3% of the total annual consumption comes from backyard gardens. This value was adopted in the assessment. This is not an unreasonable assumption given that there is a short growing season in Yellowknife due to its northern location. However, a triangular distribution was used in the pathways model spanning from 0 to 10% to cover a range of intake levels. The intakes of above ground and below ground vegetables were obtained from Richardson (1997). Table 3.4-8 summarizes the distributions from Richardson (1997). In this assessment, these values were prorated for local vegetable intake by the factor of 7.3% discussed above.

TABLE 3.4-8
SUMMARY OF DISTRIBUTIONS FOR ABOVE- AND BELOW-GROUND VEGETABLES

	Mean (g/d)[a]	Distribution[b]
Above Ground Vegetables		
Toddler, Child	98	LN(65, 2.5, 10.4, 406)
Adult	137	LN(99, 2.2, 20.5, 479)
Below Ground Vegetables		
Toddler, Child	161	LN(119, 2.2, 24.6, 576)
Adult	188	LN(147, 2.0, 36.8, 588)

Note: a – The default value is based on the arithmetic mean value presented in Richardson (1997)
b – For probabilistic assessment a lognormal distribution was used where LN(geometric mean, geometric standard deviation, minimum, maximum).

3.4.1.3 Medicinal Tea Intake

First Nations People generally use Labrador tea and other teas made from natural plants for medicinal purposes. The Dene/Metis dietary survey (Receveur *et al.* 1996), for the area only, indicated that individuals consume Labrador Tea 0.3 days per week. The survey does not provide the amount of Labrador tea or other medicinal teas that are consumed.

In an attempt to determine the amount of medicinal tea consumed by members of the community, a web search was undertaken. The web search indicated that "*Labrador Tea contains small amounts of the toxin andromedotoxin which can cause headaches, cramps, paralysis and intestinal problems if too much is consumed. As a general rule, this tea should be consumed in moderation. One cup is often considered the safe amount.*" - http://www.laurentiancenter.com/plantkey/plants/labradortea.html

Therefore, it was assumed in this assessment that an adult (70kg body weight) would consume 250 mL (1 cup) of medicinal teas for 0.3 days a week. This equates to a consumption rate of 0.04 cups/day.

3.4.1.4 Water Intake

The water intakes for an adult and child were obtained from the *"Compendium of Canadian Human Exposure Factors for Risk Assessment"* (Richardson, 1997). The average water intakes for an adult, child (5 to 11 years of age) and toddler (0.5 to 4 years of age) is estimated to be 1.5 L/d, 0.8 L/d and 0.6 L/d respectively. Lognormal distribution functions were applied to the water intakes as follows:

Toddler – LN (0.50, 1.84, 0.15, 1.69)

Child – LN (0.72, 1.49, 0.32, 1.6)

Adult – LN (1.32, 1.65, 0.48, 3.6)

These distributions in the water consumption rates were used in the probabilistic pathways assessment.

3.4.1.5 Air Intake

The air intakes for an adult and child were obtained from the *"Compendium of Canadian Human Exposure Factor for Risk Assessment"* (Richardson 1997). The average daily air intakes for an adult, child and toddler are estimated to be 15.8 m^3/d, 14.5 m^3/d and 9.3 m^3/d, respectively. The distribution functions applied to the air intakes were lognormal, based on the geometric mean and are as follows:

Toddler – LN (8.94, 1.31, 5.21, 15.3)

Child – LN (14.15, 1.25, 9.1, 22.1)

Adult – LN (15.33, 1.27, 9.5, 24.7)

3.4.1.6 Soil Intake

Soil intake rates were obtained from Health Canada (2003) and were based on the information obtained from CCME (1996) and the Massachusetts Department of Environmental Protection (MADEP 2002). The mean daily soil intake reported for an adult was 20 mg/d, and, for a child, the intakes ranged from 20 mg/d to 150 mg/d. The mean soil intake for a toddler is reported to be 80 mg/d. For the purposes of this assessment, it was assumed that the child soil intake rate was equivalent to that of the toddler. In this assessment, we have conservatively selected a mean soil intake of 100 mg/d to represent a toddler and a child exposures. The distribution functions applied to the soil intake were based on data obtained from Health Canada (1994), the U.S. EPA (1997) and the Ontario Ministry of the Environment (2001). The daily soil intake range reported for an adult was 20 to 50 mg/d, and, for a child, the intakes ranged from 50 to 150 mg/d. The distribution function derived from this data for the adult was uniform, and a triangular distribution was used for the toddler and child as follows:

Toddler, Child – T (50, 100, 150)

Adult – U (20, 50)

Where: triangular distribution - T(minimum, mode, maximum) and uniform distribution - U(minimum, maximum)

3.4.1.7 Body Weight

The body weight (bw) of a child and adult are also necessary in order to calculate a daily intake (mg/(kg (bw) d)). In this assessment, the body weights used for the toddler, child and adult receptors were 16.5 kg, 32.9 kg and 70.7 kg, respectively (Richardson 1997).

3.4.1.8 Summary of Receptor Characteristics

The nominal amount of vegetables, fruits and meat that were assumed to be consumed by the adult and child receptors in the area are summarized in Table 3.4-9. For this assessment, it was assumed that all of the traditional foods consumed by Receptors 2 and 4 (i.e., Latham Island and Dettah community receptors) comprised animal muscle and organs. Table 3.4-10 provides a summary of the intake rates for a toddler that were used in the sensitivity analysis.

As discussed above, different literature sources were used to derive intake rates for each of the four receptors. No single data source provided all required information; therefore, a summary of literature sources used for the various intake rates is summarized in Table 3.4-11.

Table 3.4-12 summarizes the nominal intakes for the sensitivity analysis involving the higher fish intakes. As discussed previously, the sensitivity analysis only refers to the individuals at Latham Island and the Dettah community under the assumptions described in Table 3.4-1.

The air, water, soil, fish, berry and vegetable concentrations used in the assessment were provided in Section 2. The meat and poultry concentrations were obtained from the pathways model as described in Appendix D.

TABLE 3.4-9
AVERAGE AIR, WATER AND LOCAL FOOD INTAKE RATES USED FOR ADULT AND CHILD HUMAN RECEPTORS IN PATHWAYS MODELLING

	Receptor			
	Giant Mine Site and City of Yellowknife (1a and 3a - Adult)	**Giant Mine Site and City of Yellowknife (1c and 3c - Child)**	**Latham Island and Dettah Community (2a and 4a - Adult)**	**Latham Island and Dettah Community (2c and 4c - Child)**
Air (m^3/d)	15.8	14.5	15.8	14.5
Water (L/d)	1.5	0.8	1.5	0.8
Soil Intake (mg/d)	20	80	20	80
Local Meat (g/d)				
Caribou	52.8	39.0	352	261
Moose	1.2	0.9	5	3.7
other small mammals	1.5	1.1	4.6	3.4
Local Poultry (g/d)				
Ground birds	0.34	0.25	1.1	0.8
water birds	0.63	0.46	2.0	1.5
Local Fish (g/d)	33.4	24.7	84	62
Total local meat, fish and poultry (g/d)	89.9	66.3	449	332
Other (g/d)				
Berries	8.7	6.4	8.7	6.4
above-ground vegetation	137	98	137	98
below-ground vegetation	188	161	188	161

Note: Adult Weight = 70.7 kg; Child Weight = 32.9 kg.

TABLE 3.4-10
AVERAGE AIR, WATER AND LOCAL FOOD INTAKE RATES USED FOR TODDLER HUMAN RECEPTORS IN PATHWAYS MODELLING

	Receptor	
	Giant Mine Site and City of Yellowknife (1t and 3t - Toddler)	**Latham Island and Dettah Community (2t and 4t - Toddler)**
Air (m^3/d)	9.3	9.3
Water (L/d)	0.6	0.6
Soil Intake (mg/d)	80	80
Local Meat (g/d)		
Caribou	27.5	183
Moose	0.62	2.6
other small mammals	0.78	2.4
Local Poultry (g/d)		
Ground birds	0.18	0.57
water birds	0.33	1.0
Local Fish (g/d)	17.4	44
Total local meat, fish and poultry (g/d)	46.7	234
Other (g/d)		
Berries	6.4	6.4
above-ground vegetables	98	98
below-ground vegetables	161	161

Note: Toddler Weight = 16.5 kg.

TABLE 3.4-11
LITERATURE SOURCES OF INTAKE RATES USED FOR HUMAN RECEPTORS IN PATHWAYS MODELLING

	Receptor			
	1a,c (Giant Mine Townsite)	**2a,c (Latham Island)**	**3a,c (City of Yellowknife)**	**4a,c (Dettah)**
Diet	High Fish	Avg. Fish	High Fish	Avg. Fish
Air (m^3/d)	Richardson 1997	Richardson 1997	Richardson 1997	Richardson 1997
Water (L/d)	Richardson 1997	Richardson 1997	Richardson 1997	Richardson 1997
Soil Intake (mg/d)	Health Canada 2003 (det.) Health Canada 1994, U.S. EPA 1997, MOE 2001 (prob.)	Health Canada 2003 (det.) Health Canada 1994, U.S. EPA 1997, MOE 2001 (prob.)	Health Canada 2003 (det.) Health Canada 1994, U.S. EPA 1997, MOE 2001 (prob.)	Health Canada 2003 (det.) Health Canada 1994, U.S. EPA 1997, MOE 2001 (prob.)
Total local meat, fish and poultry intake (g/d)	Total rate from Richardson 1997, breakdown from Receveur *et al.* 1996	Total and breakdown based on Receveur *et al.* 1996, 1998	Total rate from Richardson 1997, breakdown from Receveur *et al.* 1996	Total and breakdown based on Receveur *et al.* 1996, 1998
Berries (g/d)	Akaitcho 2000	Akaitcho 2000	Akaitcho 2000	Akaitcho 2000
Above-ground vegetables (g/d)	Total rate from Richardson 1997, breakdown from MOE 2001	Total rate from Richardson 1997, breakdown from MOE 2001	Total rate from Richardson 1997, breakdown from MOE 2001	Total rate from Richardson 1997, breakdown from MOE 2001
Below-ground vegetables (g/d)	Total rate from Richardson 1997, breakdown from MOE 2001	Total rate from Richardson 1997, breakdown from MOE 2001	Total rate from Richardson 1997, breakdown from MOE 2001	Total rate from Richardson 1997, breakdown from MOE 2001
Supermarket Foods (g/d)	Total from Health Canada 1993 for Canadian population subtracting local modelled food intakes	Receveur *et al.* 1996	Total from Health Canada 1993 for Canadian population subtracting local modelled food intakes	Receveur *et al.* 1996

Notes: "det." Refers to deterministic distribution.
"prob." Refers to probabilistic distribution.
- When separate sources of deterministic and probabilistic distributions are not noted, the source of deterministic and probabilistic distributions is the same.
- Further information is available in Appendix D.

TABLE 3.4-12
AIR, WATER AND FOOD INTAKE RATES USED FOR SENSITIVITY ANALYSIS FOR FISH FOR LATHAM ISLAND AND DETTAH COMMUNITY RECEPTORS

	Receptor Location	
	Latham Island and Dettah Community (2a and 4a - Adult)	**Latham Island and Dettah Community (2c and 4c - Child)**
Air (m^3/d)	15.8	14.5
Water (L/d)	1.5	0.8
Soil Intake (mg/d)	20	80
Meat (g/d)		
Caribou	264	195
Moose	5.9	4.4
Pther small mammals	7.3	5.4
Poultry (g/d)		
Ground birds	1.7	1.2
Water birds	3.1	2.3
Fish (g/d)	167	124
Total meat, fish and poultry (g/d)	449	332
Other (g/d)		
Berries	8.7	6.4
Above-ground vegetables	137	98
Below-ground vegetables	188	161

4.0 EXPOSURE ASSESSMENT

The exposure assessment phase of ecological and human health risk assessments entails identification of pathways of exposure and the quantification of exposures for the selected receptors. The pathways models and input parameters used to quantify exposure in the assessment are detailed in Appendix D. One of the principal sources of arsenic release to the environment associated with current or future conditions at the Giant Mine site is via the discharge of treated mine water in the short-term to Baker Creek, as currently practiced, and the release of treated mine water in the long-term to Back Bay, once a new treatment facility is constructed. In addition, even after the Giant Mine site is remediated, there will be elevated levels of arsenic in vegetation and soils on site and in surface runoff to Baker Creek. These sources of arsenic were included in the overall assessment, as were sources from Yellowknife study area in general and in market foods brought into the Yellowknife community from remote locations as discussed in the previous sections. Results of the exposure assessment are provided in Section 6.0. Appendix G presents a sample calculation to illustrate the steps involved in the exposure assessment.

4.1 WATER AND SEDIMENT QUALITY MODELLING

The dispersion of arsenic in the aquatic environment for the remediation scenario described in Section 2.2 was undertaken using a lake dispersion model (LAKEVIEW) to predict arsenic levels throughout Back Bay and Yellowknife Bay. The model accounts for exchange between adjacent segments of a water body, removal of contaminants from the water column on settling matter and via diffusion from the water column to the sediment porewater, and exchange between bottom sediments and the water column. The model also accounts for arsenic speciation in the lake sediments, based on chemical (solid-liquid) equilibrium and adsorption processes, as described in detail in Appendix B. The model uses equations that are widely accepted in the scientific literature.

Application of the model for prediction of arsenic levels in the water columns and sediments of Back Bay and Yellowknife Bay first involved calibrating the model to past monitoring data. For assessment purposes, the key area was subdivided into three segments (Segment 1 – Back Bay, Segment 2 – North Yellowknife Bay, Segment 3 – South Yellowknife Bay) as shown on Figure 4.1-1.

The decision to sub-divide the bay into three segments was based on physical considerations and the fact that this is an assessment of long-term exposures. Latham Island is a natural protrusion into the bay that separates Back Bay from Yellowknife Bay. Most of the surface water runoff into Back Bay originates from Baker Creek. North Yellowknife Bay was defined as the second segment due to the fact that water quality in this segment is strongly influenced by the inflow

from the Yellowknife River, which drains a large watershed to the north and east of the study area. South Yellowknife Bay by contrast, receives little direct surface water runoff, but is influenced by water exchange with North Yellowknife Bay to the north and Great Slave Lake to the south. It is acknowledged that from time-to-time aquatic organisms (e.g. fish) or people could be exposed to high arsenic levels for short periods of time at specific locations in these water bodies (e.g. at the mouth of Baker Creek where it flows into Back Bay). However, further subdivision of the three segments was determined not to be necessary as the objective of the assessment was focussed on long-term average exposures. For example, fish caught in Back Bay are likely to move throughout the bay. Hence, their exposure would best be represented by the average arsenic concentration in the bay, not the concentration in any particular area of the bay. Also, due to the effects of wind and current movement, the arsenic level in any particular area will change continuously. Thus, someone living on Latham Island, on the opposite side of Back Bay from the Giant Mine site, who might obtain his/her drinking water from the bay would be exposed to an average arsenic concentration.

In calibrating the model, best estimates (i.e. nominal values) of the input parameters were obtained by simultaneously comparing predicted and measured water and sediment concentrations in each of the bay segments. The calibration comprised quantifying: sediment porewater, surface water and sediment solids interactions; arsenic loadings; and convective/dispersive transport of dissolved arsenic in surface water. The results of the simulation using the calibrated parameter values showed good agreement between predicted and measured surface water and sediments of each of the three segments of Yellowknife Bay. The model calibration procedure and results are discussed in Appendix B, Section B6.0.

Once calibrated, the model was then applied to predict future arsenic concentrations in Back Bay and the northern and southern segments of Yellowknife Bay. Model simulations were carried out for a period of 100 years, until a quasi steady-state was obtained. To account for uncertainty in past and future arsenic loadings and in several of the model input parameters, the model was run in a probabilistic mode. In other words, uncertainty in the arsenic loads was directly accounted for by specifying probability distributions to the load estimates. In running the model probabilistically, a different load was used in each trial by sampling the input load distribution using a Monte Carlo random sampling protocol to generate a distribution of predicted outcomes.

The model calibration procedure provided an estimate of the historical loads along with an uncertainty estimate. A normal distribution was used to represent the variability and uncertainty in the estimated historical and future loads. A summary of the distributions used in the modelling is provided in Table 4.1-1.

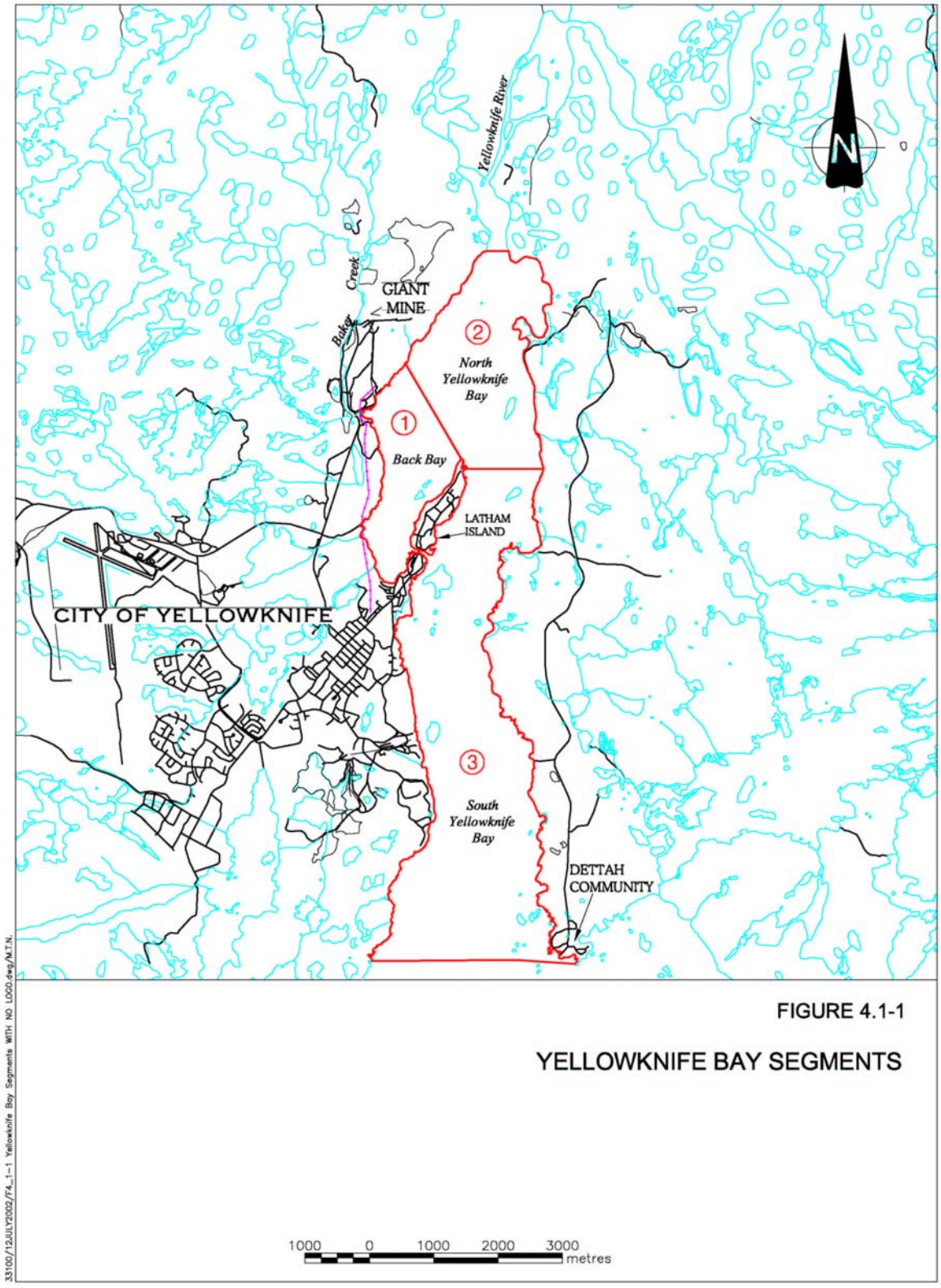


FIGURE 4.1-1

YELLOWKNIFE BAY SEGMENTS

TABLE 4.1-1
SUMMARY OF PROBABILITY DISTRIBUTIONS
USED FOR LOAD INPUTS TO BAKER CREEK

Time Frame		Load Entry Point	Estimated Load (kg/y)			
			Mean	Standard Deviation	Minimum	Maximum
1950	1971	Baker Creek	25000	2625	17125	32875
1972	1982	Baker Creek	14000	1970	8090	19910
1983	1993	Baker Creek	1700	180	1160	2240
1994	2007	Baker Creek	800	80	560	1040
1994	2007	Back Bay	110	10	80	140
2008+	Remediation Case	Baker Creek	480	50	330	630
2008+	Remediation Case	Back Bay	210	20	150	270

4.2 PATHWAYS MODELLING

4.2.1 Model Description

Exposure calculations for the assessment were completed using the INTAKE pathways model, which is described in detail in Appendix D. The INTAKE model was developed by SENES for use in simulating environmental transfer, uptake and risk due to radionuclides, stable metals and other inorganic species released to the environment (e.g., air, water, groundwater, soil). The model has an extensive history of development and quality assurance and has been validated through these processes. The model can be run in a deterministic mode or in a probabilistic framework to facilitate uncertainty and sensitivity analyses. For this application, which was designed to assess the relative impacts of a range of arsenic loadings to the aquatic environment, the model was run probabilistically (Monte Carlo). A listing of the numerous input parameters used in the Giant application is presented in Appendix D. Limited calibration of the pathways model for this application is presented in Appendix D.

The INTAKE model was used to assess the effects of arsenic levels in Baker Creek, Back Bay and Yellowknife Bay on exposures for aquatic and terrestrial biota, as well as, humans. Model predictions of arsenic levels in water and sediment in each segment of Yellowknife Bay were used in the estimation of exposure to aquatic species and in the estimation of uptake through the food chain to the ecological and human receptors specified in Section 3.0. In addition, the contributions of arsenic present in air, soil, garden produce, berries and other terrestrial vegetation were incorporated into the pathways assessment of exposure to the ecological and human receptors. The transport steps are relatively straightforward, as detailed in Appendix D. The exposure pathways are explained in Section 4.3.

4.2.2 Site-Specific Distribution Coefficients and Transfer Factors

Pathways modelling involves the use of transfer factors to estimate arsenic levels in the tissue/flesh of all components of the food chain (e.g. in plant tissue and animal flesh). In many instances, site-specific data are not available to permit the estimation of site-specific transfer factors and hence, it is common to use literature-derived values. In this assessment, arsenic levels measured in several environmental components were used to derive site-specific transfer factors. The factors derived from these data are summarized below. The measured concentrations used for these calculations are discussed and presented in Appendix A.

The calculation of site-specific distribution coefficients and transfer factors did not consider values that were reported as less than the detection limit. These results were not included in averages, number of samples or summary statistics.

The derivation and rationale behind the development of site-specific transfer factors for water-to-aquatic vegetation, water-to-fish, water-to-snail and soil-to-vegetation are provided in Appendix A. The water-to-sediment distribution coefficient was derived from calibration of the LAKEVIEW model as described in Appendix B. Table 4.2-1 summarizes the probability distributions used for the pathways modelling for this assessment. Because the water-to-sediment distribution coefficient was derived through model calibration, it was assigned a constant value.

TABLE 4.2-1
DERIVED SITE-SPECIFIC DISTRIBUTION
COEFFICIENTS AND TRANSFER FACTORS

Parameter	Distribution Specifiers	Units
water-to-sediment distribution coefficient	C(0.005)	m^3/g (dw)
water-to-aquatic vegetation transfer factor	LN($6.3x10^{-5}$, 1.8, $1.9x10^{-5}$, $2.0x10^{-4}$)	m^3/g (dw)
water-to-fish transfer factor	LN($5.2x10^{-5}$, 2.8, $6.6x10^{-6}$, $4.1x10^{-4}$)	m^3/g (ww)
water-to-snail transfer factor	LN($3.2x10^{-5}$, 3.5, $2.6x10^{-6}$, $3.9x10^{-4}$)	m^3/g (ww)
soil-to-above-ground vegetation*	β(0.0002, 0.018, 0.77, 4.63)	g (dw) / g (ww)
soil-to-below-ground vegetation*	β(0.0003, 0.003, 2.84, 6.04)	g (dw) / g (ww)

Note: Lognormal Distribution - LN (geometric mean, geometric standard deviation, minimum, maximum).
Beta Distribution - β (minimum, maximum, alpha, beta).
* These transfer factors were only used to derive concentrations in browse and forage.

4.2.3 Arsenic Bioavailability/Bioaccessibility

Bioavailability of a compound can be defined as the fraction of an administered dose that reaches the central (blood) compartment, whether through the gastrointestinal tract, skin or lungs (NEPI 2000). This type of bioavailability is known as "absolute bioavailability". The evaluation of the effect of different routes of exposure on the bioavailability of a compound is known as "relative bioavailability" or bioaccessibility. Bioaccessibility represents the comparative bioavailabilities of different forms of a compound (e.g., metal species) or for different exposure media (e.g., soil vs. water) (NEPI 2000). Consideration of bioaccessibility allows intakes to be adjusted to reflect differences in the bioavailability of a compound in various exposure media (MOE 2001). The use of bioaccessibility in risk assessments is accepted by regulatory agencies in Canada such as Health Canada.

Oral exposures are described in terms of an external dose or intake, as opposed to an absorbed dose or uptake (MOE 2001). Intake occurs as an agent enters the body of a human or animal without passing an absorption barrier (e.g., through ingestion or inhalation), while uptake occurs as an agent passes across the absorption barrier (IPCS 2000). Not all materials (e.g., metals, nutrients) that enter the body as intake are absorbed into the body as uptake. Many are passed through the body and expelled without effect.

When calculating the intake via the oral route of exposure, it is customary to take into account the food, water and soil pathways. The default bioaccessibility value used in screening level (Tier 1) calculations is 100%. In a Tier 2 assessment, adjustments in bioaccessibility of contaminants is allowed to account for the fact that not all of the contaminants present in food, water or soil are available for uptake through the gut.

The bioavailability/bioaccessibility of contaminants (in this case arsenic) is affected by a variety of factors such as pH, the binding of metals to colloids or other particles, and speciation, to name a few.

Sediment samples were collected in Baker Creek and were subjected to sequential extraction as described in Appendix C. The results of the extraction indicated that about 17% of the arsenic in the sediments was bioaccessible. Therefore, in this assessment a bioaccessibility factor of 17% for the arsenic in soil and sediments was used. The assumption of 100% bioaccessibility was still used in the water and food ingestion pathways.

5.0 HAZARD ASSESSMENT

The hazard assessment phase of ecological and/or human health risk assessments involves identification of contaminant concentrations or intakes which have been shown to have adverse effects on the receptors (ecological species or humans) of concern. The exposure concentrations or intakes are generally determined from controlled laboratory tests or from epidemiology studies and are used to establish Toxicity Reference Values (TRVs) which are protective of the receptors.

5.1 TOXICITY REFERENCE VALUES FOR ECOLOGICAL SPECIES

The objective of an ecological risk assessment is to evaluate the potential for adverse effects on a population basis.

5.1.1 Aquatic Toxicity Reference Values

In this assessment, EC_{25} (effects concentration) values which have the potential to affect 25% of the population were used to determine whether arsenic is likely to cause adverse effects in aquatic receptors in Baker Creek or Yellowknife Bay. The EC_{25} value is suggested by Environment Canada for use in Tier 2 assessments (Environment Canada 1997). An EC_{25} concentration was chosen as a TRV because effects or changes in populations in this range are generally not distinguishable from natural variation. In addition, a 25% reduction in rapidly growing populations (e.g. phytoplankton or zooplankton) might be quickly offset by reproduction once the chemical stress is removed or may be offset by growth and immigration from non-affected nearby locations. However, it is recognized that Environment Canada is in the process of reviewing the appropriate aquatic toxicity benchmarks; therefore, this assessment examined the effect of using a lower benchmark of EC_{10} in a sensitivity analysis.

It was not the intent of this assessment to extensively search the primary literature to obtain TRVs, rather this assessment relied on TRVs that have been collated and peer reviewed by various agencies for use in risk assessments. To this end, the rationale for selecting the aquatic TRVs was as follows: the U.S. DOE database (Suter *et al.* 1996) on aquatic TRVs was considered to be a primary source of toxicity information. This database contains TRVs for the protection of aquatic life from contaminants in water. EC_{20} values provided in this database were selected as appropriate TRVs. These TRVs were developed for use in risk assessments, have been peer reviewed and are routinely used in risk assessments. This database provides documentation on the sources and derivations of the values and discusses the relative conservatisms in the TRVs. The selection of the species generally relied on species that were present at the site, but also encompassed species found in the general area. If data were not available from the DOE database then the U.S. EPA database AQUIRE was examined for infilling purposes. The data summarized in this database are from a variety of sources, including peer reviewed literature. Toxicity information provided in the CCME Water Quality Guidelines was also used in the development of TRVs for this assessment. It is acknowledged that these

databases may not have the most recent data; however, the TRVs selected are, in our opinion, appropriate.

Decision rules for the selection of test species were developed around the available data. For aquatic plants, the lowest of the toxicity values for *Lemna* sp. or *Myriophllum* sp. test species was chosen. These two species are considered to be the most sensitive aquatic plant species for which toxicity data are available. For benthic invertebrates, the lowest available toxicity values for any invertebrate test species were used. For the fish species, data were chosen for the species based on feeding habits (i.e., predatory habits or bottom feeders). For example, for the bottom feeding fish, the available toxicity data for fathead minnow, white sucker, goldfish, tilapia, stickleback and snakehead catfish, were considered. For the predatory fish, toxicity data for brook trout, rainbow trout and walleye were considered. The lowest toxicity value of these species was chosen to represent the respective predatory or bottom feeder fish.

In summary, arsenic TRVs provided in Table 5.1-1 were obtained from the U.S.EPA AQUIRE database, as well as the CCME. When toxicity studies provided data in the form of a lethal concentration (LC_{50}), a factor of 4 was used to convert between an LC_{50} and an EC_{25} based on an empirical relationship based on comparison of reported data for LC_{50} and EC_{25} for a number of aquatic species and contaminants. As mentioned above, a sensitivity analysis was carried out using an EC_{10} TRV. These were obtained by linear interpolation of the EC_{20} values.

TABLE 5.1-1
SUMMARY OF EC_{25} TOXICITY REFERENCE VALUES FOR AQUATIC SPECIES

Aquatic Receptor	Arsenic (mg/L)				
	Test Species	LC/EC_{50}	Toxicity Reference Value	Reference	Comments
Aquatic Plants	*Lemna minor*	0.63	0.32	Jenner & Janssen-Mommen (1993)	14-d EC_{50} (pop); used an EC_{25} from linear interpolation
Benthic Invertebrates	*Calanus sp.*		0.34	Borgmann *et al.* (1980)	from CCME (1999); 14-d EC_{20}; used as an EC_{25}
Predator Fish	Rainbow Trout	0.55	0.14	Birge *et al.* (1979b)	from CCME (1999); 28-d LC_{50}; derived using a factor of 4 based on an empirical relationship between a chronic LC_{50} and an EC_{25}.
Bottom Feeder Fish	Goldfish	0.49	0.12	Birge *et al.* (1979a)	from U.S. EPA AQUIRE; Only test species for which data exist - 7-d LC_{50} (mor); derived using a factor of 4 based on an empirical relationship between a chronic LC_{50} and an EC_{25}.

5.1.2 Sediment Benchmarks

The potential ecological effects of sediment contamination were addressed in part through the examination of potential effects on benthic invertebrates. The establishment of sediment quality guidelines is a relatively new area of environmental science and several approaches have been attempted to evaluate different measurements of sediment quality and toxicity and convert those measurements into regulatory guidelines/standards.

Table 5.1-2 outlines selected sediment benchmarks for arsenic that are available in the literature. As seen in the table, there is a range of data for possible effects. Guidelines developed by the Canadian Council of Ministers of the Environment (CCME) provide what are designated Threshold Effect Levels (TELs) and the Probable Effect Levels (PELs). In narrative description, a TEL represents the concentration below which adverse biological effects are expected to occur rarely (i.e., fewer than 25% of adverse effects occur below the TEL). A PEL defines the level above which adverse effects are expected to occur frequently (i.e., more than 50% of adverse effects occur above the PEL, or above which adverse effects are usually or always observed).

The Canadian sediment quality guidelines were developed with the intention of being conservative (CCME 1999). A TEL is calculated as the geometric mean of the lower 15th percentile of the effect data set and the 50th percentile of the no-effect data set, with an application factor applied if the uncertainty is high. A PEL is the geometric mean of the 50th percentile in the effects data set and the 85th percentile in the no effects data set. Therefore, a TEL represents the upper limit of the range of sediment contaminant concentrations dominated by no effects data. A PEL represents the lower limit of the range of contaminant concentrations that are usually or always associated with adverse biological effects (MacDonald *et al.* 1994). The CCME acknowledge the associative basis of the guidelines and maintain that the use of SQGs (TELs) in exclusion of other information (such as background concentrations of naturally occurring substances and biological tests) can lead to erroneous conclusions.

The arsenic benchmarks from Thompson *et al*. (2002) are specific to northern Saskatchewan and northern Ontario, but given the similar climatic conditions may be appropriate to consider for the Yellowknife area. Liber and Sobey (2000) also provide sediment benchmark levels for arsenic, which have been derived for Saskatchewan.

Given that each methodology discussed above has limitations, it is felt that the use of the various benchmarks in the assessment was appropriate.

TABLE 5.1-2
SEDIMENT QUALITY BENCHMARK VALUES

	CCME 1999		**Thompson *et al.* 2002**		**Liber and Sobey[a] 2000**			
	TEL	**PEL**	**LEL**	**SEL**	**NOEC**	**LOEC**	**Growth Effects**	
							IC_{25}	**IC_{50}**
Metal (µg/g)								
Arsenic	5.9	17	10	346	-	39	174	342

Notes:
TEL threshold effects level.
PEL probable effect level.
LEL lowest effect level.
SEL severe effect level.
NOEC no observed effect concentration.
LOEC lowest observed effect concentration.
IC_{25} Inhibitory concentration that effects growth in 25% of the population.
IC_{50} Inhibitory concentration that affects growth in 50% of the population.
Dash (-) means insufficient data available to establish benchmark.
[a] The data from Liber and Sobey (2000) for arsenic are based on *Chironomus tetans* in spiked-sediment toxicity tests. The authors found that *Chironomus tetans* was more sensitive to arsenic exposure than *Hyalella azteca*.

It is important to note that these benchmarks should only be used for screening purposes. An exceedance of any of these benchmarks does not mean that an adverse effect would be observed; rather, it means that other factors need to be considered to determine whether there are effects in sediment dwelling populations.

5.1.3 Terrestrial Toxicity Reference Values

For slower reproducing, less dense populations (such as the larger mammals), a 25% decrease in population may not be acceptable. In this assessment where realistic assumptions of exposure were being made, a LOAEL (Lowest Observable Adverse Effects Level) was chosen as the appropriate TRV. In general, NOAELs (No Observable Adverse Effects Levels) are used for screening level type assessments, whereas LOAELs are used in assessments where specific assumptions have been made to obtain more realistic estimates of COPC exposures (Sample *et al*. 1996). These TRVs are generally based on data from laboratory animals.

The studies used for mammals exposed to arsenic are presented in Table 5.1-3 along with the basis for the derivation of NOAELs and LOAELs. For terrestrial birds, a similar analysis is also presented in Table 5.1-3. As seen in the table, the TRVs for arsenic exposure to small mammals (hare, mink, muskrat) are derived from the U.S. DOE database by Sample *et al*. (1996), which provides information on laboratory tests in mice exposed to arsenite. For large mammals, such as moose and caribou, TRVs were derived from cattle. Since the digestive processes are similar for cattle and moose and caribou, the use of cattle as a representative species was deemed to be more appropriate than using mice data. In addition, the chemical species of arsenic for which toxic doses were reported was arsenic trioxide. For bear and wolf, toxicity data pertaining to beagle dogs exposed to arsenite were used (ATSDR 2000), as dogs were deemed to be a more appropriate surrogate than mice.

The toxicity data for mammals were scaled by body weight for the various wildlife species using equation (5.1-2) presented in Sample *et al*. (1996).

$$LOAEL_{wildlife} = LOAEL_{testspecies}\left(\frac{BodyWeight_{testspecies}}{BodyWeight_{wildlife}}\right)^{0.25} \qquad (5.1\text{-}2)$$

TABLE 5.1-3
SUMMARY OF TOXICOLOGICAL REFERENCE VALUES FOR ARSENIC BASED ON LABORATORY ANIMAL AND BIRD STUDIES

	Small Mammals (muskrat, mink, hare)	**Large Mammals (wolf, bear)**	**Large Mammals (moose, caribou)**	**Birds**
Source of Reference Values	Sample *et al*. (1996)	ATSDR (2000)	Puls (1994)	Sample *et al*. (1996)
Original Reference	Schroeder & Mitchener (1971)	Byron *et al.* (1967)	-	USFWS 1964
Form of chemical	Arsenite (As^{3+})	Arsenite (As^{3+})	Arsenic Trioxide	Arsenite (As^{3+})
Test Species	Mouse	Beagle Dog	Cattle	Mallard duck
Body Weight (g)	30	11,800	500,000	1000
Study Duration	3 generations (>1yr)	2 years	Not given	128 d (>10wks)
Endpoint	Reproduction	Blood, gastrointestinal and liver effects	Colic, partial paralysis, salivation, watery or bloody diarrhea, dehydration, depression, loss of appetite, seizures, hematuria and hypothermia or fever	Mortality
Comments	The study was carried out during a critical life stage and is considered to be chronic exposure	Study is considered a long term or chronic study.	-	The study was carried out over a period greater than 10 weeks and is considered to be chronic exposure
NOAEL (mg/kg d)	**-**	**1.0**	**-**	**5.15**
LOAEL (mg/kg d)	**1.26**	**2.4**	**1.1**	**12.84**
Logic	Mice displayed declining litter sizes with each successive generation at a dose of 1.26 mg/kg/d and thus this dose was considered to be a chronic LOAEL	Mild anemia was found in dogs fed arsenite for 2 years at 2.4 mg As/kg/d but no haematological effects were observed at 1 mg As/kg/d. Dogs fed a diet of 2.4 mg/kg/d arsenite for 2 years were reported to have some bleeding in the gut. Dogs did not show any clinically significant hepatic injury and thus this dose was considered to be a chronic LOAEL	Arsenic may act as an essential trace element in cattle but no deficiencies have been noted. Cattle will develop a tolerance to arsenic if fed over a period of time. Can subsequently develop a dependence on these high levels. Excess arsenic is rapidly excreted. The toxic dose in cattle is reported to be 33-45 mg/kg/d. A factor of 10 was used to convert the lowest concentration in the range to a LOAEL and a factor of 3 was used for the uncertainty in the database.	Over 128 days at a dose of 100 mg/kg sodium arsenite (51.35 mg/kg As^{+3}) the ducks experienced no mortality and thus this value is considered to be a NOAEL. A dose of 250 mg/kg (128.375 mg/kg As^{+3}) resulted in 12% mortality and is considered to be a chronic LOAEL

The above data was applied to the assessment as follows: The LOAEL data from the mouse study was scaled for body weight for hare, mink and muskrat as per equation (5.1-2). Dog data were scaled for bear and wolf and the cattle toxicity value was scaled for caribou and moose. For birds, the NOAEL and LOAEL values provided in Table 5.1-3 is used for all bird species regardless of body weight. Table 5.1-4 provides a summary of the TRVs for terrestrial species used in this assessment.

TABLE 5.1-4
SUMMARY OF ARSENIC TOXICOLOGICAL REFERENCE VALUES FOR TERRESTRIAL SPECIES

Contaminant		Species	LOAEL (mg/(kg d))
Arsenic	Birds	Duck	12.84
		Grouse	12.84
	Mammals	Bear	1.15
		Caribou	1.62
		Hare	0.48
		Mink	0.52
		Moose	1.05
		Muskrat	0.5
		Wolf	1.74

5.2 TOXICITY REFERENCE VALUES FOR HUMANS

Toxicological Reference Values derived by the U.S. EPA and Health Canada were considered for use in this assessment. Health Canada has indicated that the TRV's for federally funded projects selected should be obtained from their organization. The U.S. EPA has classified arsenic as a Group A carcinogen (human carcinogen). This means that any intake is assumed to result in a risk of cancer formation. Arsenic is also known to have non-carcinogenic effects; however, for this assessment, we have evaluated the most sensitive endpoint which is cancer. The U.S. EPA has developed a carcinogenic slope factor for oral exposure based on the potential to develop skin cancer. Health Canada provides a recent re-evaluation of the available data that focussed on internal cancers (Health Canada 2004). This is discussed in more detail below.

Carcinogenic Effects

Arsenic exposure via the oral route is considered to be carcinogenic based on the incidence of skin cancers in epidemiological studies examining human exposure through drinking water (Tseng *et al.* 1968; Tseng 1977). A slope factor of 1.5 $(mg/(kg\ d))^{-1}$ was derived from these

studies by the U.S. EPA. For FCSAP assessments, Health Canada recommends a slope factor of 2.8 $(mg/(kg\ d))^{-1}$. This slope factor is higher than the U.S. EPA slope factor that was used in previous risk assessments for the Giant Mine site.

In the recent proposed Health Canada drinking water guideline for arsenic, new data have become available that suggest that the risk of internal cancers due to ingestion of drinking water is greater than previously believed (Health Canada 2004). An evaluation was completed by Health Canada of the cancer potency indices for liver, lung, bladder and kidney cancers. The lifetime risks of cancer in the general population in Canada associated with ingestion of 1 µg/L of arsenic in drinking water were estimated to be 4.5×10^{-6} (based on kidney cancer in men) to 2.2×10^{-5} (based on lung cancer in women) (Health Canada 2004). Based on the 95% upper bound level, an oral slope factor of 1.2 $(mg/(kg\ d))^{-1}$ can be determined.

In the most recent analysis of the data presented by Health Canada (2004), the cancer risk models based on Taiwanese data have been updated (Morales *et al.* (2000)). These studies analyze data obtained from Taiwanese populations chronically exposed to high levels of arsenic in their drinking water. Several documents and authors point to the difficulty in using the Taiwanese data to regulate exposure in North American populations because of different general health conditions, ethnic mix and other risk factors. Taking these uncertainties into consideration, it is questionable whether the risks estimated for overseas populations are directly applicable in the North American context. The ATSDR (2000) document discusses the limits of these studies, and in particular, the limits of the Taiwanese data used in the current cancer risk estimates for arsenic from drinking water. The main argument stems from the fact that cancers were observed at exposure levels of 0.022 mg/(kg d) to 0.064 mg/(kg d); however, it is difficult to extrapolate these effects down to lower levels of exposure since the dose response curve is non-linear. Other issues are as follows (ATSDR 2000, page 121):

- Host and environmental factors among the Taiwanese not applicable elsewhere;
- A possible threshold for arsenic carcinogenicity;
- Differences in health and nutrition between Taiwan and the U.S. that might increase cancer in Taiwan;
- The possibility that arsenic is an essential nutrient at lower doses; and,
- The possibility of significant exposure to arsenic from sources other than well water.

As well as the concerns raised by the above, there are epidemiologic issues: the use of ecologic measures of exposure - rather than individual measures; and genetic issues - these populations are quite distinct and may have differences in the metabolism of arsenic (Cantor 2001) that may result in uncertainty in the risk estimates.

Despite the above concerns, Health Canada used the Taiwanese data to develop a slope factor of 1.2 $(mg/(kg\ d))^{-1}$ based on kidney cancers in men and lung cancer in women, which was applied in the development of the proposed drinking water guideline (Health Canada 2004). The use of this slope factor results in an acceptable concentration of 0.4 µg/L arsenic in drinking water at a risk level of 1 in 100,000 (which is deemed by Health Canada as an "essentially negligible" level of risk). However, the level of practical treatment of arsenic in source water supplies is 5 µg/L. At this concentration, the risk level is 1.2 in 10,000, (i.e. 1.2×10^{-4}).

In summary, the TRV used in this assessment is 1.2 $(mg/(kg\ d))^{-1}$) for estimating the increased risk of internal cancer. In taking into account the higher slope factor provided by Health Canada for use in FCSAP assessments, a sensitivity analysis was carried out using a slope factor of 2.8 $(mg/(kg\ d))^{-1}$.

6.0 RISK CHARACTERIZATION

The risk characterization phase combines the information gathered in the exposure and hazard assessment phases and characterizes the potential for adverse ecological and human health effects. This section of the report: presents the results of surface water and sediment quality predictions and compares them to published quality guidelines; evaluates the risks to aquatic species via comparison of exposure levels to toxicity reference values; evaluates the risks to terrestrial species via comparison of estimated intakes to toxicity reference values; and, evaluates the risks of carcinogenic effects to humans.

6.1 AQUATIC ENVIRONMENT

The prediction of water and sediment quality impacts of the post-remediation arsenic loadings described in Section 2.2.1 was carried out using the lake dispersion model described in Appendix B. The model was run probabilistically to account for uncertainty and/or natural variability in many of the model input parameters. Water and sediment quality simulations for both cases were carried out over a 150-year time period spanning from 1950 through 2100. The simulations were run in one-year time steps commencing in 1950 to account for the effects of past operations on environmental levels of arsenic. For assessment purposes, it was assumed that the load associated with the remediation scenario would commence in 2008. While the selection of this start date was arbitrary, it should be noted that the resulting impacts on Baker Creek, Back Bay and Yellowknife Bay would be very similar regardless of the start date; just the time frame would shift.

Each simulation comprised 1000 probabilistic (Monte Carlo) trials, which has been found in other similar work to be sufficient to define the shapes of the model output curves and for the characterization of the statistical attributes of the model outputs. The model results were processed in this assessment to determine the mean, 5^{th} percentile and 95^{th} percentile values of water and sediment quality levels at several points in time over the simulation interval. These results are presented in the following discussion.

6.1.1 Water Quality Predictions

Baker Creek System

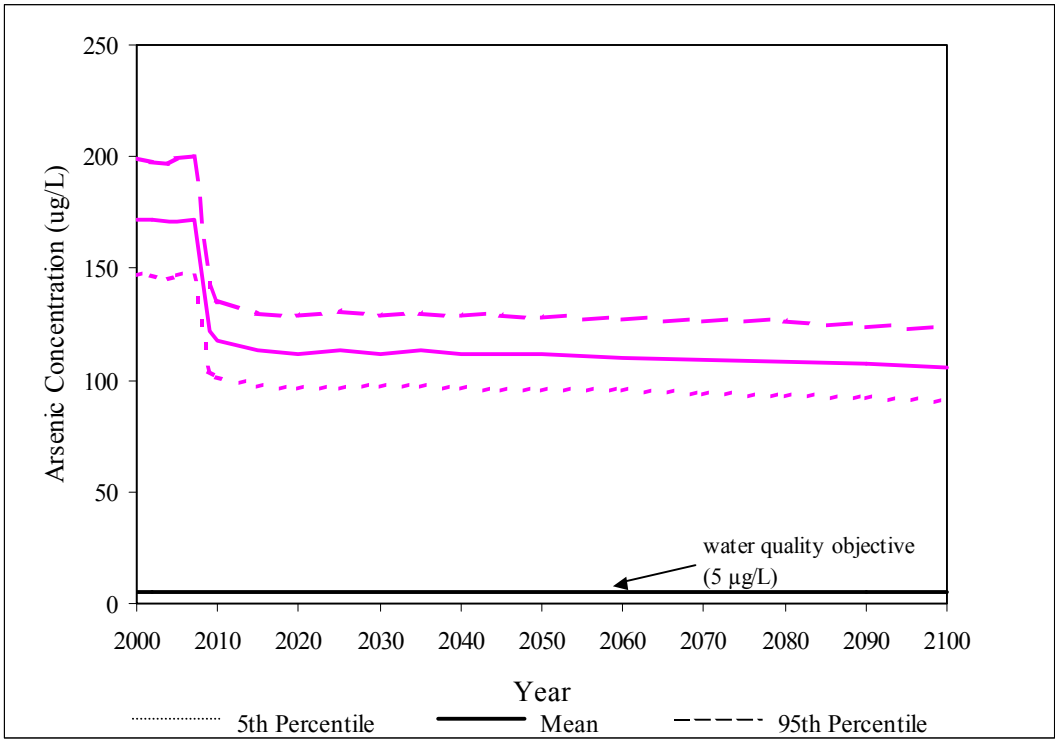
The predicted mean, 5^{th} and 95^{th} percentile arsenic levels in Baker Creek over the 2000 through 2100 period are shown in Figure 6.1-1. As shown, the arsenic level is seen to decrease sharply and significantly between 2008 and 2010 then continue to decline very gradually over the remainder of the simulation period. This sharp decline is a result of the implementation of the remediation measures in 2008.

The predicted mean, 5th and 95th percentile arsenic concentrations for Baker Creek in 2010 are presented in Table 6.1-1. As seen in both Figure 6.1-1 and Table 6.1-1, the predicted arsenic concentrations exceed both the Canadian environmental quality guidelines for the protection of aquatic life (5 µg/L) (CCME 2002) and the new proposed guideline for drinking water (5 µg/L) (Federal-Provincial-Territorial Committee on Drinking Water, 2004). Section 6.2 presents the results of the ecological risk assessment performed on Baker Creek.

TABLE 6.1-1
PREDICTED ARSENIC CONCENTRATIONS IN WATER IN THE FOUR STUDY SEGMENTS IN YEAR 2010 FOR THE REMEDIATION CASE

Study Segment	Arsenic Water Concentrations (µg/L)		
	5th Percentile	Mean	95th Percentile
Baker Creek	101	118	136
Back Bay	2.8	3.0	3.3
North Yellowknife Bay	1.3	1.4	1.6
South Yellowknife Bay	0.54	0.59	0.65
CCME (2002) Guideline	5		
Health Canada (2004) Proposed Drinking Water Guideline	5		

FIGURE 6.1-1
PREDICTED ARSENIC LEVELS – BAKER CREEK WATER

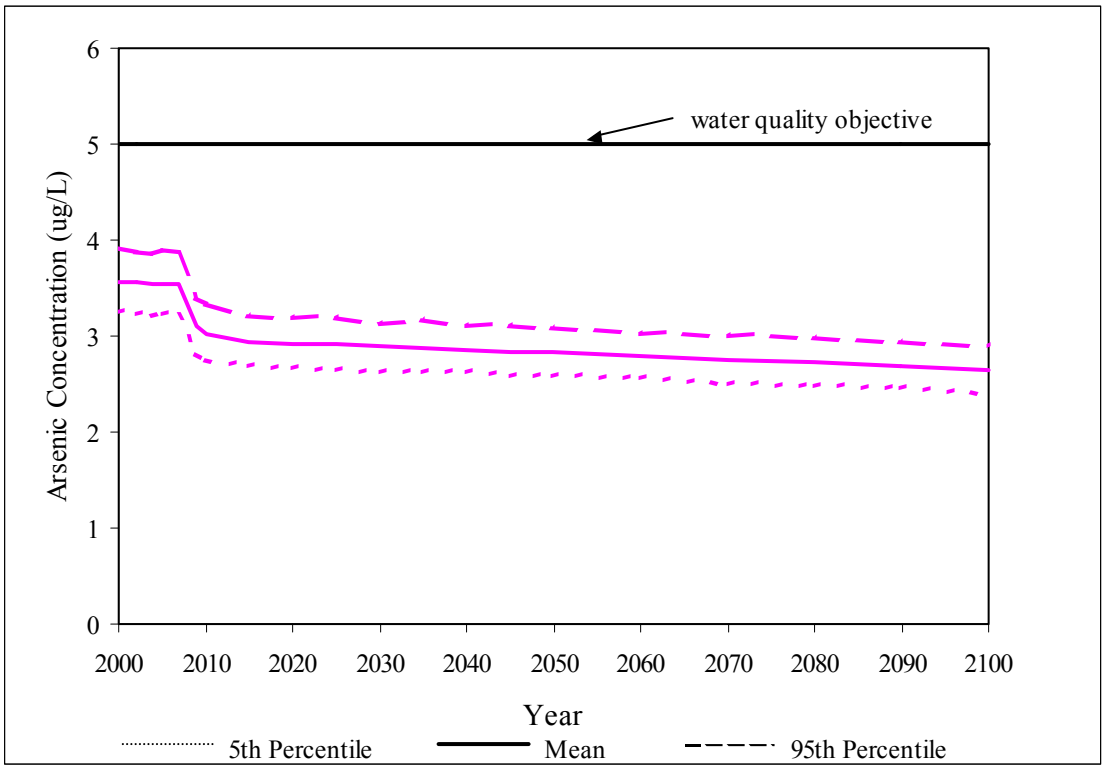


Back Bay

The predicted arsenic levels in Back Bay for the remediation case are provided in Figure 6.1-2. As shown in this figure, a sharp decrease in the arsenic level between 2008 and 2010 is visible, reflecting the decrease in the loading from Baker Creek, which was assumed to occur in 2008. Over the long term, the predicted arsenic concentrations are predicted to gradually decline as the system slowly recovers from the historic accumulation of arsenic in the lake sediments.

The predicted mean, 5th and 95th percentile values for Back Bay in 2010 are presented in Table 6.1-1, and Figure 6.1-2. The arsenic concentration in the water column of Back Bay is predicted to remain below the proposed drinking water quality guideline of 5 µg/L and the surface water quality guideline of 5 µg/L for the protection of aquatic life.

FIGURE 6.1-2
PREDICTED ARSENIC LEVELS – BACK BAY WATER

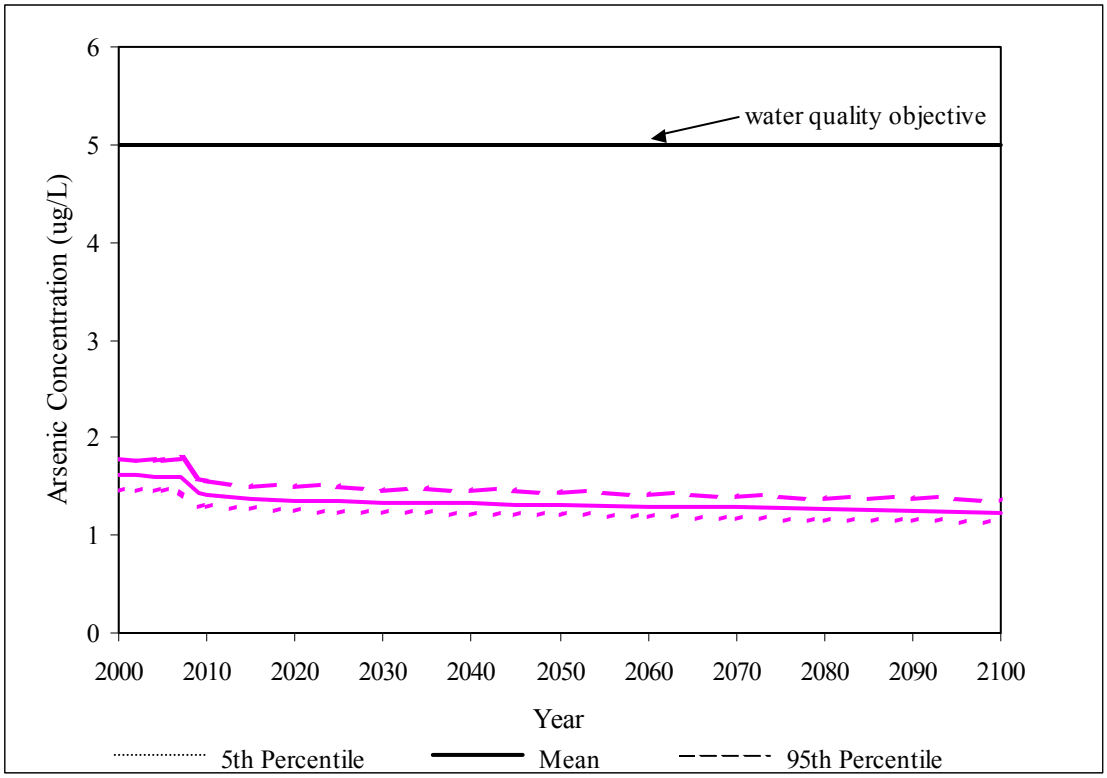


North Yellowknife Bay

The predicted arsenic levels in North Yellowknife Bay water are plotted in Figure 6.1-3. A small, rapid drop followed by a gentle decline in arsenic levels in North Yellowknife Bay can be seen. The predicted mean, 5th and 95th percentile values for North Yellowknife Bay in 2010 are presented in Table 6.1-1.

The predicted arsenic concentrations in the water column of North Yellowknife Bay are well below both the proposed drinking water quality guideline and the current surface water quality guideline of 5 µg/L.

FIGURE 6.1-3
PREDICTED ARSENIC LEVELS – NORTH YELLOWKNIFE BAY WATER

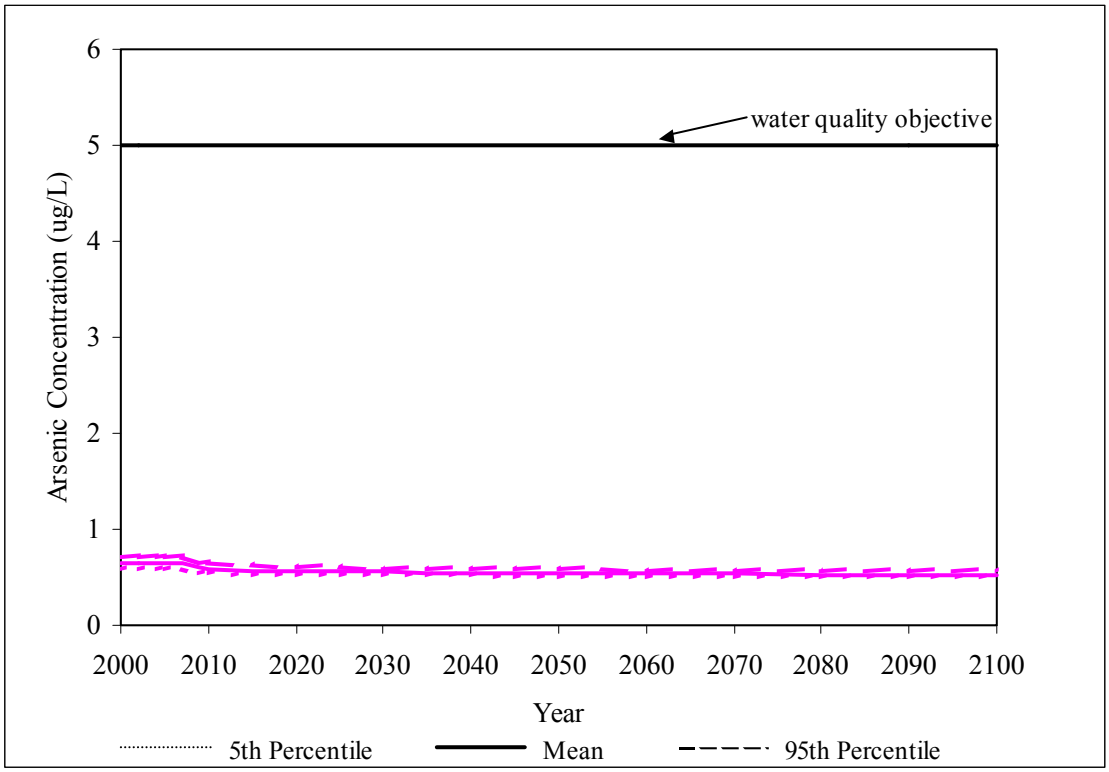


South Yellowknife Bay

The water quality predictions for arsenic in South Yellowknife Bay surface water are provided in Figure 6.1-4. As seen from Figure 6.1-4, there is a very slight initial decrease in arsenic concentrations predicted in South Yellowknife Bay.

The predicted mean, 5th and 95th percentile values for South Yellowknife Bay in 2010 are presented in Table 6.1-1. The arsenic concentrations in the water column in South Yellowknife Bay are well below the proposed drinking water guideline and the current surface water quality guideline. The predicted levels are approximately twice the baseline level of 0.3 µg/L measured in the Yellowknife River.

FIGURE 6.1-4
PREDICTED ARSENIC LEVELS – SOUTH YELLOWKNIFE BAY WATER



6.1.2 SEDIMENT QUALITY PREDICTIONS

Baker Creek System

Sediment samples collected on Baker Creek during the early 1990's had an average arsenic content of approximately 2,340 µg/g (dry weight basis). Figure 6.1-5 shows the predicted arsenic concentrations in Baker Creek sediment for the remediation scenario. Over time, a decreasing trend is evident. In this scenario, it was assumed that there was no direct remediation of sediments in the lower reach of Baker Creek. Rather, the predicted improvements in sediment quality is due to the fact that the arsenic loads in surface water runoff are below historical levels. The arsenic concentrations in the creek sediments are expected to decrease as a result of arsenic diffusion from the sediment to the water phase.

Table 6.1-2 compares predicted mean sediment concentrations in 2010 and 2100 to toxicity reference values published by several different authors (discussed in Section 5.0) for the remediation case. The screening index values presented in the table are the calculated ratios of the predicted sediment concentrations to the respective toxicity reference values (TRVs), which are shown at the top of each column. A screening index value greater than 1 indicates that the predicted concentration exceeds the TRV. An exceedance of the TRV does not necessarily mean that a negative impact will occur, rather that field and/or laboratory investigations (benthic community surveys, toxicity studies, etc.) are necessary to determine if predicted or measured concentrations are likely to have an adverse effect.

In this regard, field surveys were undertaken on Baker Creek in the fall of 2001 and again in the summer of 2002 (Dillon 2002a, 2002b). Sampling in the fall of 2001 found a low abundance of benthic invertebrates downstream of the mine site although Dillon (2002a) suggested that this was due to the timing of the sampling in early winter. In the second sampling program completed in the summer of 2002 (Dillon 2002b), Hester Dendy Plates (multi-plate artificial substitute samplers) were used to obtain sufficient benthic invertebrate biomass for tissue metal analyses. Benthic samples collected from downstream locations were found to generally contain higher arsenic concentrations than benthos from upstream of the Giant Mine site. In both surveys, metal sensitive benthic species were found to be largely absent from the downstream sites. The predominant taxa at the downstream sampling locations were those that are generally considered indicators of poor water quality.

TABLE 6.1-2
COMPARISON OF PREDICTED (MEAN) ARSENIC CONCENTRATIONS IN BAKER CREEK SEDIMENT TO TOXICITY REFERENCE VALUES

	Sediment Concentration (µg/g)	CCME 1999		Thompson *et al.* 2002		Liber and Sobey 2000			
		TEL	PEL	LEL	SEL	NOEC	LOEC	Growth Effects	
								IC_{25}	IC_{50}
Toxicity Reference Values (µg/g)	-	5.9	17	10	346	-	39	174	342
		Screening Index Values							
Remediation Case – 2010	2210	**375**	**130**	**221**	**6.4**	-	**57**	**13**	**6.5**
Remediation Case - 2100	1660	**281**	**98**	**166**	**4.8**	-	**43**	**10**	**4.9**

Notes:

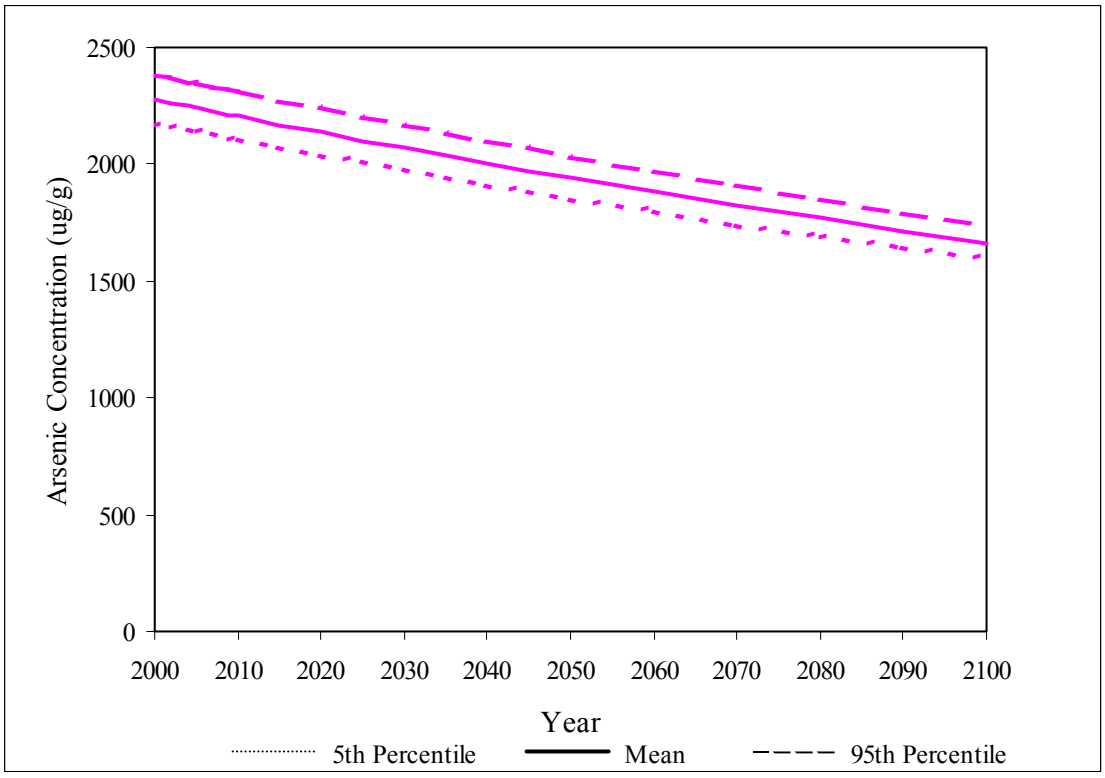
TEL – Threshold Effect Level LEL - Lowest Effect Level NOEC - No-observed-effect-concentration

PEL - Probable Effect Level SEL - Severe Effect Level LOEC – Lowest-observed-effect-concentration

Screening Index (SI) values equal the ratio of predicted arsenic concentrations to toxicity reference values.

Bold SI values indicate that the predicted concentration exceeds the toxicity reference value.

FIGURE 6.1-5
PREDICTED ARSENIC LEVELS – BAKER CREEK SEDIMENT



Back Bay

The predicted arsenic concentrations in Back Bay sediment (Figure 6.1-6) show a decreasing trend. Table 6.1-3 presents the screening index values calculated for the remediation case in years 2010 and 2100, based on the mean predicted concentrations divided by the toxicity reference values shown at the top of each column.

The range of screening index values in Table 6.1-3 indicates the uncertainty present in the establishment of toxicity reference values for sediment. The results suggest that there is a potential for adverse effects on the benthic community in Back Bay based on the predicted arsenic concentrations in sediment.

A benthic invertebrate investigation performed in 1975 by Moore *et al*. (1978) showed that benthic diversity increased with distance from the outlet of Baker Creek and that recovery of the benthic population was evident at distances of greater than 1,000 m from Baker Creek. In 2004, Golder Associates carried out sampling of benthic invertebrates in Back Bay in the vicinity of the tailings depositional area along the north shore of the bay. Preliminary results provided by the investigators via personal communication indicated that as the arsenic concentration

increases, the benthic invertebrate abundance decreases. In addition, a change in overall benthic invertebrate community structure was detected, which the investigators attributed largely to a lower abundance of most invertebrate groups at locations with elevated arsenic levels in the sediment solids. Also, it was found that the benthic invertebrate richness (based on the lowest level of taxonomic identification) and the abundance of metal-sensitive benthic invertebrate groups was lower at locations with higher arsenic levels.

The findings of the benthic community surveys support the results of the risk assessment. In the short-term, therefore, it is expected that the diversity of the benthic community will be impaired. In the long-term (>100 years) however, as sediment quality improves, it is expected that a diverse benthic community will re-establish itself in most parts of Back Bay. Benthic community surveys and other toxicity tests will be necessary to verify this conclusion.

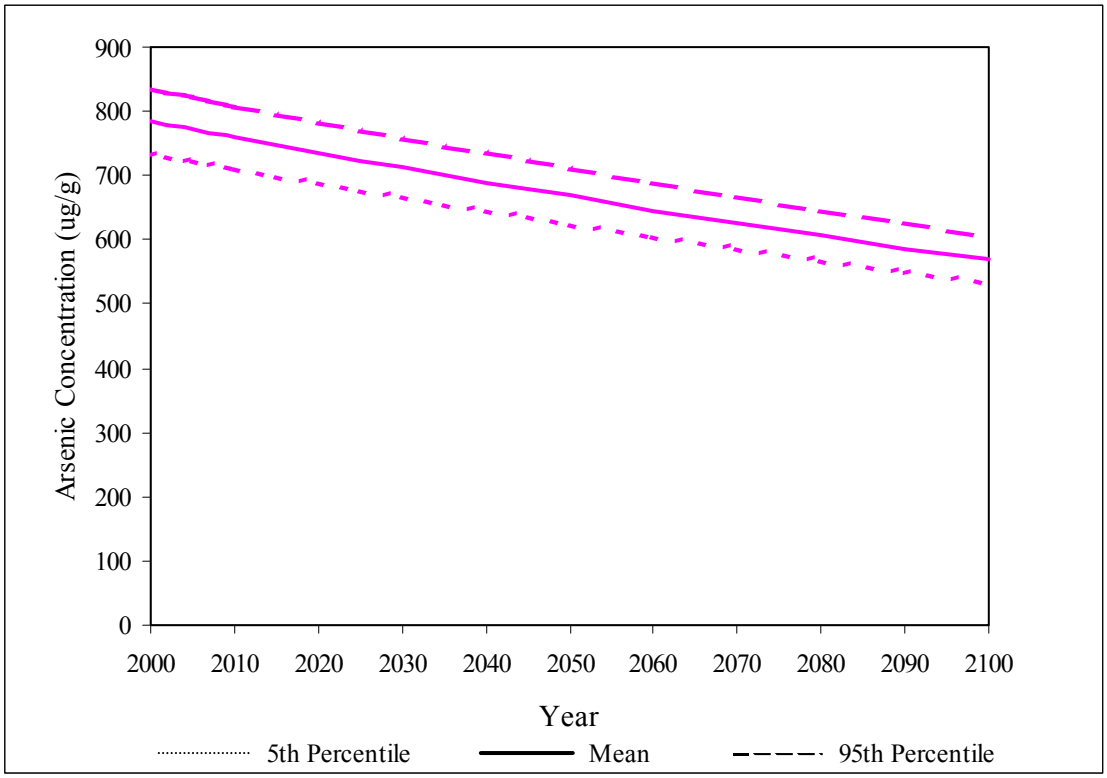
It should be noted that the background sediment arsenic concentration in the Yellowknife area is in the order of 15 µg/g, based on sediment core samples taken from South Yellowknife Bay (Murdoch *et al.* 1989). This background concentration exceeds a number of sediment toxicity guidelines used in the assessment, including the CCME (1999) TEL and the Thompson *et al.* (2002) LEL benchmark values.

TABLE 6.1-3
COMPARISON OF PREDICTED (MEAN) ARSENIC CONCENTRATIONS IN BACK BAY SEDIMENT TO TOXICITY REFERENCE VALUES

	Sediment Concentration (µg/g)	CCME 1999		Thompson *et al.* 2002		Liber and Sobey 2000			
		TEL	PEL	LEL	SEL	NOEC	LOEC	Growth Effects	
								IC_{25}	IC_{50}
Toxicity Reference Values (µg/g)	-	5.9	17	10	346	-	39	174	342
		Screening Index Values							
Remediation Case – 2010	759	**129**	**45**	**76**	**2.2**	**-**	**19**	**4.4**	**2.2**
Remediation Case - 2100	569	**96**	**33**	**57**	**1.6**	**-**	**15**	**3.3**	**1.7**

Notes:
TEL Threshold Effect Level LEL Lowest Effect Level NOEC No-observed-effect-concentration
PEL Probable Effect Level SEL Severe Effect Level LOEC Lowest-observed-effect-concentration
Screening Index (SI) values equal the ratio of predicted arsenic concentrations to toxicity reference values.
Bold SI values indicate that the predicted concentration exceeds the toxicity reference value.

FIGURE 6.1-6
PREDICTED ARSENIC LEVELS – BACK BAY SEDIMENT



North Yellowknife Bay

The predicted arsenic concentrations in North Yellowknife Bay sediment (Figure 6.1-7) show a decreasing trend from the current level for the remediation scenario. Table 6.1-4 provides a comparison of predicted sediment concentrations in North Yellowknife Bay for the remediation case for the years 2010 and 2100 to a range of toxicity reference values (discussed in Section 5.0).

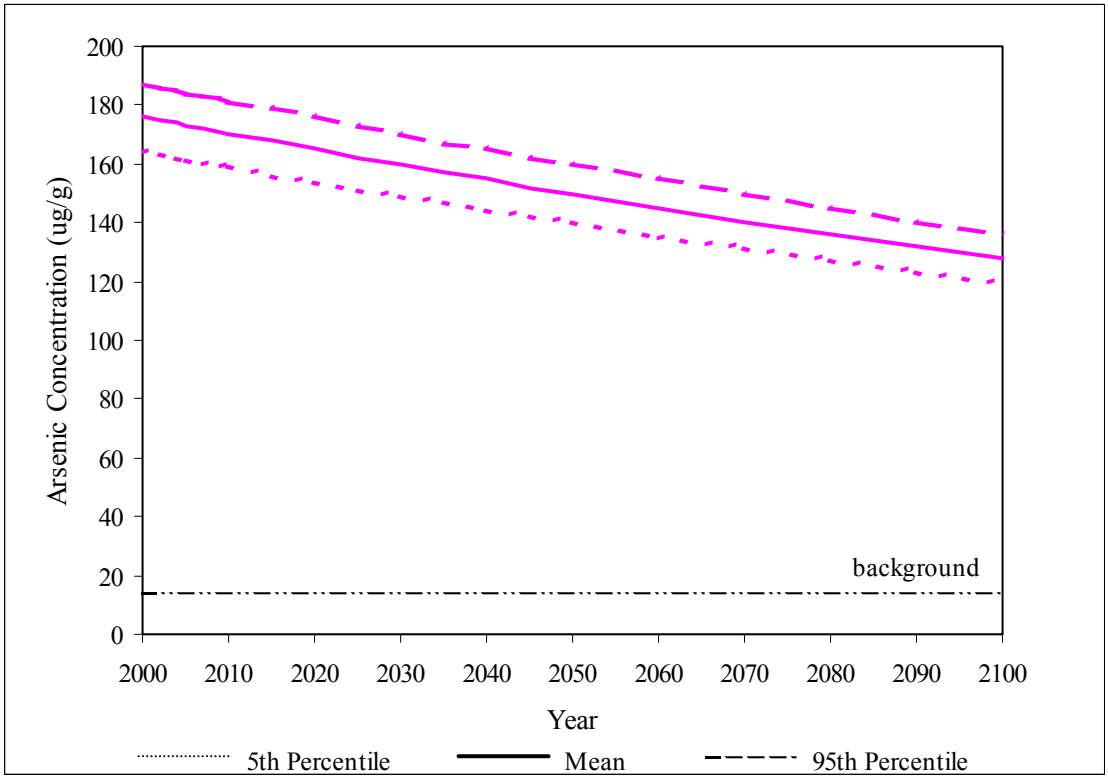
Table 6.1-4 shows that a number of the sediment toxicity reference values are exceeded by the predicted sediment concentrations in North Yellowknife Bay (i.e. a number of the screening index values are greater than 1). Only the SEL benchmark proposed by Thompson *et al.* (2002) and the IC_{50} reference value proposed by Liber and Sobey (2000) are not exceeded. Whether the predicted future sediment arsenic levels will have an effect on the health of the benthic community in North Yellowknife Bay is uncertain.

TABLE 6.1-4
COMPARISON OF PREDICTED (MEAN) ARSENIC CONCENTRATIONS IN NORTH YELLOWKNIFE BAY SEDIMENT TO TOXICITY REFERENCE VALUES

	Sediment Concentration (µg/g)	CCME 1999		Thompson *et al.* 2002		Liber and Sobey 2000			
		TEL	PEL	LEL	SEL	NOEC	LOEC	Growth Effects	
								IC_{25}	IC_{50}
Toxicity Reference Values (µg/g)	-	5.9	17	10	346	-	39	174	342
		Screening Index Values							
Remediation Case – 2010	170	**29**	**10**	**17**	0.49	-	**4.4**	**1.0**	0.50
Remediation Case - 2100	128	**22**	**7.5**	**13**	0.37	-	**3.3**	0.7	0.37

Notes:
TEL Threshold Effect Level
PEL Probable Effect Level
LEL Lowest Effect Level
SEL Severe Effect Level
NOEC No-observed-effect-concentration
LOEC Lowest-observed-effect-concentration
Screening Index (SI) values equal the ratio of predicted arsenic concentrations to toxicity reference values.
Bold SI values indicate that the predicted concentration exceeds the toxicity reference value.

FIGURE 6.1-7
PREDICTED ARSENIC LEVELS – NORTH YELLOWKNIFE BAY SEDIMENT



South Yellowknife Bay

The predicted arsenic concentrations in the sediment for South Yellowknife Bay all decline to near background in the remediation case over the simulation period. Figure 6.1-8 shows the predicted sediment concentrations over time for South Yellowknife Bay and Table 6.1-5 summarizes the screening index values calculated as the ratio of predicted sediment concentrations to the sediment toxicity reference values for the remediation case.

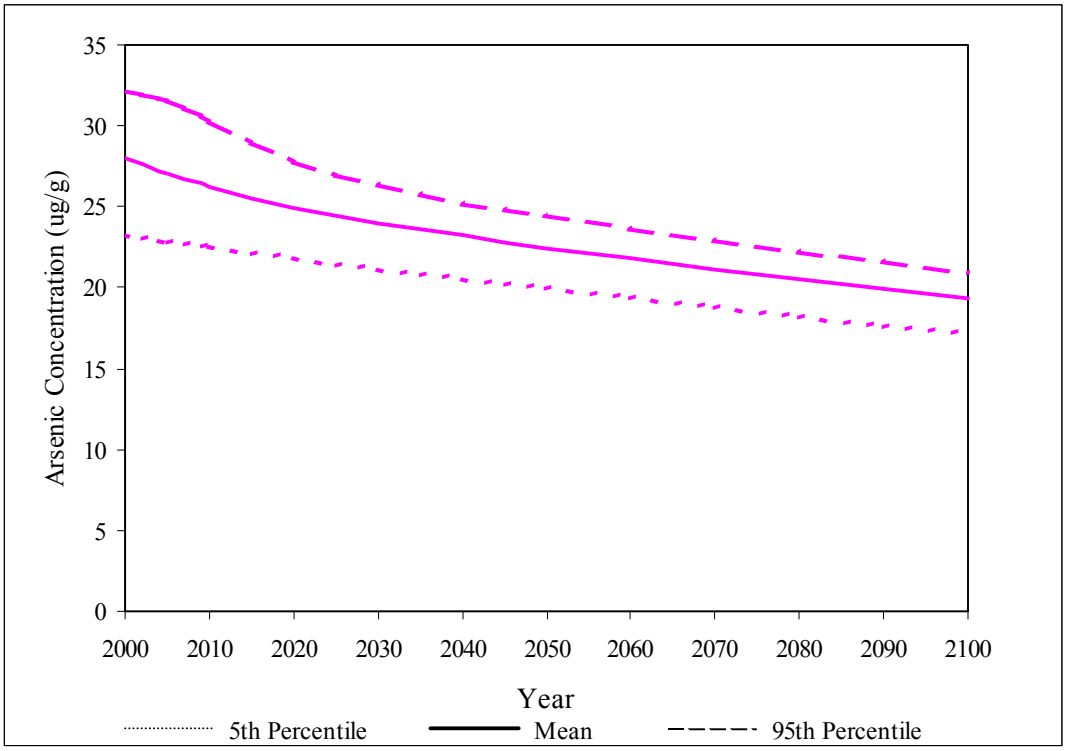
The background arsenic concentration in lake sediment has been estimated to equal 15 µg/g based on core sample results in Yellowknife Bay, as previously noted. It is noteworthy that this value exceeds the CCME (1999) TEL and the Thompson *et al*. (2002) benchmark values. Taking into account natural conditions in the Yellowknife study area, it is anticipated that existing arsenic levels in South Yellowknife Bay are not adversely affecting the benthic community; however, a field investigation would be required to confirm this assumption.

TABLE 6.1-5
COMPARISON OF PREDICTED (MEAN) ARSENIC CONCENTRATIONS IN SOUTH YELLOWKNIFE BAY SEDIMENT TO TOXICITY REFERENCE VALUES

	Sediment Concentration (µg/g)	CCME 1999		Thompson *et al.* 2002		Liber and Sobey 2000			
		TEL	PEL	LEL	SEL	NOEC	LOEC	Growth Effects	
								IC_{25}	IC_{50}
Toxicity Reference Values (µg/g)	-	5.9	17	10	346	-	39	174	342
		Screening Index Values							
Remediation Case – 2010	26.2	**4.4**	**1.5**	**2.6**	0.08	-	0.67	0.15	0.08
Remediation Case - 2100	19.3	**3.3**	**1.1**	**1.9**	0.06	-	0.49	0.11	0.06

Notes:
TEL Threshold Effect Level LEL Lowest Effect Level NOEC No-observed-effect-concentration
PEL Probable Effect Level SEL Severe Effect Level LOEC Lowest-observed-effect-concentration
Screening Index (SI) values equal the ratio of predicted arsenic concentrations to toxicity reference values.
Bold SI values indicate that the predicted concentration exceeds the toxicity reference value.

FIGURE 6.1-8
PREDICTED ARSENIC LEVELS – SOUTH YELLOWKNIFE BAY SEDIMENT



6.2 ECOLOGICAL ASSESSMENT

Potential toxic effects of arsenic can be measured at different levels of biological and ecological organization. In this study, ecological impacts of arsenic were characterized by the calculation of screening index values. Screening index values provide an integrated description of the potential hazard, the exposure (or dose)-response relationship and the exposure evaluation (U.S. EPA 1992, AIHC 1992). An index value is calculated by dividing the expected exposure or dose concentration by the selected arsenic toxicity reference value for each ecological receptor.

The ecological risk assessment reported below characterizes the nature and magnitude of potential impacts of arsenic carried with surface runoff from the mine site and surrounding landscape and future releases from the underground mine workings, as well as, the contributions of other background sources (e.g. soil, terrestrial vegetation). The following text discusses the results of the ecological exposure assessments for aquatic and terrestrial species that reside on Baker Creek, Back Bay, North Yellowknife Bay and South Yellowknife Bay. Uncertainties involved in the risk assessment include the estimates of arsenic loadings to Baker Creek from the mine site and surrounding landscape, from the underground mine workings and the treated effluent mine water discharge, the estimates of exposure concentrations and the toxicity data used to define the arsenic toxicity reference values for each of the ecological receptors. These

uncertainties are taken into account by specification of a range of input values in the model simulations.

Since the remediation efforts at the Giant Mine site will focus on both surface reclamation as well as the *in-situ* management of arsenic trioxide dust currently stored in vaults in the Giant Mine, both the aquatic and terrestrial environments will potentially be affected. Therefore, the ecological risk assessment included a variety of receptors receiving a large portion of their intake through water pathways, as well as, a variety of receptors receiving a large portion of the intake via terrestrial pathways. The aquatic receptors chosen for the assessment represent several trophic levels in typical creek and lake ecosystems. The screening index values for all species were estimated for the remediation case for the year of maximum predicted arsenic concentrations in the receiving waters and terrestrial environment. The results of 1000 probabilistic trials using the lake dispersion model (see Appendix B) were input to the pathways model (see Appendix D) to estimate a range of intakes or dose estimates for the ecological receptors.

Specifically, the risk assessment evaluated potential adverse effects of arsenic on:

- aquatic receptors (aquatic plants, benthic invertebrates, predatory fish and bottom feeder fish) based on the predicted arsenic concentrations in surface water; and
- terrestrial receptors (bear, caribou, moose, wolf, snowshoe hare, spruce grouse, mink, muskrat and three types of ducks – common merganser, mallard and scaup) based on estimated exposure to arsenic in water, aquatic food sources, sediment, soil and terrestrial vegetation.

The predicted concentrations of arsenic in water and sediment over time are presented in Section 6.1.

As previously discussed, the pathways analysis was carried out in a probabilistic framework (Monte Carlo) in order to investigate the effect of uncertainty in arsenic loadings and other model input parameters. Probability distribution functions were assigned to key parameters as shown in Appendix D. The model outputs (predictions) for the risk assessment component were processed to obtain summary statistics for presentation purposes. Mean, 5^{th} and 95^{th} percentile values are provided below.

It should be noted that the screening index (SI) values reported in this section are not estimates of the probability of ecological impact. Rather, the index values are positively correlated with the potential of an effect, that is, higher index values imply greater potential of an effect. Different magnitudes of the screening index have been used in other studies to screen for the potential ecological effects. A screening index value of 1.0 has been used in some instances (e.g. Suter 1991). In other work, Cadwell *et al.* (1993) suggested an index value of 0.3, based upon a cautious approach designed to account for potential chronic toxicity and chemical synergism. In

this study, an index value of 1.0 was used to examine the potential adverse effects of arsenic for aquatic receptors having endpoints of interest such as an effects level concentration (EC_{25}) for aquatic species and a Lowest Observable Adverse Effects Level (LOAEL) for terrestrial animals. Another reason for selecting an SI value of 1.0 is that calculations of arsenic in water and sediment incorporate background levels in addition to the source contributions.

6.2.1 Aquatic Impacts

Tables 6.2-1 to 6.2-4 summarize the screening index values calculated for the aquatic receptors in Baker Creek, Back Bay and North and South Yellowknife Bay, respectively, for the remediation scenario in years 2010 and 2100. The tables show the screening index values based on the predicted mean concentrations, as well as uncertainty bounds in the mean estimates (i.e. the predicted 5^{th} and 95^{th} percentile concentrations). This same information is displayed on Figures 6.2-1 to 6.2-4 in the form of bar charts for the respective water bodies.

The results for Baker Creek (Table 6.2-1 and Figure 6.1-1) show that even though the arsenic concentration is above the CCME surface water quality guideline for the protection of aquatic life in Baker Creek, aquatic plants, benthic invertebrates in the water column, bottom-feeding fish and predator fish are not expected to be adversely effected by the remediation scenario. The screening index values for fish are close to one in 2010 but decrease by 2100.

In a field study undertaken in 1975, Moore *et al.* (1978) found that the aquatic life in the upper reaches of Baker Creek was abundant and diverse, whereas below the mine very few species were encountered. The arsenic concentration in Baker Creek water downstream of the mine at the time was 9.1 mg/L (Moore *et al*. 1978). By contrast, the arsenic concentration currently averages approximately 0.08 mg/L near the mouth of the Baker Creek in the spring when there is no effluent discharge. In a survey of Baker Creek carried out in June 2002 (Dillon 2002b), both predator (northern pike) and bottom-feeding fish (long nose sucker) were found upstream and downstream of the mine site. This finding supports the results of this assessment which suggest that fish are not at risk of adverse effects at current arsenic levels in Baker Creek water when there is no effluent release to the watershed from the mine water treatment plant.

While the results of the aquatic assessment may seem to be contradictory to the sediment assessment, it should be pointed out that the aquatic assessment only considers water column impacts. As indicated in the previous section, further tests are necessary on sediments to establish a site-specific Toxicity Reference Value (TRV) for organisms residing in the sediment.

In Back Bay and Yellowknife Bay (see Tables 6.2-2 to 6.2-4 and Figures 6.2-2 to 6.2-4), the screening index values are all well below 1 which suggests that none of the aquatic species are expected to be adversely affected at the predicted arsenic concentrations in the water column.

In summary, the results of the aquatic ecological risk assessment demonstrate that there is the potential that the predicted arsenic levels could affect fish species in Baker Creek at arsenic loadings greater than assumed in the remediation scenario. This arsenic discharge scenario however, was predicted to be below levels which would have an adverse effect on the aquatic communities in Back Bay or Yellowknife Bay. Additional discussion regarding the ecological significance of the assessment results is provided in Section 6.2.3.

TABLE 6.2-1
SCREENING INDEX VALUES FOR AQUATIC SPECIES - BAKER CREEK – REMEDIATION CASE

	Aquatic Plants			**Benthic Invertebrates**			**Predatory Fish**			**Bottom-feeding Fish***		
Toxicity Reference Values (mg/L)	0.32			0.34			0.19			0.18		
Year	**Screening Index Values**											
	5th	Mean	95th	5th	Mean	95th	5th	Mean	95th	5th	Mean	95th
2010	0.32	0.37	0.43	0.30	0.35	0.40	0.72	0.84	0.97	0.84	0.98	**1.13**
2100	0.28	0.33	0.38	0.27	0.31	0.36	0.65	0.76	0.88	0.75	0.88	**1.03**

Notes: n/a – Not applicable as the habitat in Baker Creek is not conducive to the presence of predatory fish
* bottom-feeding fish such as lake whitefish and white sucker
Screening Index (SI) values equal the ratio of predicted arsenic concentrations to toxicity reference values.
Bold SI values indicate that the predicted concentration exceeds the toxicity reference value.

TABLE 6.2-2
SCREENING INDEX VALUES FOR AQUATIC SPECIES - BACK BAY – REMEDIATION CASE

	Aquatic Plants			**Benthic Invertebrates**			**Predatory Fish***			**Bottom-feeding Fish***		
Toxicity Reference Values (mg/L)	0.32			0.34			0.19			0.18		
Year	**Screening Index Values**											
	5th	Mean	95th	5th	Mean	95th	5th	Mean	95th	5th	Mean	95th
2010	0.009	0.009	0.010	0.008	0.009	0.010	0.020	0.022	0.024	0.023	0.025	0.028
2100	0.008	0.008	0.009	0.007	0.008	0.009	0.017	0.019	0.021	0.020	0.022	0.024

Notes: * predatory fish such as lake trout; bottom-feeding fish such as lake whitefish and white sucker
Screening Index (SI) values equal the ratio of predicted arsenic concentrations to toxicity reference values.
Bold SI values indicate that the predicted concentration exceeds the toxicity reference value.

TABLE 6.2-3
SCREENING INDEX VALUES FOR AQUATIC SPECIES - NORTH YELLOWKNIFE BAY – REMEDIATION CASE

	Aquatic Plants			Benthic Invertebrates			Predatory Fish			Bottom-feeding Fish		
Toxicity Reference Values (mg/L)	0.32			0.34			0.19			0.18		
Year	**Screening Index Values**											
	5th	Mean	95th	5th	Mean	95th	5th	Mean	95th	5th	Mean	95th
2010	0.004	0.004	0.005	0.004	0.004	0.005	0.009	0.010	0.011	0.011	0.012	0.013
2100	0.004	0.004	0.004	0.003	0.004	0.004	0.008	0.009	0.010	0.009	0.010	0.011

Notes: Screening Index (SI) values equal the ratio of predicted arsenic concentrations to toxicity reference values.
Bold SI values indicate that the predicted concentration exceeds the toxicity reference value

TABLE 6.2-4
SCREENING INDEX VALUES FOR AQUATIC SPECIES - SOUTH YELLOWKNIFE BAY – REMEDIATION CASE

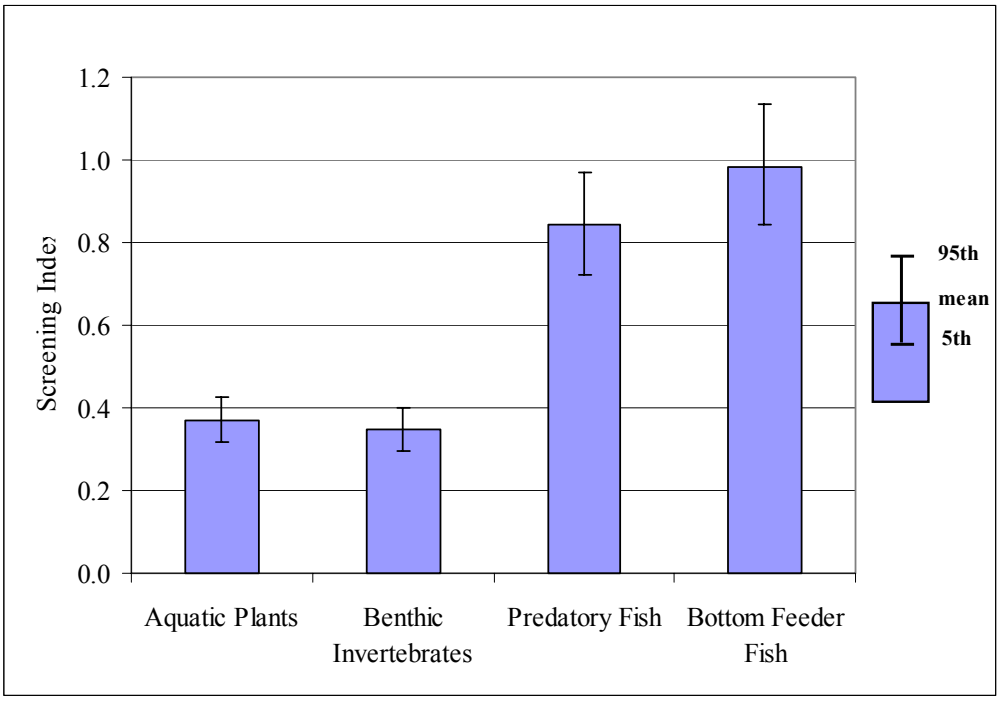
	Aquatic Plants			Benthic Invertebrates			Predatory Fish			Bottom-feeding Fish		
Toxicity Reference Values (mg/L)	0.32			0.34			0.19			0.18		
Year	**Screening Index Values**											
	5th	Mean	95th	5th	Mean	95th	5th	Mean	95th	5th	Mean	95th
2010	0.002	0.002	0.002	0.002	0.002	0.002	0.004	0.004	0.005	0.004	0.005	0.005
2100	0.002	0.002	0.002	0.001	0.002	0.002	0.004	0.004	0.004	0.004	0.004	0.005

Notes: Screening Index (SI) values equal the ratio of predicted arsenic concentrations to toxicity reference values.
Bold SI values indicate that the predicted concentration exceeds the toxicity reference value

6.2.1.1. Uncertainty in Assessment of Aquatic Species

As mentioned in Section 5.1.1, Environment Canada is in the process of reviewing the appropriate TRV to be used in aquatic assessments, and is considering lowering the TRV to an EC_{10} or lower benchmark. In recognition of this review, a sensitivity analysis was carried out using an EC_{10}. Tables 6.2-5 to 6.2-8 summarize the results using the EC_{10} values.

FIGURE 6.2-1
SCREENING INDICES FOR AQUATIC SPECIES - BAKER CREEK, 2010 - REMEDIATION CASE



SCREENING INDICES FOR AQUATIC SPECIES - BAKER CREEK, 2100 - REMEDIATION CASE

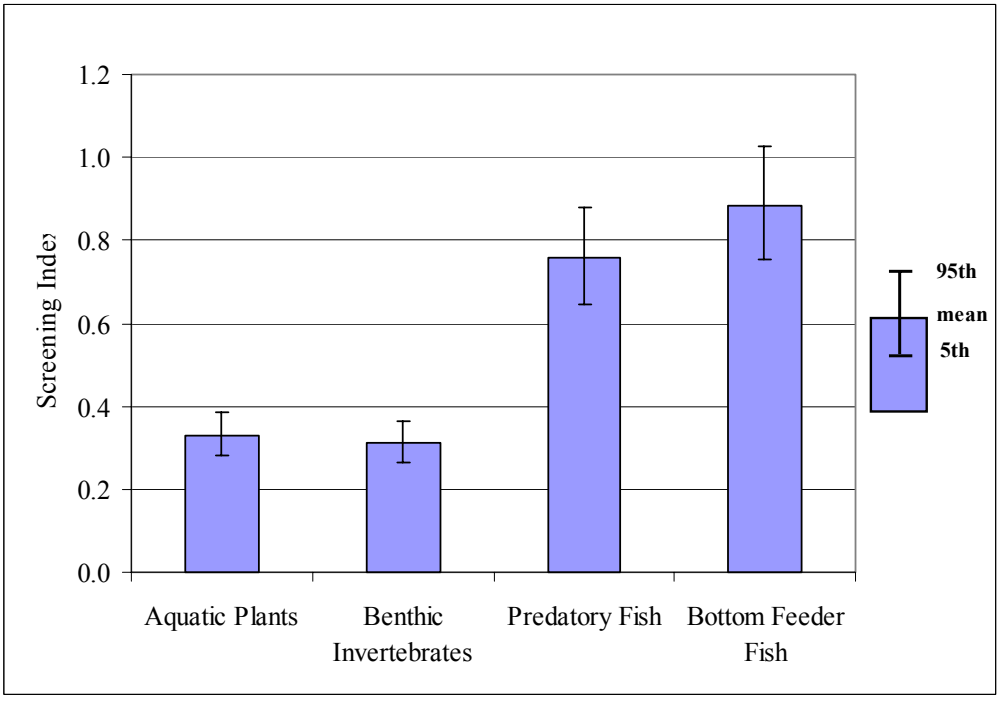
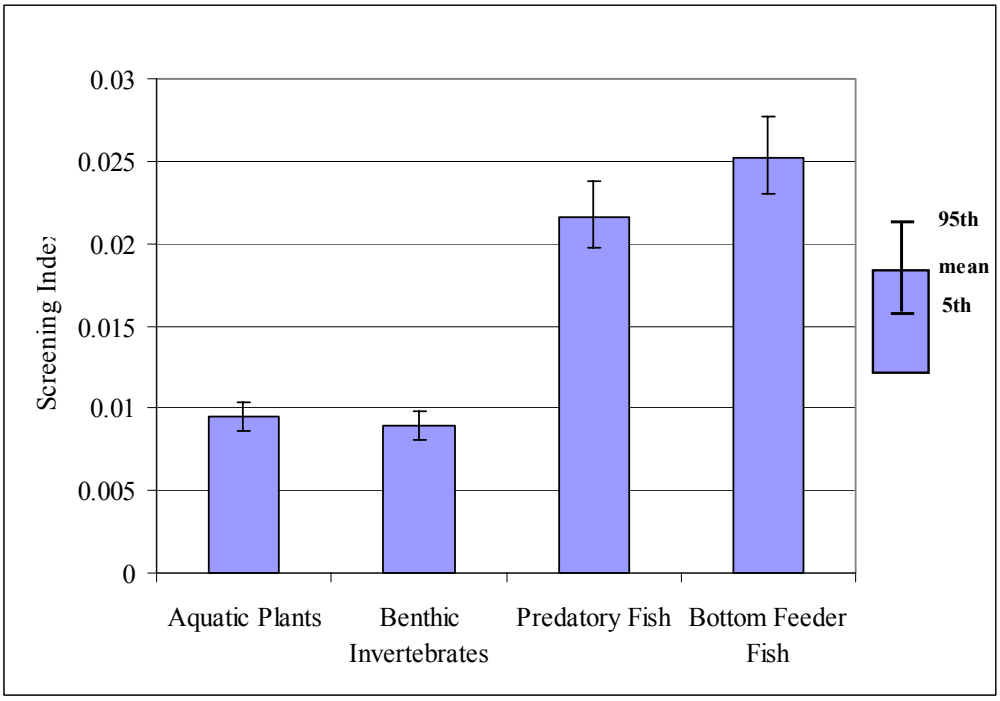


FIGURE 6.2-2
SCREENING INDICES FOR AQUATIC SPECIES - BACK BAY, 2010 – REMEDIATION CASE



SCREENING INDICES FOR AQUATIC SPECIES - BACK BAY, 2100 – REMEDIATION CASE

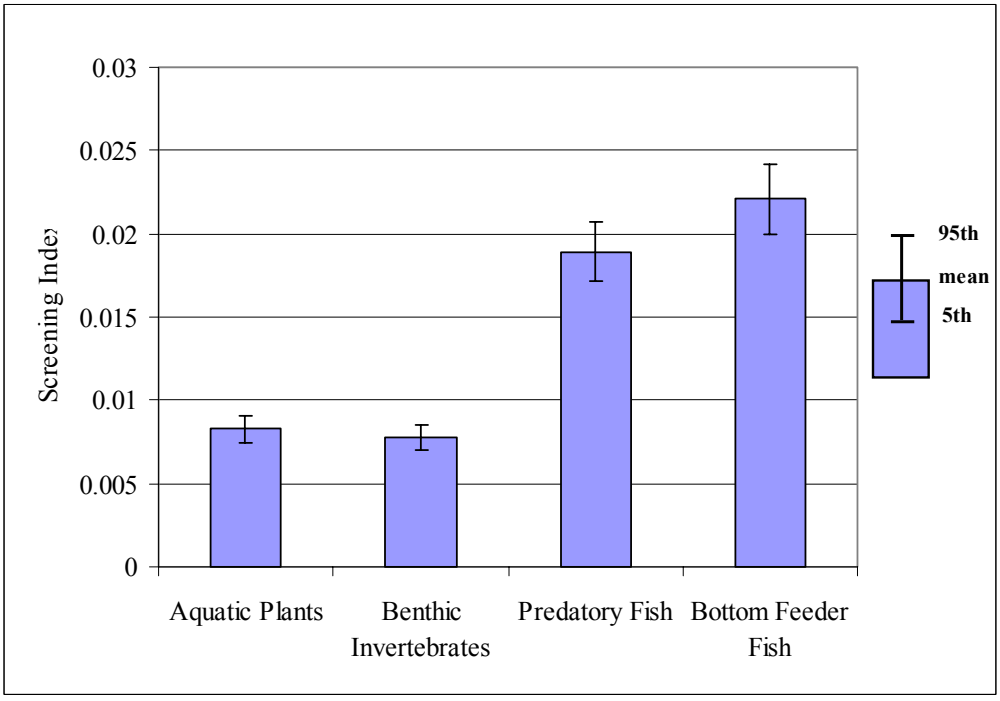
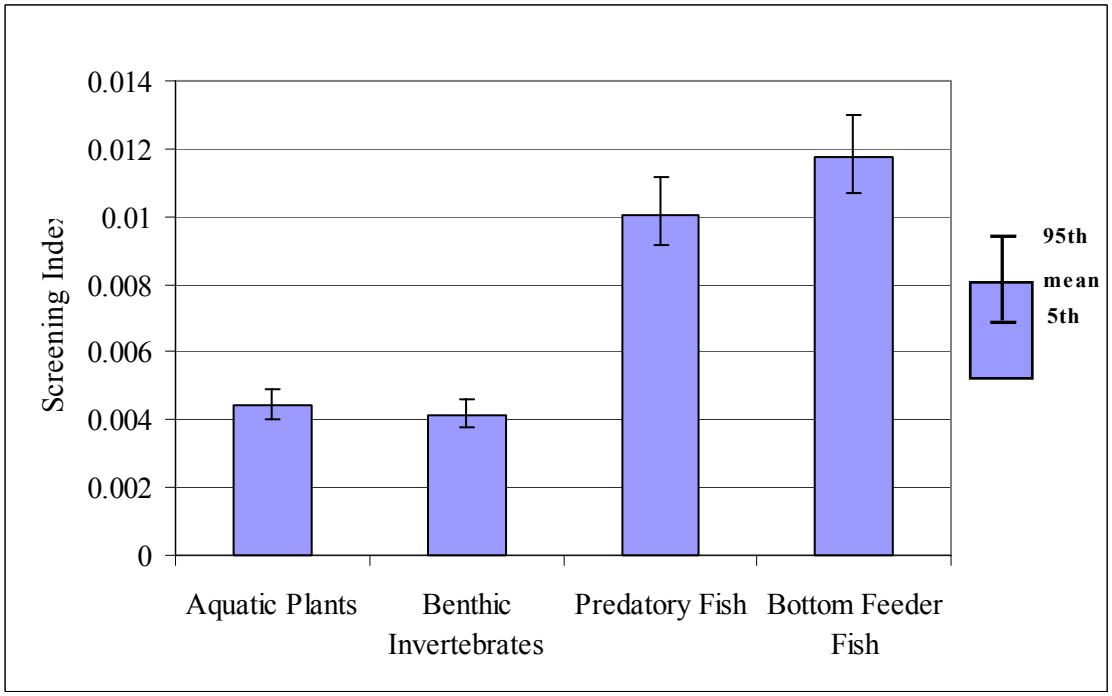


FIGURE 6.2-3
SCREENING INDICES FOR AQUATIC SPECIES - N. YELLOWKNIFE BAY, 2010 - REMEDIATION CASE



SCREENING INDICES FOR AQUATIC SPECIES - N. YELLOWKNIFE BAY, 2100 - REMEDIATION CASE

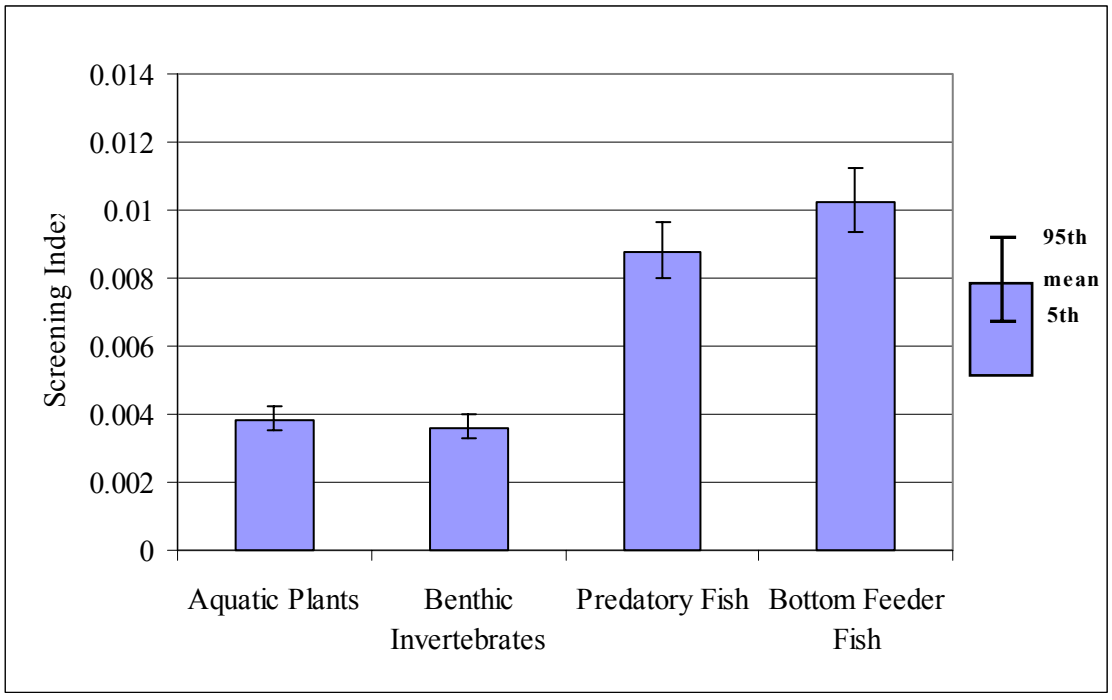
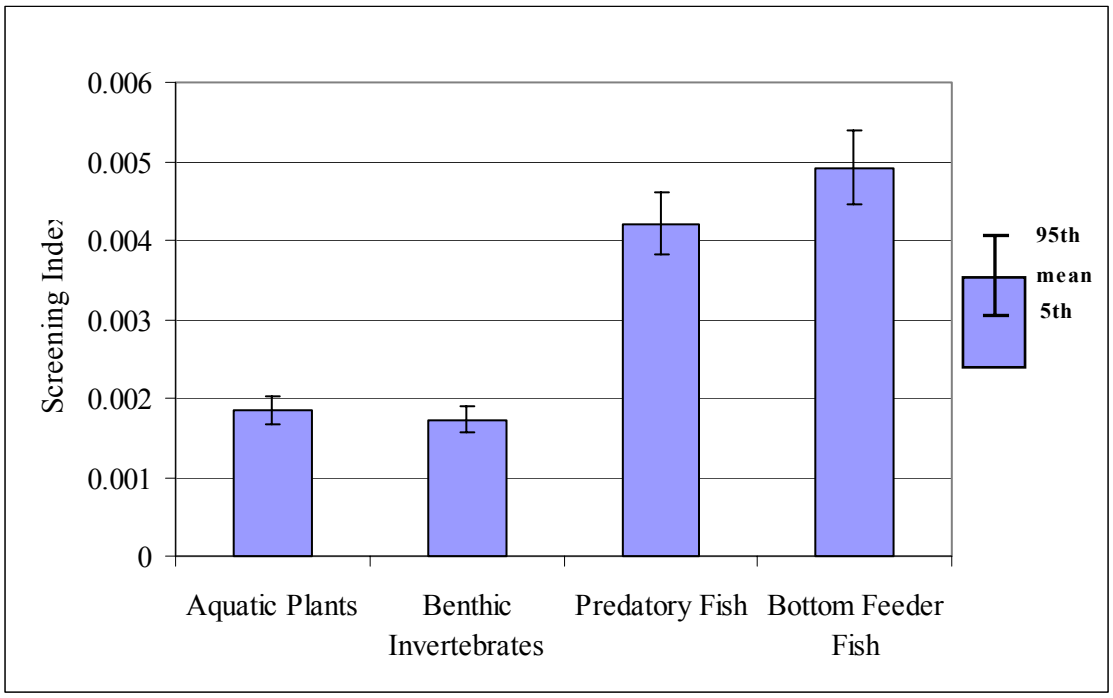


FIGURE 6.2-4
SCREENING INDICES FOR AQUATIC SPECIES - S. YELLOWKNIFE BAY, 2010 - REMEDIATION CASE



SCREENING INDICES FOR AQUATIC SPECIES - S. YELLOWKNIFE BAY, 2100 - REMEDIATION CASE

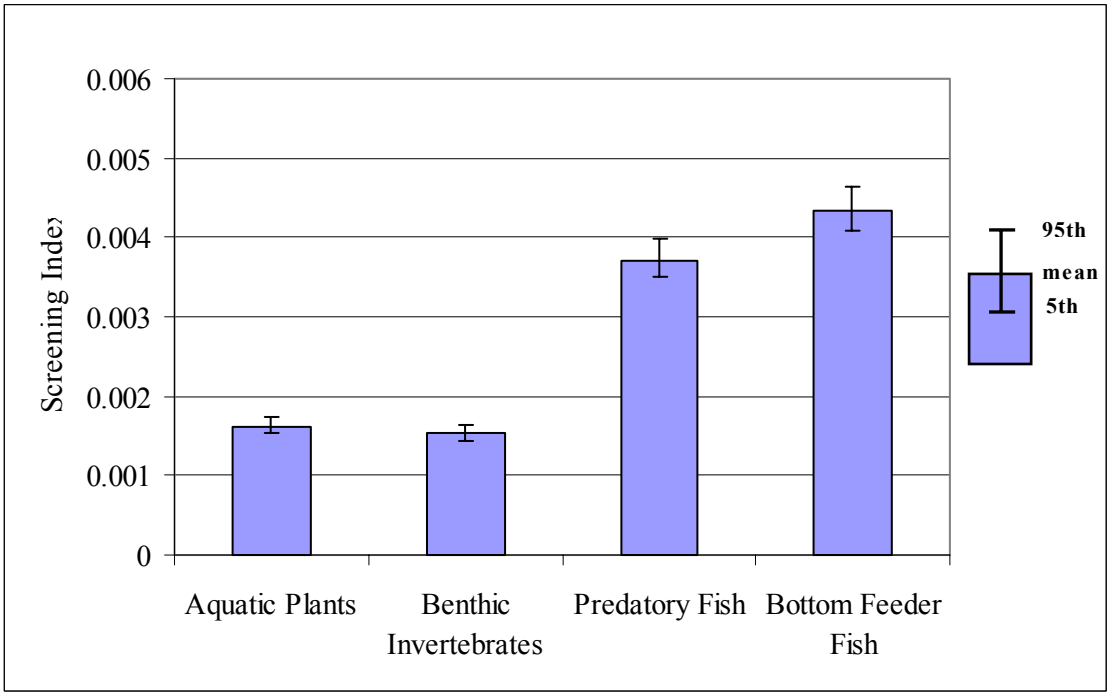


TABLE 6.2-5
SCREENING INDEX VALUES FOR AQUATIC SPECIES USING EC_{10} APPROACH-BAKER CREEK

Proposed Environment Canada Approach Using EC_{10} Values												
EC_{10} (mg/L)	Aquatic Plants			Benthic Invertebrates			Predatory Fish			Bottom-feeding Fish*		
	0.13			0.17			0.06			0.05		
Year	**Screening Index Values**											
	5th	Mean	95th	5th	Mean	95th	5th	Mean	95th	5th	Mean	95th
2010	0.80	0.94	***1.1***	0.59	0.69	0.80	***1.8***	***2.1***	***2.4***	***2.1***	***2.5***	***2.8***
2100	0.72	0.84	0.98	0.53	0.62	0.72	***1.6***	***1.9***	***2.2***	***1.9***	***2.2***	***2.6***

Notes: n/a – Not applicable as the habitat in Baker Creek is not conducive to the presence of predatory fish.
* bottom-feeding fish such as lake whitefish and white sucker.
Screening Index (SI) values equal the ratio of predicted arsenic concentrations to toxicity reference values.
Bold Italic SI values indicate that the predicted concentration exceeds the toxicity reference value.

TABLE 6.2-6
SCREENING INDEX VALUES FOR AQUATIC SPECIES USING EC_{10} APPROACH - BACK BAY

Proposed Environment Canada Approach Using EC_{10} Values												
EC_{10} (mg/L)	Aquatic Plants			Benthic Invertebrates			Predatory Fish			Bottom-feeding Fish*		
	0.13			0.17			0.06			0.05		
Year	**Screening Index Values**											
	5th	Mean	95th	5th	Mean	95th	5th	Mean	95th	5th	Mean	95th
2010	0.02	0.02	0.03	0.02	0.02	0.02	0.05	0.05	0.06	0.06	0.06	0.07
2100	0.02	0.02	0.02	0.01	0.02	0.02	0.04	0.05	0.05	0.05	0.06	0.06

Notes: * predatory fish such as lake trout; bottom-feeding fish such as lake whitefish and white sucker.
Screening Index (SI) values equal the ratio of predicted arsenic concentrations to toxicity reference values.
Bold Italic SI values indicate that the predicted concentration exceeds the toxicity reference value.

TABLE 6.2-7
SCREENING INDEX VALUES FOR AQUATIC SPECIES USING EC_{10} APPROACH - NORTH YELLOWKNIFE BAY

Proposed Environment Canada Approach Using EC_{10} Values												
EC_{10} (mg/L)	Aquatic Plants			Benthic Invertebrates			Predatory Fish			Bottom-feeding Fish*		
	0.13			0.17			0.06			0.05		
Year	**Screening Index Values**											
	5th	Mean	95th	5th	Mean	95th	5th	Mean	95th	5th	Mean	95th
2010	0.010	0.011	0.012	0.008	0.008	0.009	0.023	0.025	0.028	0.027	0.029	0.033
2100	0.009	0.010	0.011	0.007	0.007	0.008	0.020	0.022	0.024	0.023	0.026	0.028

Notes: * predatory fish such as lake trout; bottom-feeding fish such as lake whitefish and white sucker.
Screening Index (SI) values equal the ratio of predicted arsenic concentrations to toxicity reference values.
Bold Italic SI values indicate that the predicted concentration exceeds the toxicity reference value.

TABLE 6.2-8
SCREENING INDEX VALUES FOR AQUATIC SPECIES USING EC_{10} APPROACH - SOUTH YELLOWKNIFE BAY

Proposed Environment Canada Approach Using EC_{10} Values												
EC_{10} (mg/L)	Aquatic Plants			Benthic Invertebrates			Predatory Fish			Bottom-feeding Fish*		
	0.13			0.17			0.06			0.05		
Year	**Screening Index Values**											
	5th	Mean	95th	5th	Mean	95th	5th	Mean	95th	5th	Mean	95th
2010	0.004	0.005	0.005	0.003	0.003	0.004	0.010	0.011	0.012	0.011	0.012	0.013
2100	0.004	0.004	0.004	0.003	0.003	0.003	0.009	0.009	0.010	0.010	0.011	0.012

Notes: * predatory fish such as lake trout; bottom-feeding fish such as lake whitefish and white sucker.
Screening Index (SI) values equal the ratio of predicted arsenic concentrations to toxicity reference values.
Bold Italic SI values indicate that the predicted concentration exceeds the toxicity reference value.

As seen from the tables, the use of the lower TRVs (EC_{10} values) does not change the results of the assessment for Back Bay (Table 6.2-6), North Yellowknife Bay (Table 6.2-7) or South Yellowknife Bay (Table 6.2-8). In Baker Creek however, the use of the lower benchmark results in exposure for predator fish and bottom feeder fish exceeding the TRV, which suggests that there is a possibility for adverse effects in these species. However, surveys in Baker Creek in June 2002 (Dillon 2002b) indicate that both predator (northern pike) and bottom-feeder fish (long nose sucker) were found both upstream and downstream of the mine. This observation implies that the use of the EC_{10} as a benchmark may be overly conservative for species present in Baker Creek. However, more biological studies are needed on aquatic species in Baker Creek to validate the results of this assessment.

6.2.2 Terrestrial Species

The potential adverse effects on terrestrial species were evaluated by comparing the intake of arsenic in various terrestrial receptors to lowest observable adverse effects level (LOAELs) toxicity reference values (TRVs). An exceedance of a LOAEL TRV indicates the possibility of an adverse effect. In this assessment, it was assumed that all terrestrial receptors other than ducks were located along Baker Creek. Ducks were considered to also be present in Back Bay and Yellowknife Bay. The results of the analysis for terrestrial species, with the exception of ducks, are provided in Figures 6.2-5a to 6.2-5h.

Figures 6.2-5a to h provide the estimated arsenic intakes of the given ecological receptors for the remediation scenario. The intakes associated with the terrestrial and aquatic components of the animal diets are indicated on the figures to highlight whether aquatic or terrestrial pathways are the main contributors. The figures also provide the TRVs (both NOAEL and LOAEL) used in the assessment.

The figures indicate that arsenic in the terrestrial environment contributes to the majority of the estimated intake by caribou, grouse, hare and wolf, whereas the aquatic pathways are relatively minor. Estimated arsenic intakes by caribou, grouse and wolf, which were assumed to roam and obtain their drinking water and food from the mine site watershed during that fraction of the year they are in the study area, are seen to be well below the respective TRVs. It is concluded therefore, that these species are not expected to be adversely affected. In contrast, the estimated arsenic intake by hare, which were assumed to obtain all their drinking water and vegetation from the Baker Creek area, were generally equal to or greater than the LOAEL TRV. These results suggest that hare may potentially be adversely affected. As evident from examination of Figure 6.2-5e, ingestion of terrestrial vegetation accounts for the majority of the arsenic intake by hare.

For bear, mink, moose and muskrat, the aquatic pathways are predicted to dominate the arsenic intakes by each species. For the bear and moose, estimated arsenic intakes are well below the TRVs for these species. In contrast, the predicted arsenic levels in Baker Creek have the potential to adversely impact mink and muskrat. A field survey carried out in 2003 on Baker Creek identified several active muskrat dens along the entire reach of Baker Creek above and below the mine facilities, however, due to the timing of the survey there was no evidence of mink in the immediate area. Subsequently, muskrat were captured from both upstream and downstream locations for arsenic analyses on muscle and organ samples. The ecological significance of results of this investigation is discussed in Section 6.2.3.

Frequency histograms of the estimated intakes for muskrat, mink and hare (the most exposed terrestrial species) are provided in Figure 6.2-6a, 6.2-6b and 6.2-6c respectively (i.e. an arsenic

loading of 690 kg/y). The histograms are all seen to feature lognormal characteristics with a tail to the right however, the mink intake histogram is seen to be the most skewed. The shapes of the intake histograms are very much influenced by the arsenic content of the food sources. In the case of the mink, its diet comprises multiple items including duck, fish, hare, aquatic plants, benthic invertebrates and sediment. In contrast, the hare's diet is limited to forage and soil. The underlying arsenic content of these food sources is obviously quite different.

Figures 6.2-7a to 6.2-7c summarize the estimated intakes by ducks on Baker Creek and Back Bay. In each case, the water pathways represent the total arsenic intake. From the figures, it can be seen that the arsenic loadings result in intakes below the NOAELs and the LOAELs. Therefore, in both Baker Creek and Back Bay, the arsenic loadings do not represent a cause for concern for ducks.

6.2.3 Overall Ecological Significance

The ecological risk assessment has demonstrated that fish in Back Bay and Yellowknife Bay are not adversely impacted for the remediation scenario. Similarly, the risk assessment results suggest that fish in Baker Creek are not currently at risk, but are approaching the risk threshold above which they would be at risk.

With respect to benthic invertebrates, the risk assessment results indicate that there is a potential risk of lower species diversity in Baker Creek, Back Bay and perhaps North Yellowknife Bay at existing arsenic levels in the sediments. Field investigations on Baker Creek and Back Bay have shown reduced densities and species diversity in areas with elevated arsenic levels in the bottom sediments. Field findings therefore support the risk assessment results. In the future, the arsenic concentration in the sediments are predicted to decline; however, future field studies of benthic invertebrates will be necessary to determine whether adverse effects are occurring in benthic communities.

Besides the assessment of risks to aquatic species from exposure to arsenic in the water column and sediments of each water body, the pathways model was used to predict arsenic concentrations in aquatic biota including fish. Figure 6.2-8 provides a frequency histogram plot of the predicted arsenic concentrations in the flesh of fish (including predatory fish such as lake trout and bottom feeder fish such as lake whitefish) in Back Bay and North Yellowknife Bay for the remediation case (i.e. with a total arsenic loading of 690 kg/y to Baker Creek and Back Bay). The histograms are based on 1000 trials, which have been shown to be sufficient to provide a good definition of the output distribution in probabilistic simulations (SENES 1985). The skewed shape of the histograms is typical of environmental data and therefore is not unexpected. The figures provide a useful visual perspective of the possible outcomes of the model simulations.

FIGURE 6.2-5a,b,c,d
PREDICTED ARSENIC INTAKES BY CARIBOU, MOOSE, BEAR AND WOLF IN 2010

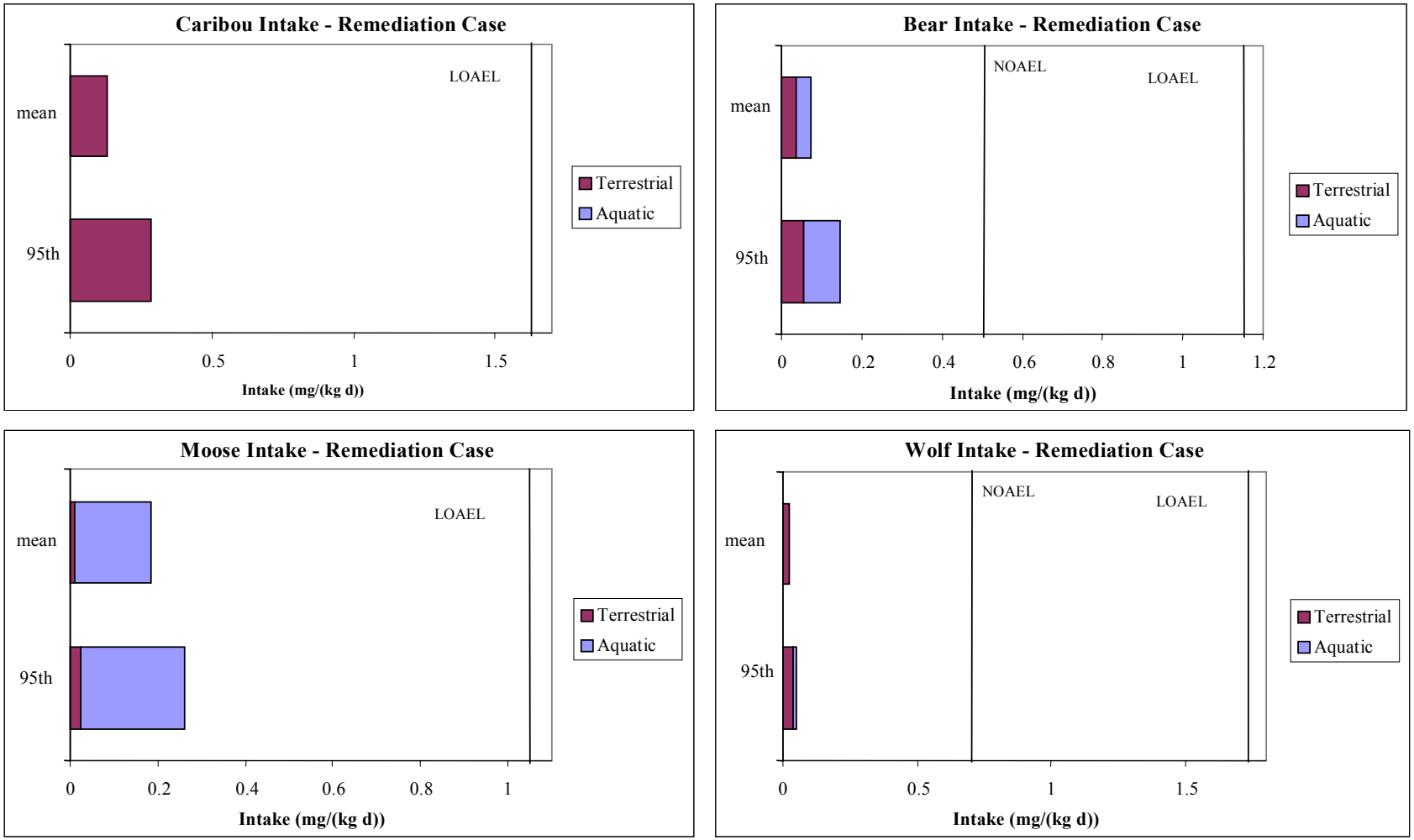


FIGURE 6.2-5 e,f,g,h
PREDICTED ARSENIC INTAKES BY HARE, GROUSE, MUSKRAT AND MINK IN 2010

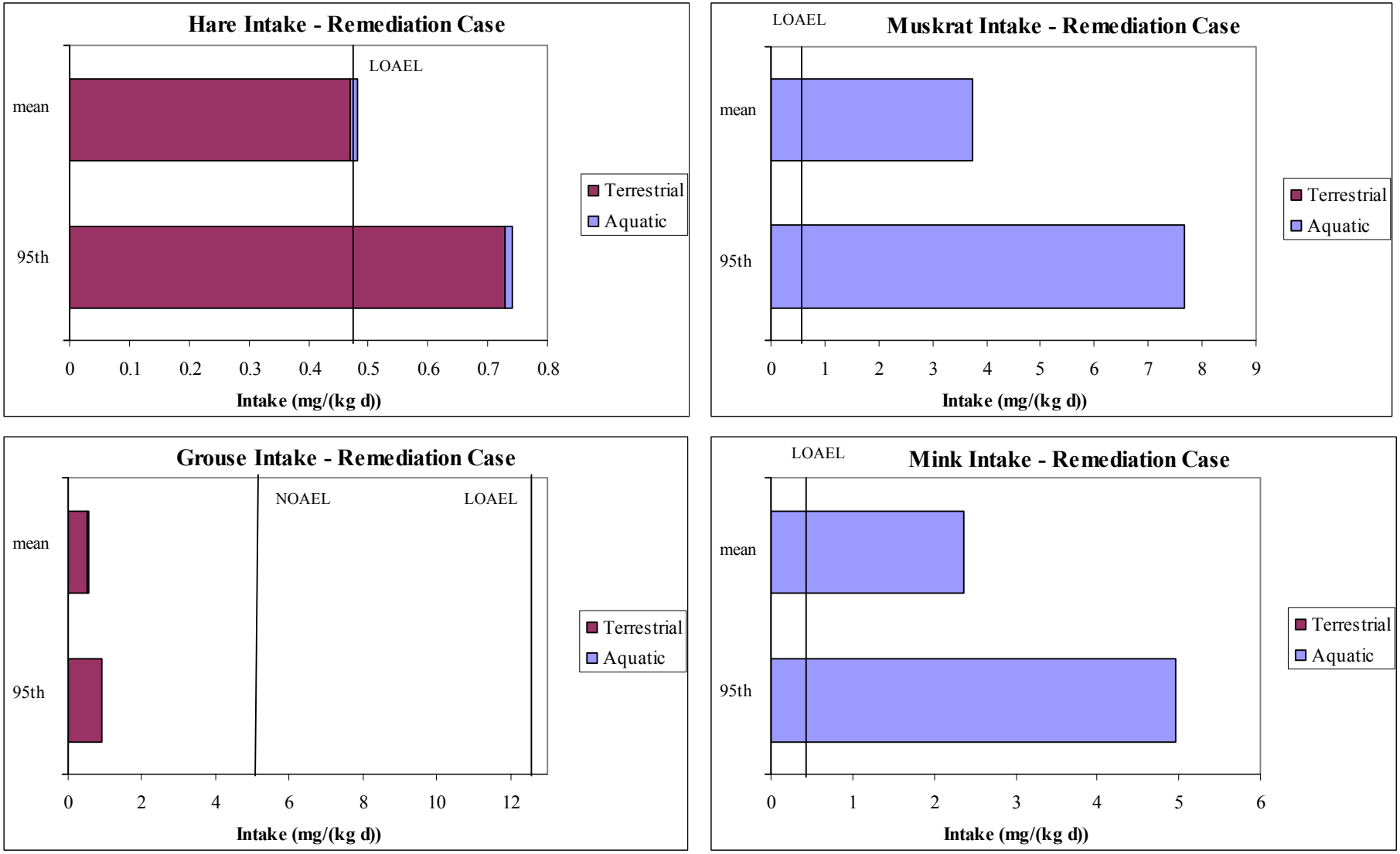


FIGURE 6.2-6a,b,c
FREQUENCY HISTOGRAM FOR MUSKRAT, MINK AND HARE INTAKE
2010, BAKER CREEK

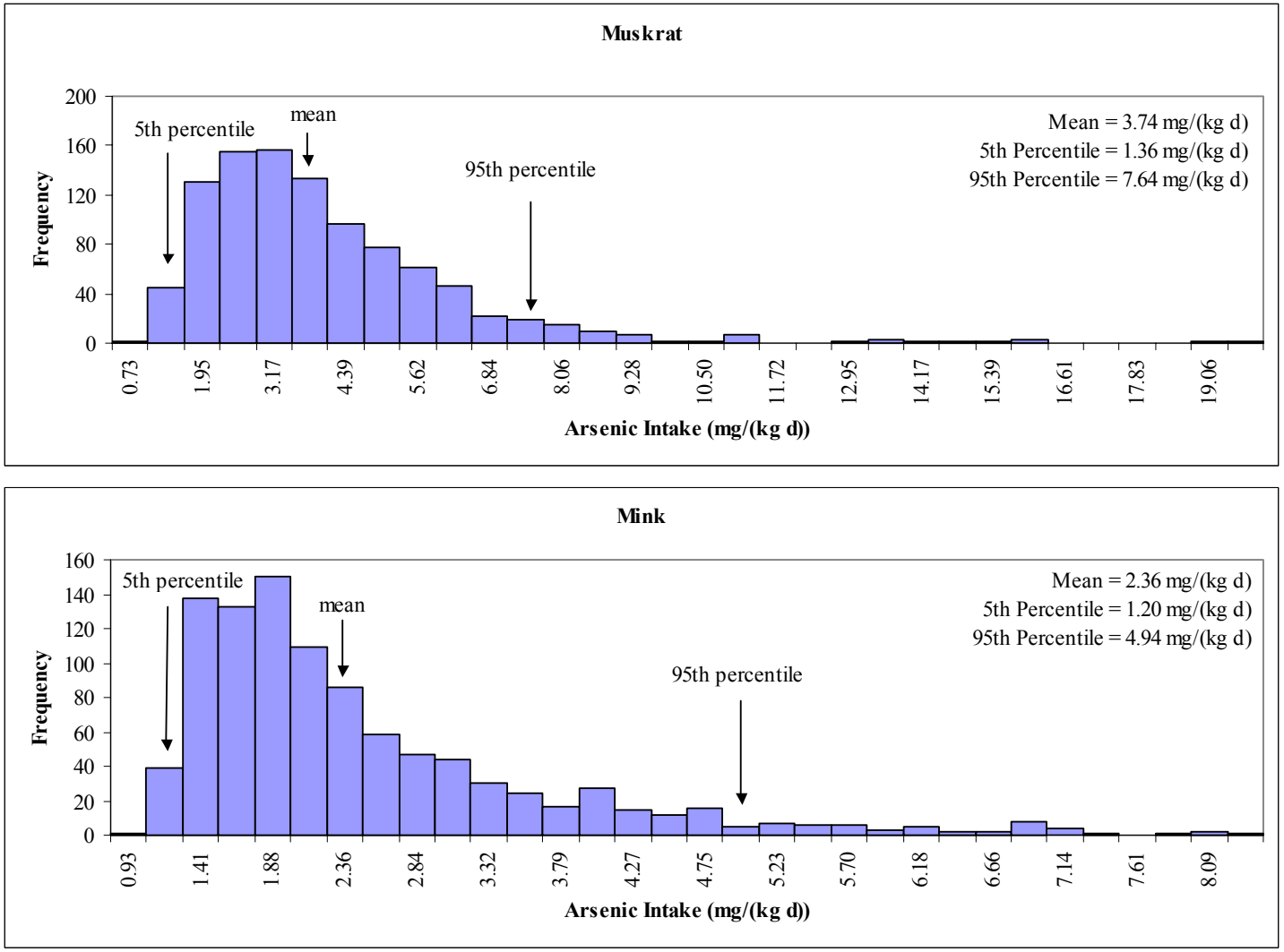


FIGURE 6.2-6a,b,c (Cont'd)
FREQUENCY HISTOGRAM FOR MUSKRAT, MINK AND HARE INTAKE
2010, BAKER CREEK

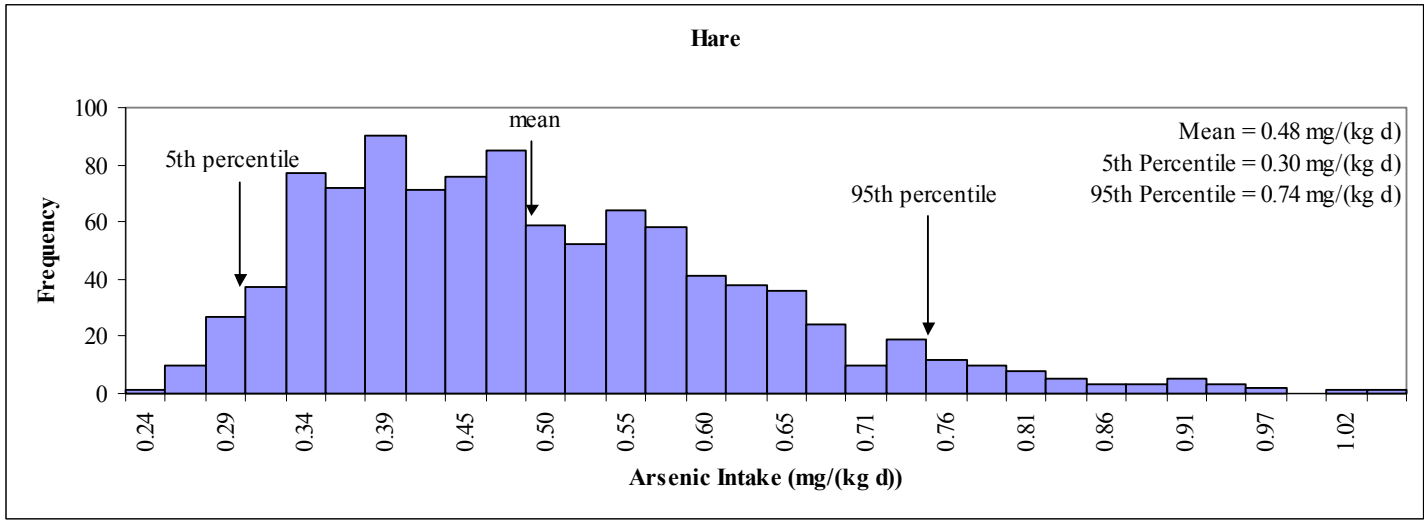
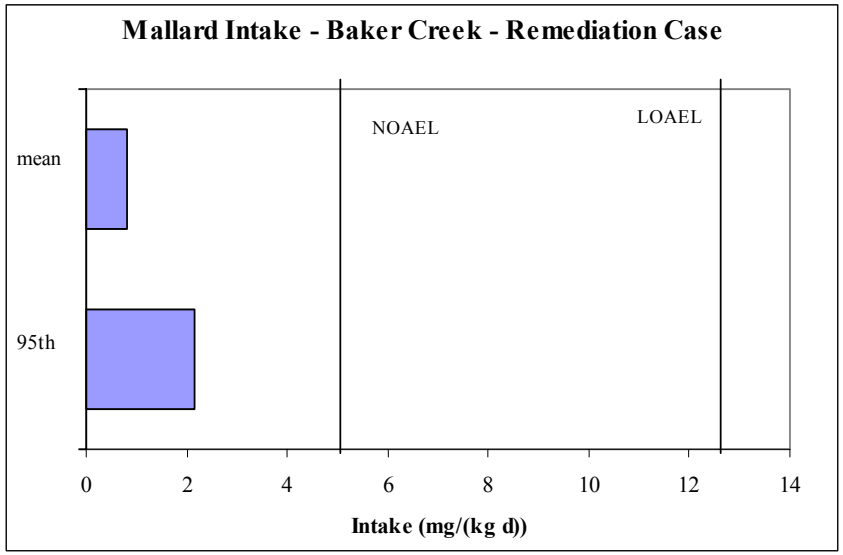
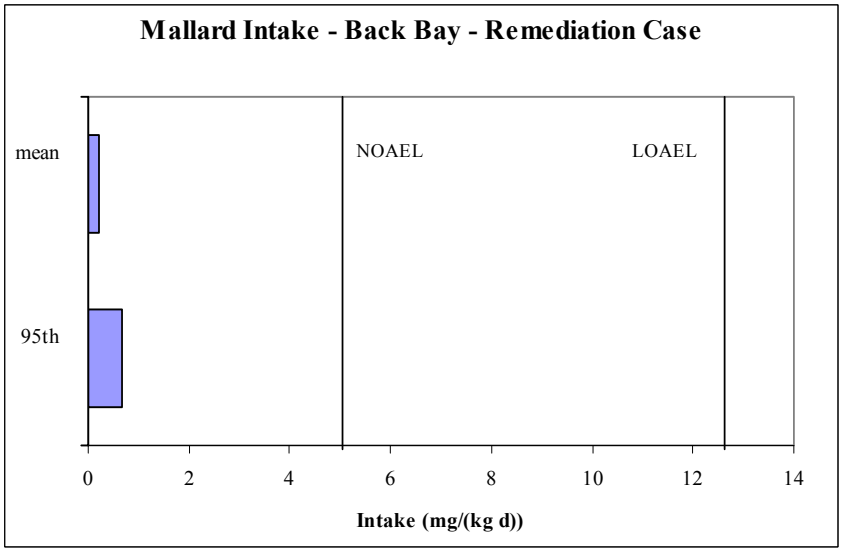
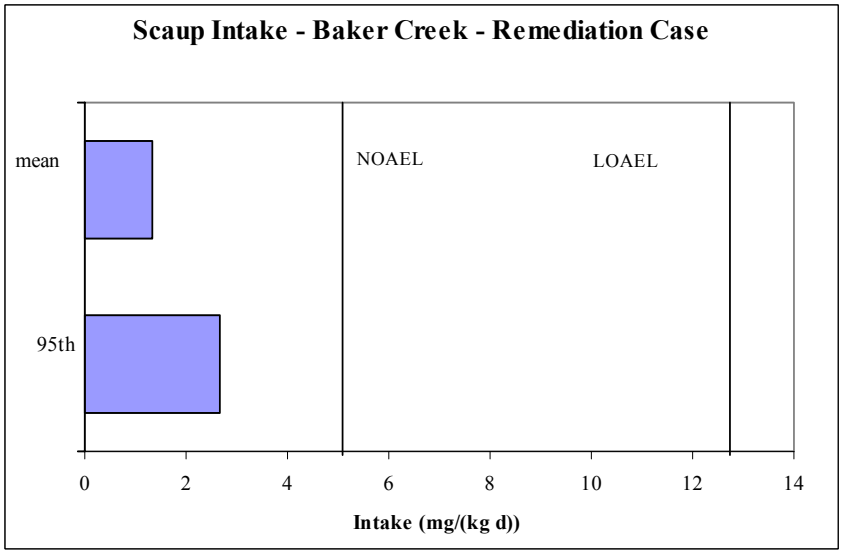


FIGURE 6.2-7a,b
COMPARISON OF ARSENIC INTAKES FOR DUCKS IN 2010 TO ARSENIC TOXICITY REFERENCE VALUES







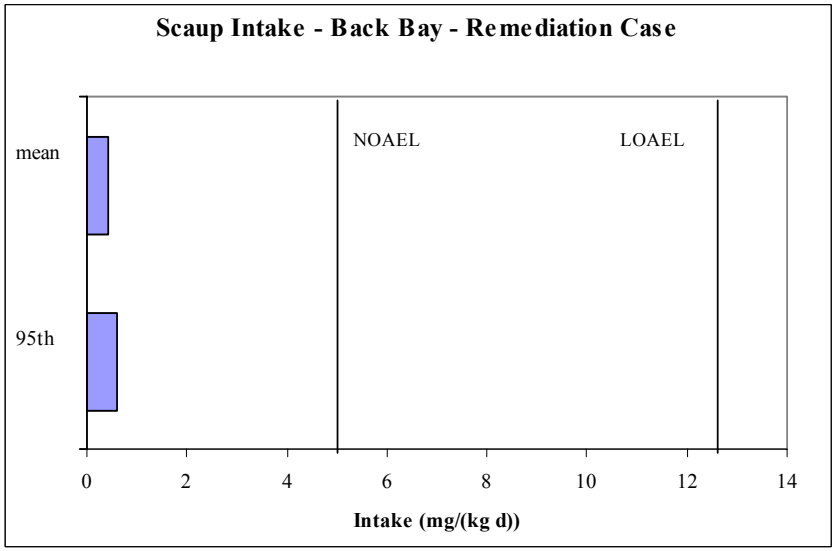
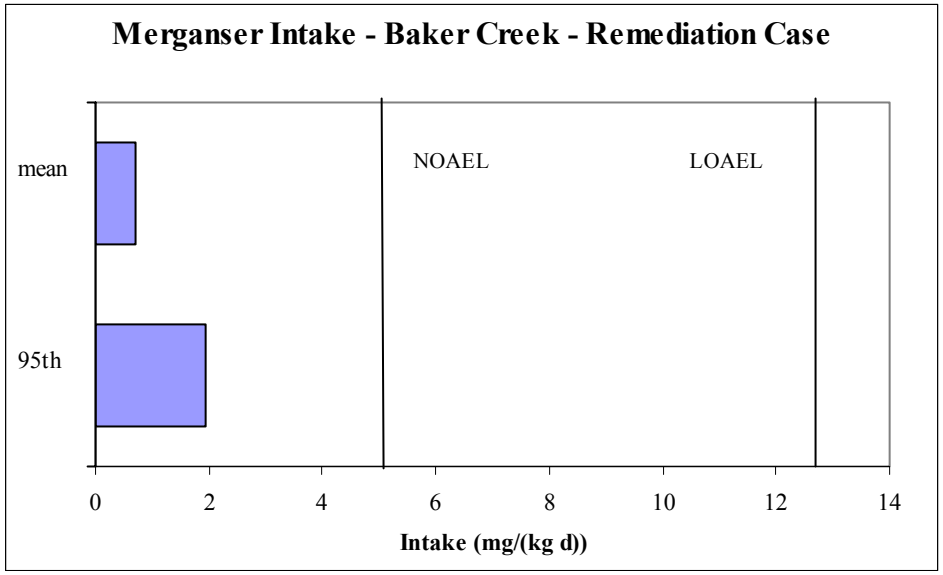


FIGURE 6.2-7c
COMPARISON OF ARSENIC INTAKES FOR DUCKS IN 2010 TO ARSENIC TOXICITY REFERENCE VALUES



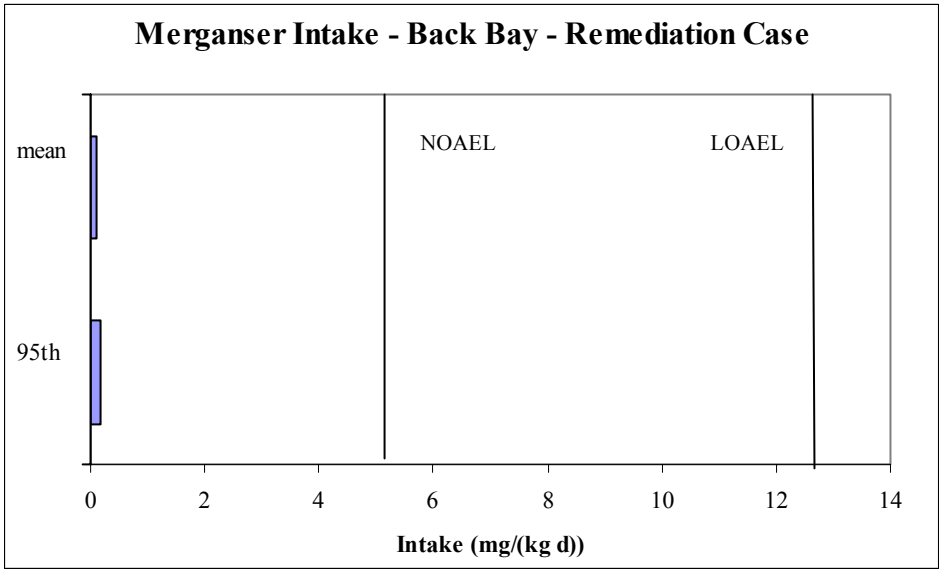


Table 6.2-9 provides a comparison of the predicted arsenic concentrations in fish tissue in Back Bay and Yellowknife Bay to measured arsenic fish tissue concentrations at different time intervals. As seen from the table, there is good agreement between the predicted and measured concentrations of arsenic in fish in Yellowknife Bay. The measured and predicted levels are both well below levels that have been found to affect the health or reproductive capacity of fish as reported by the Society of Environmental Toxicology and Chemistry (SETAC). A search of the SETAC database, which summarizes tissue concentrations and associated effects of arsenic and other metals on fish health, reports no effects levels in rainbow trout (the only species for which data were available) in the range of 1 to 3.4 µg/g wet weight tissue. As seen from inspection of Figure 6.2-8, the maximum predicted arsenic levels in fish flesh are well below the no effects range.

TABLE 6.2-9
PREDICTED AND MEASURED ARSENIC LEVELS IN FISH TISSUE

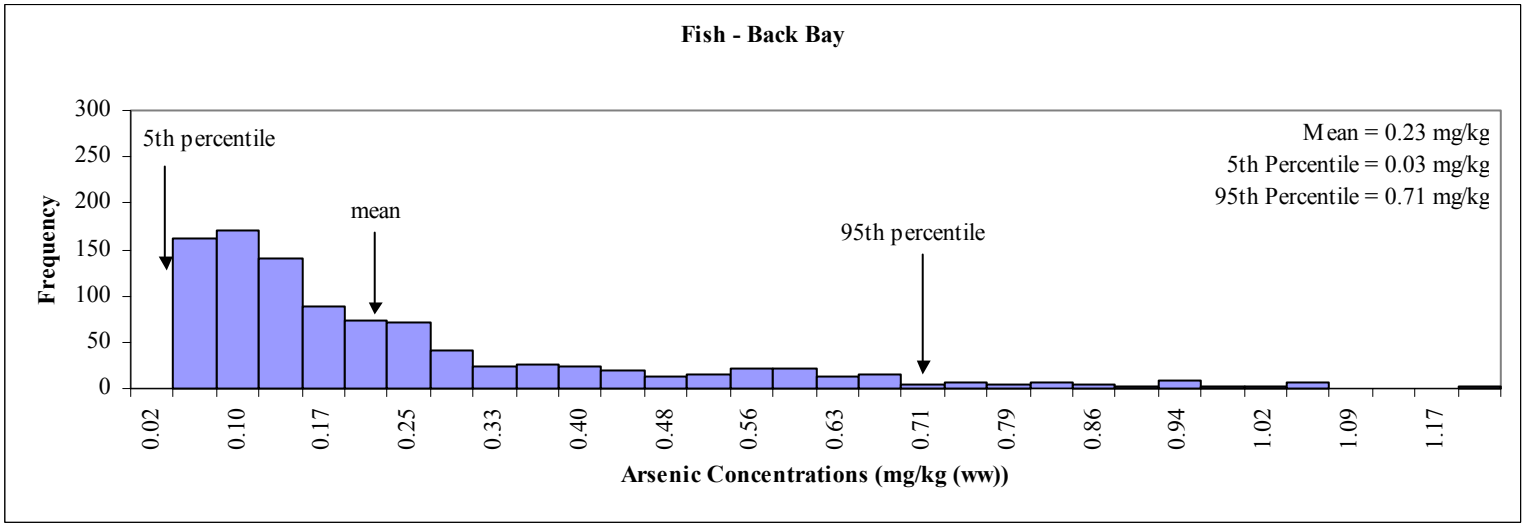
Time Period	Predicted Arsenic Concentrations (µg/g (ww))			Time Period	Measured Arsenic Concentrations (µg/g (ww))	
	Back Bay	North Yellowknife Bay	South Yellowknife Bay		Mean	Range
1990	0.41	0.17	0.06	1992 to 1993	0.16	0.01 to 1.11
1995	0.29	0.13	0.05	-	-	-
2003	0.27	0.12	0.05	2003	0.15	0.04 to 0.42

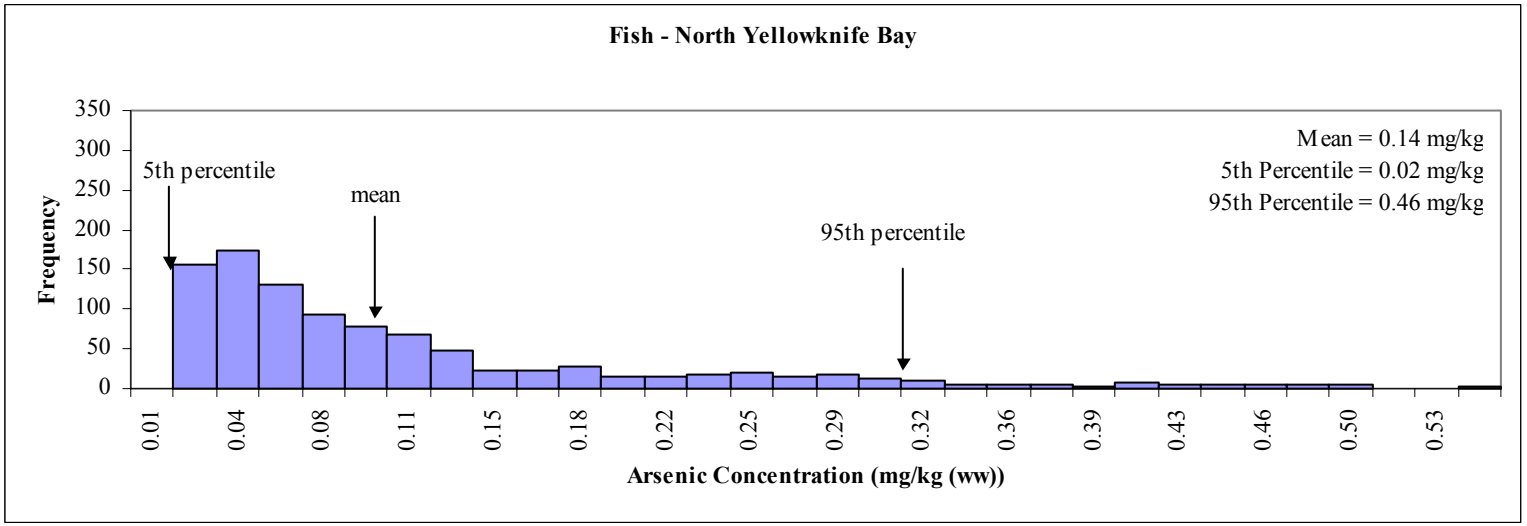
In Baker Creek, the predicted arsenic fish tissue concentrations range from 24 µg/g (ww) in 1990, to 8.92 µg/g (ww) in 2010. While the concentrations of arsenic in fish tissue in Baker Creek are predicted to decrease as time progresses, the predicted concentrations are above the range where adverse effects may be expected. The arsenic levels measured in sixteen fish samples collected from Baker Creek in 2002 by Dillon (2002b) at both upstream and downstream locations ranged between <2 to 30 µg/g (dw) (i.e., between <0.4 to 6 µg/g (ww) assuming a moisture content of 80%). In this investigation, the highest arsenic levels were measured on fish captured at the upstream location. A comparison of the predicted arsenic concentrations in fish in Baker Creek to the measured levels indicates a reasonable agreement; in fact the predicted levels are higher than the measured data.

The significance of the predicted and measured levels in fish from Baker Creek may be inferred by comparison to effects data reported in the literature. Data contained in the SETAC database indicates that at a concentration of between 3 to 13.5 µg/g (ww) a 50% decrease in growth and survival of rainbow trout (the only species for which data are reported) has been observed. While the data for rainbow trout are not necessarily directly applicable to other fish species, the data do support the results of the risk assessment that suggest that fish in Baker Creek may potentially be at risk of adverse effects.

FIGURE 6.2-8a,b

FREQUENCY HISTOGRAM OF PREDICTED ARSENIC CONCENTRATIONS IN PREDATOR AND BOTTOM FEEDER FISH PRESENT IN BACK BAY AND NORTH YELLOWKNIFE BAY IN 2010





The risk assessment also indicates that mink, muskrat and hare in the Baker Creek watershed could be potentially affected. To assess whether there was evidence of an adverse effect, biological surveys were carried out in 2003 and 2004 that specifically targeted the muskrat population on Baker Creek as described in Section 2.3.3. The results showed that, muskrat collected from downstream locations had, on average, approximately two times higher arsenic concentrations than the ones collected at upstream locations. Table 6.2-10 provides a summary of the measured data.

TABLE 6.2-10
SUMMARY OF MEASURED ARSENIC LEVELS IN TISSUES OF MUSKRATS FROM BAKER CREEK WATERSHED

	Measured Arsenic Concentrations (µg/g (ww))					
	Upstream			Downstream		
	Liver	Kidney	Muscle	Liver	Kidney	Muscle
Mean	0.66	0.7	0.24	1.39	2.64	0.51
Maximum	1.18	1.4	0.5	1.76	7.18	0.63

To provide a perspective on the measured levels of arsenic in muskrat tissue from the Giant Mine site area, a comparison between measured levels and literature-based reports on normal, high and toxic levels was prepared. Table 6.2-11 presents a summary of the available literature data regarding arsenic levels in a variety of domestic animals. As seen from this table, there is little difference between reported arsenic levels in liver and kidney of the animals listed in the table.

A comparison of the measured levels in the muskrat tissue from Baker Creek to the reported literature values indicates that arsenic levels measured upstream of the mine workings are higher than reported normal levels in muskrats. The mean measured levels in liver and kidney in downstream muskrats are higher than the normal levels but within the range of high arsenic levels reported for cattle, dogs and horses. The maximum measured tissue levels in muskrat from Baker Creek are in the lower range of the toxicity data reported in the literature for some of the other animals (e.g. rabbits, dogs, pigs).

Coupling this information with the results of biological survey carried out for muskrats, which indicated that there are active dens that support a substantial population on Baker Creek (Jacques Whitford 2003), it can be reasonably concluded that it is unlikely that the presence of arsenic in the sediments of Baker Creek is adversely affecting the muskrat population on Baker Creek. Such a conclusion can be drawn because more weight is given to field studies than to modelled risk assessment results. Field studies provide the evidence to substantiate or refute modelled conclusions.

If these results are extrapolated to mink or other small terrestrial mammals that have a significant aquatic based diet, it is unlikely that adverse effects will be observed in any of these populations. Biological surveys have not been conducted on the snowshoe hare to provide evidence supporting or refuting the results of the risk assessment, which identified hare as a receptor at risk of adverse health effects.

TABLE 6.2-11
SUMMARY OF REPORTED ARSENIC LEVELS IN TISSUES (µg/g (ww)) OF VARIOUS ANIMALS

	Cattle		Dogs		Horses		Pigs		Rabbit		Muskrat	
	Liver	**Kidney**	**Liver**	**Kidney**	**Liver**	**Kidney**	**Liver**	**Kidney**	**Liver**	**Kidney**	**Liver**	**Kidney**
Normal	0.004 – 0.4	0.018-0.4	<0.1-0.3	<0.1-0.4	<0.4	<0.4	0.003-0.2	0.003-0.1	<0.01-0.5	<0.01-0.5	<0.5	<0.5
High	1.0-50	1.5-5	0.5 –1	0.5-1	1.0-5.0	n/a	n/a	n/a	n/a	n/a	n/a	n/a
Toxic	7.0-100	3.5-38	>10	>10	7.0-15.0	>10	6.3-28	10.0-20	6.0-28.0	5.0-26.0	n/a	n/a

Note: n/a – not available.

6.3 HUMAN HEALTH ASSESSMENT

While the emphasis in this human health risk assessment is on the exposure of humans to arsenic releases from the Giant Mine to the aquatic and terrestrial environments, people living in the study area are exposed to arsenic via several other pathways. These pathways include inhalation of arsenic present in air and ingestion of soil and vegetation, which contain elevated levels of arsenic. People living in the study area may also be exposed indirectly to arsenic through consumption of fish and wildlife that obtain their water and food from Baker Creek, Back Bay and/or Yellowknife Bay.

For the purpose of estimating arsenic exposure to members of the public in the vicinity of the Giant Mine site, representative receptors were identified at four locations, as described in Section 3.2. A number of cautious assumptions were made in order to assess different levels of impacts on the human receptors. The assumptions include differences in the amount of country food eaten by each receptor; the source of drinking water and fish; the source of small game and large game; soil characteristics at the receptor locations; and the source and amount of locally grown garden produce and berries consumed by the receptors. The lifestyle and dietary characteristics assumed for the human receptors are detailed in Section 3.4. To account for these differences, a total of eight receptors were considered including 4 adults and 4 children (aged 5 to 11 years). A sensitivity analysis was also performed to explore the upper bound exposures experienced by receptors receiving higher-than-average arsenic exposure. Toddlers were also considered in the sensitivity analysis.

6.3.1 Background Exposure in Canada

Arsenic is ubiquitous in nature. It is ranked as the twentieth most abundant element in the earth's crust. Thus, exposure to arsenic in daily life occurs from background concentrations that are present in the air, water and food wherever a person lives. Typical concentrations of arsenic in drinking water supplies in Canada range from <1 μg/L to 5 μg/L. Arsenic is also present at low concentrations in most foods. Terrestrial animals and plants do not tend to accumulate arsenic.

Arsenic may enter the body by ingestion, inhalation, or by absorption through the skin. A number of factors influence how much arsenic is taken up in the body such as its speciation and solubility in body fluids. Environment Canada (1993) has carried out an assessment of exposure of Canadians to background levels of arsenic in air, water, soil and food. This study indicated that the major pathways of exposure to background levels of arsenic were ingestion of water and food. Air and soil pathways were insignificant contributors to overall exposure, representing approximately 0.1% and 1% of the typical exposure, respectively. Based on an arsenic concentration of 5 μg/L in drinking water supplies and background levels in food, Environment Canada estimated that a typical daily intake for an adult ranges from 1.0×10^{-4} mg/(kg d) to

7×10^{-4} mg/(kg d) and that the typical daily intake for a child (5 to 11 yrs) is from 2.0×10^{-4} mg/(kg d) to 2.1×10^{-3} mg/(kg d) (Environment Canada 1993).

6.3.2 Intake of Arsenic from Market (Store-bought) Foods

The information presented in Receveur *et al.* (1996) regarding the amount of market foods consumed in the Dogrib region was used for Receptors 2 (Latham Island) and 4 (Dettah). The intakes of market foods for Receptors 1 (Giant Mine Townsite) and 3 (City of Yellowknife) were based on intake rates for the general Canadian population (see Table 3.4-5). The market food intake rate was combined with the arsenic concentration in market foods presented by Dabeka *et al.* (1993), to provide an estimate of the arsenic exposure from market foods (described in Appendix D). It must be acknowledged that there is a large uncertainty in these arsenic market food intakes since they are for the general Canadian population and are being applied to this northern context. Additionally, there are other studies that present arsenic intakes in the diet (Meacher *et al.* 2002, Schoof *et al.* 1999, Yost *et al.* 1998, 2004). These studies provide dietary intakes of arsenic for adults and children which are similar to those derived using Dabeka *et al.* (1993). In fact, Yost *et al.* (1998) indicates that there a good agreement between the arsenic concentrations reported by the Ontario Ministry of the Environment (used in his analysis) and by Dabeka *et al.* (1993). It should also be noted that Health Canada in their recent update of the Arsenic Drinking Water Guideline cite the dietary intakes by Dabeka *et al.* (Health Canada 2004).

Table 6.3-1 provides a summary of the arsenic intakes from the consumption of market food input to the assessment of total arsenic intakes by each receptor. Differences in the arsenic intakes by Receptors 1 and 3 versus Receptors 2 and 4 reflect differences in the amounts of market foods consumed by the respective receptor groups.

Probability distribution functions based on the estimated arsenic intakes were developed from the numbers presented in Table 6.3-1, the functions were then sampled and added to the predicted arsenic intakes from other sources of arsenic on the Giant Mine site and the Yellowknife area to determine the total arsenic intakes for the receptors.

TABLE 6.3-1
ESTIMATED INTAKE OF ARSENIC FROM MARKET FOODS

Receptor	Estimated Arsenic Intake (mg/(kg d))			
	5^{th} Percentile	Median	Mean	95^{th} Percentile
1a, 3a	1.6×10^{-4}	5.0×10^{-4}	5.5×10^{-4}	1.1×10^{-3}
1c, 3c	4.5×10^{-4}	1.0×10^{-3}	1.1×10^{-3}	1.9×10^{-3}
2a, 4a	2.1×10^{-4}	2.8×10^{-4}	2.8×10^{-4}	3.7×10^{-4}
2c, 4c	4.0×10^{-4}	5.2×10^{-4}	5.2×10^{-4}	6.8×10^{-4}

Note: a = adult receptor; c = child receptor.
Intake rates are expressed as milligrams of arsenic per kilogram of body weight per day.

As well, it was assumed that the intake of arsenic from market foods was entirely in the inorganic form (i.e. the most toxic form). This is a cautious assumption as information contained in other assessments suggests that not all of the arsenic present in market foods exists as inorganic arsenic. A portion of the arsenic in market foods has been reported to be present as non-toxic organic arsenic.

6.3.3 Arsenic Exposure from Fish and Wildlife

In the Canadian population, exposure to inorganic arsenic is primarily from water, where nearly 100% is inorganic, and from food, which varies in its inorganic arsenic content. Yost (1998) suggests that inorganic arsenic may account for 21 to 40% of the total arsenic present in a mixed diet. Of the food commitment, most is from fish and seafood, which contain organic arsenic. Some studies indicate that arsenobetaine is the major organic compound in fish (Kohlmeyer *et al.* 2002, Maher 1985, Francesconi and Edmonds 1987, Hanaoka *et al.* 1987, Suner *et al*. 2002).

A study on the speciation of arsenic in fish from Great Slave Lake was conducted by Koch (1998). This study is of particular interest as the fish were collected in the Giant Mine site area, at the outlet of Baker Creek. While the information relates to a limited number of fish (9 samples), it provides an indication of the inorganic and organic content of the fish in Baker Creek. Some of the relevant points provided are:

- Inorganic arsenic (As(V)) was found at concentrations below the detection limit (<0.01 μg/g);
- Arsenobetaine was present in all fish species and individuals;
- Arsenosugar XI was present in suckers;
- The major arsenic compound in pike is DMA (dimethylarsenic acid);
- Arsenosugars may be broken down to DMA; and,
- Unknown arsenic-containing compounds elute with arsenosugars.

The above information indicates that the inorganic arsenic content of fish is very low; however, given that there were low extraction efficiencies, the inorganic content of the fish was not known precisely.

To address the issues surrounding the concentration of organic arsenic species in fish and to attempt to fill some of the data gaps in the Koch (1998) work, a study was performed to determine arsenic speciation and concentrations in fish from Back Bay close to the Giant Mine area (DIAND, 2004). The fish sample sizes were larger than for the Koch (1998) study, and included samples of lake whitefish, northern pike, walleye and white sucker. The measured arsenic concentrations were similar to those measured in the Koch (1998) study. However, although the analytical method used was able to detect concentrations of different arsenic

species, including As(III), As(V), DMA, MMA (monomethylarsonic acid) and arsenobutaine, greater than 50% of the organic arsenic in the fish tissue could not be directly identified. Only in northern pike could 90% of arsenic species be identified. Limitations of the analytical method included an inability to distinguish between methylated trivalent and pentavalent arsenic species, and an inability to measure trimethylated arsenic (TMA) compounds. In general, the study results suggest that the majority of arsenic in fish from Back Bay is found as organic arsenic species. As some organic compounds are believed to be just as toxic as inorganic arsenic, it was cautiously assumed that the unidentified organic species were toxic. The inorganic arsenic content of fish from the Yellowknife study area was found in the work referenced above to equal 3% of the total arsenic content with the remainder being in the organic form. Of the organic arsenic content in fish, the study could not distinguish the speciation of 78% of the organic arsenic. Therefore, for the purposes of this assessment it has been assumed that the organic arsenic contribution from fish was 78% toxic.

No literature data reporting arsenic speciation in caribou, duck, hare, grouse or moose meat were found. It was assumed therefore, that the arsenic intake from ingestion of small and large game was entirely inorganic arsenic, which possibly results in an overestimate of the true exposure.

6.3.4 Estimated Exposures to Toxic Arsenic (Inorganic and Toxic Organic Arsenic)

The 2010 estimated mean daily total arsenic intakes (including the contribution from toxic organic and inorganic forms of arsenic) by the eight adult and child receptors living in the study area are presented on Figures 6.3-1 to 6.3-4. The bar graphs presented on the figure show the contributions of market foods, terrestrial pathways and aquatic pathways to the total estimated daily intakes for both of the arsenic loading cases. The terrestrial pathways bar segments include the contributions from air inhalation, garden produce and berry consumption, caribou, hare and grouse consumption and soil ingestion. The aquatic pathways bar segments include water consumption, fish ingestion and moose and duck consumption. Moose and duck were included in the aquatic pathways as they obtain most of their arsenic intake from consumption of water, aquatic biota and sediments. Also shown on the figures are the typical ranges of daily intakes of inorganic arsenic by adults and children taken from an Environment Canada (1993) study.

Detailed summaries of the contributions of each pathway to the total daily intake estimates are provided in Table 6.3-2 for the remediation case in year 2010 when the intakes are highest. The pathways contributions are expressed as daily intake rates in mg/(kg d) (top half of the table) and as a percent of the mean daily intakes (bottom half of the table). The summary statistics presented on the tables include the 5^{th} percentile, mean, median (50^{th} percentile) and 95^{th} percentile values.

TABLE 6.3-2
ESTIMATED DAILY TOXIC ARSENIC INTAKES BY PATHWAY FOR EACH RECEPTOR FOR REMEDIATION CASE IN 2010

Receptor	Total (mg/kg/d)				Mean Intake (mg/kg/d)												
					Water	Inhalation	Vegetation	Fish	Meat						Soil	Medicinal Tea	Market Foods
	5th	Mean	Median	95th					Moose	Caribou	Hare	Grouse	Mallard	Total			
1a.Townsite - adult	$3.7x10^{-4}$	$8.8x10^{-4}$	$8.1x10^{-4}$	$1.6x10^{-3}$	$6.2x10^{-6}$	$9.0x10^{-7}$	$1.4x10^{-4}$	$1.0x10^{-4}$	$1.7x10^{-6}$	$3.2x10^{-5}$	$4.6x10^{-8}$	$3.4x10^{-6}$	$1.9x10^{-5}$	$5.6x10^{-5}$	$1.5x10^{-5}$	NA	$5.6x10^{-4}$
1c.Townsite - child	$8.4x10^{-4}$	$1.6x10^{-3}$	$1.5x10^{-3}$	$2.7x10^{-3}$	$7.0x10^{-6}$	$1.7x10^{-6}$	$2.3x10^{-4}$	$1.5x10^{-4}$	$2.6x10^{-6}$	$4.7x10^{-5}$	$7.1x10^{-8}$	$5.1x10^{-6}$	$2.8x10^{-5}$	$8.3x10^{-5}$	$7.1x10^{-5}$	NA	$1.1x10^{-3}$
2a.Latham Is. - adult	$4.0x10^{-4}$	$7.7x10^{-4}$	$6.7x10^{-4}$	$1.5x10^{-3}$	$6.1x10^{-6}$	$9.0x10^{-7}$	$2.6x10^{-5}$	$2.0x10^{-4}$	$6.7x10^{-6}$	$2.0x10^{-4}$	$1.2x10^{-7}$	$8.8x10^{-6}$	$1.4x10^{-5}$	$2.3x10^{-4}$	$8.7x10^{-6}$	$1.1x10^{-5}$	$2.8x10^{-4}$
2c.Latham Is. - child	$7.0x10^{-4}$	$1.3x10^{-3}$	$1.1x10^{-3}$	$2.5x10^{-3}$	$6.9x10^{-6}$	$1.7x10^{-6}$	$4.1x10^{-5}$	$3.4x10^{-4}$	$1.1x10^{-5}$	$3.5x10^{-4}$	$2.0x10^{-7}$	$1.5x10^{-5}$	$2.3x10^{-5}$	$4.0x10^{-4}$	$4.2x10^{-5}$	NA	$5.2x10^{-4}$
3a.Yellowknife - adult	$2.3x10^{-4}$	$6.7x10^{-4}$	$6.1x10^{-4}$	$1.3x10^{-3}$	$6.3x10^{-6}$	$9.0x10^{-7}$	$2.3x10^{-5}$	$4.3x10^{-5}$	$1.8x10^{-6}$	$3.4x10^{-5}$	$4.3x10^{-8}$	$3.2x10^{-6}$	$1.2x10^{-6}$	$4.1x10^{-5}$	$3.4x10^{-6}$	NA	$5.5x10^{-4}$
3c.Yellowknife - child	$6.2x10^{-4}$	$1.3x10^{-3}$	$1.2x10^{-3}$	$2.3x10^{-3}$	$6.9x10^{-6}$	$1.7x10^{-6}$	$3.7x10^{-5}$	$6.7x10^{-5}$	$2.7x10^{-6}$	$5.6x10^{-5}$	$6.7x10^{-8}$	$5.1x10^{-6}$	$2.0x10^{-6}$	$6.6x10^{-5}$	$1.6x10^{-5}$	NA	$1.1x10^{-3}$
4a.Dettah - adult	$3.3x10^{-4}$	$5.6x10^{-4}$	$4.8x10^{-4}$	$1.1x10^{-3}$	$6.1x10^{-6}$	$9.0x10^{-7}$	$1.7x10^{-5}$	$4.0x10^{-5}$	$1.1x10^{-6}$	$2.0x10^{-4}$	$1.2x10^{-7}$	$8.6x10^{-6}$	$7.3x10^{-7}$	$2.1x10^{-4}$	$1.8x10^{-6}$	$2.2x10^{-6}$	$2.8x10^{-4}$
4c.Dettah - child	$5.7x10^{-4}$	$1.0x10^{-3}$	$8.3x10^{-4}$	$2.0x10^{-3}$	$6.9x10^{-6}$	$1.7x10^{-6}$	$2.6x10^{-5}$	$6.5x10^{-5}$	$1.7x10^{-6}$	$3.5x10^{-4}$	$1.9x10^{-7}$	$1.5x10^{-5}$	$1.2x10^{-6}$	$3.6x10^{-4}$	$8.6x10^{-6}$	NA	$5.2x10^{-4}$

Receptor	Total (mg/kg/d)				Breakdown by Pathway (%)												
					Water	Inhalation	Vegetation	Fish	Meat						Soil	Medicinal Tea	Market Foods
	5th	Mean	Median	95th					Moose	Caribou	Hare	Grouse	Mallard	Total			
1a.Townsite - adult	$3.7x10^{-4}$	$8.8x10^{-4}$	$8.1x10^{-4}$	$1.6x10^{-3}$	0.7%	0.1%	16%	11%	0.2%	4%	<0.1%	0.4%	2.1%	6%	1.7%	NA	64%
1c.Townsite - child	$8.4x10^{-4}$	$1.6x10^{-3}$	$1.5x10^{-3}$	$2.7x10^{-3}$	0.4%	0.1%	14%	9%	0.2%	3%	<0.1%	0.3%	1.7%	5%	4.4%	NA	67%
2a.Latham Is. - adult	$4.0x10^{-4}$	$7.7x10^{-4}$	$6.7x10^{-4}$	$1.5x10^{-3}$	0.8%	0.1%	3%	26%	0.9%	26%	<0.1%	1.1%	1.8%	30%	1.1%	1.5%	37%
2c.Latham Is. - child	$7.0x10^{-4}$	$1.3x10^{-3}$	$1.1x10^{-3}$	$2.5x10^{-3}$	0.5%	0.1%	3%	25%	0.8%	26%	<0.1%	1.1%	1.7%	30%	3.1%	NA	39%
3a.Yellowknife - adult	$2.3x10^{-4}$	$6.7x10^{-4}$	$6.1x10^{-4}$	$1.3x10^{-3}$	0.9%	0.1%	4%	6%	0.3%	5%	<0.1%	0.5%	0.2%	6%	0.5%	NA	82%
3c.Yellowknife - child	$6.2x10^{-4}$	$1.3x10^{-3}$	$1.2x10^{-3}$	$2.3x10^{-3}$	0.5%	0.1%	3%	5%	0.2%	4%	<0.1%	0.4%	0.2%	5%	1.3%	NA	85%
4a.Dettah - adult	$3.3x10^{-4}$	$5.6x10^{-4}$	$4.8x10^{-4}$	$1.1x10^{-3}$	1.1%	0.2%	3%	7%	0.2%	35%	<0.1%	1.5%	0.1%	37%	0.3%	0.4%	50%
4c.Dettah - child	$5.7x10^{-4}$	$1.0x10^{-3}$	$8.3x10^{-4}$	$2.0x10^{-3}$	0.7%	0.2%	3%	7%	0.2%	35%	<0.1%	1.5%	0.1%	37%	0.9%	NA	52%

Note: Total arsenic includes toxic organic and inorganic arsenic.

Receptors 1a and 1c – Giant Mine Townsite Residents

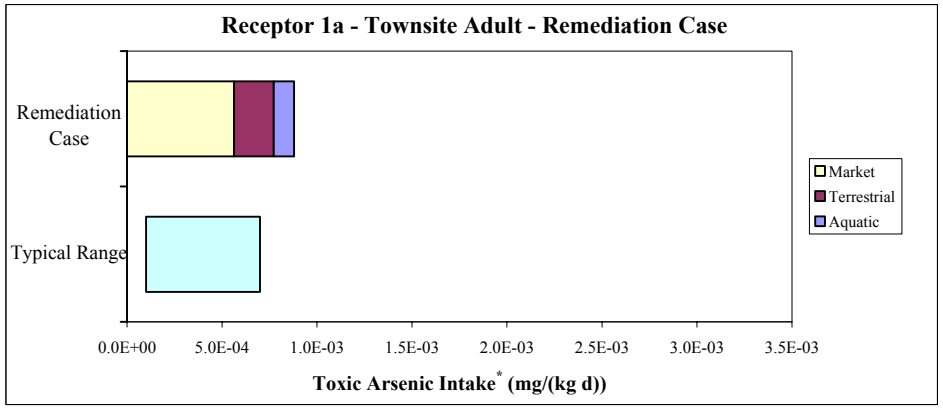
The estimated mean total daily toxic arsenic intakes for Receptors 1a and 1c with a diet similar to that of a typical Canadian, are presented on Figure 6.3-1. These receptors were assumed to obtain their drinking water from the municipal supply, and soil, garden produce and berries from the Giant Mine Townsite. It was assumed that ducks and fish were obtained from Back Bay, while large game and small game were assumed to be taken from the Baker Creek watershed to maximize the exposure estimate.

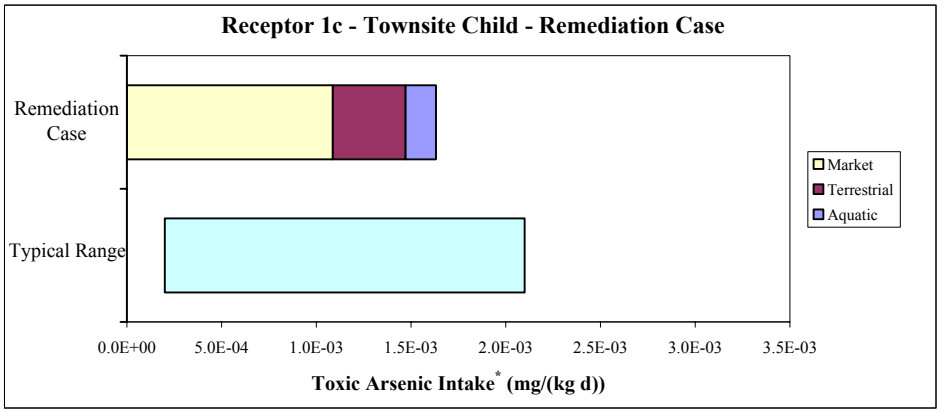
The main pathway of arsenic intake by Receptor 1a and 1c in the remediation case is associated with consumption of market foods, as demonstrated in Figure 6.3-1. The aquatic exposure pathway contribute the least amount of the total toxic arsenic intake estimates. Details of the contributions of all pathways are summarized in Table 6.3-2. A review of this table indicates that the main contributors to the intakes by Receptor 1a and 1c associated with the Giant Mine site and Yellowknife study area are backyard garden produce (14 to 16%) and fish (9 to 11%). Soil contributes up to 4.4% of the total intake.

In comparison to the arsenic intake range reported for Canadian children, the estimated intakes of total arsenic for Receptor 1c fall within the range in the remediation scenario. For the adult receptor, however, the estimated total toxic arsenic intakes exceed the typical range. As seen from Table 6.3-2, the drinking water pathway accounts for between 0.4% and 0.7% of the arsenic intake, whereas consumption of food and soil accounts for about 99% of the intakes.

The results of the sensitivity analyses performed for this receptor are presented in Table 6.3-3. It is clear that arsenic exposures for the Giant Mine Townsite adult and child receptors are anticipated to be higher when the source of fish is changed from Back Bay to Baker Creek. This results from the higher arsenic levels present in Baker Creek. The second sensitivity analysis explored the removal of garden produce from the diet. Arsenic exposures for this analysis are slightly lower than those experienced under the remediation scenario. However, the reduction is not significant (i.e. from 8.8×10^{-4} mg/(kg d) for adult in the basic scenario to 7.7×10^{-4} mg/(kg d) in the sensitivity analysis). In conclusion, the sensitivity analyses showed that the source of fish can have a large impact on exposure, while the consumption of garden produce has limited impact on exposure for individuals living at the Giant Mine Townsite.

FIGURE 6.3-1
ESTIMATED MEAN DAILY INTAKE OF TOXIC ARSENIC BY RECEPTOR 1 (GIANT MINE TOWNSITE) ADULT AND CHILD IN 2010





* Includes inorganic + toxic organic arsenic. Note all modelled market foods, 78% of fish and all other aquatic and terrestrial intakes were assumed to be in a toxic form. Typical range estimates are from Environment Canada (1993) and are an estimate of the inorganic arsenic only.

TABLE 6.3-3
RESULTS OF SENSITIVITY ANALYSIS FOR RECEPTORS 1a AND 1c
(GIANT MINE TOWNSITE)

Receptor	Analysis	Description	Total Toxic Arsenic Exposure (mg/(kg d))			
			5th	Mean	Median	95th
Townsite – Adult	Base Case	Fish from Back Bay; Garden produce	3.7×10^{-4}	8.8×10^{-4}	8.1×10^{-4}	1.6×10^{-3}
Townsite – Child	Base Case	Fish from Back Bay; Garden produce	8.4×10^{-4}	1.6×10^{-3}	1.5×10^{-3}	2.7×10^{-3}
Townsite – Adult	1	Fish from Baker Creek	1.1×10^{-3}	4.7×10^{-3}	3.4×10^{-3}	1.3×10^{-2}
Townsite – Child	1	Fish from Baker Creek	1.8×10^{-3}	7.4×10^{-3}	5.2×10^{-3}	2.0×10^{-2}
Townsite – Adult	2	No garden produce	3.2×10^{-4}	7.7×10^{-4}	7.0×10^{-4}	1.5×10^{-3}
Townsite – Child	2	No garden produce	7.6×10^{-4}	1.5×10^{-3}	1.4×10^{-3}	2.6×10^{-3}

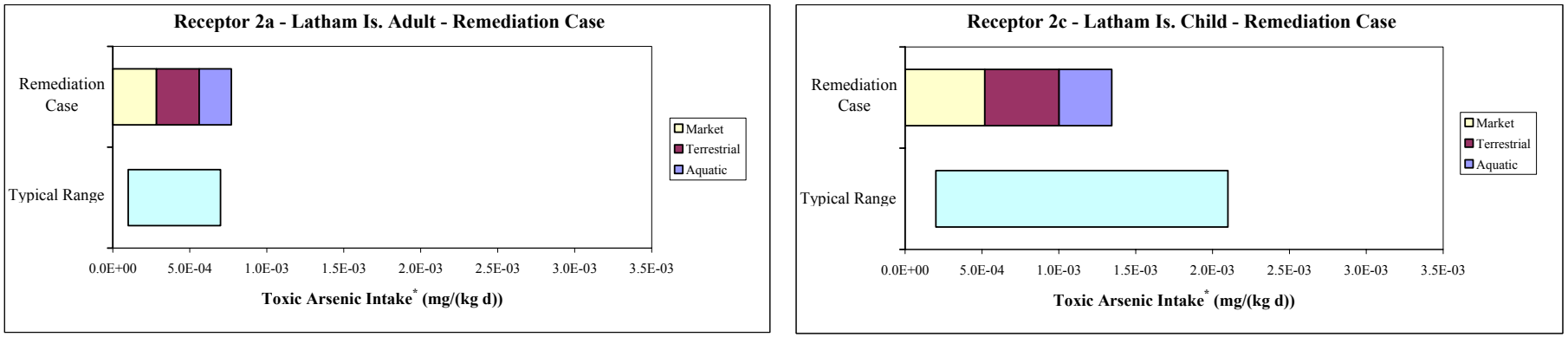
The final sensitivity analysis explored the arsenic exposures experienced by a toddler living at the Giant Mine Townsite under the remediation scenario. The toddler was anticipated to experience higher exposure than either the adult or child receptors, with a mean arsenic exposure of 2.7×10^{-3} mg/(kg d) and a 95th percentile exposure of 4.3×10^{-3} mg/(kg d). The toddler experiences higher exposure than the adult and child due to its higher intake rate to body weight ratio. In other words, the toddler consumes more food for its size than do the adult or child. The main pathway of exposure is market foods which accounts for 64% of the total intake. Soil accounts for 5% of the total intake. It is not known whether toddlers are present currently at the Townsite or will be in the future; however, their exposures would be the highest at this location.

Receptors 2a and 2c – Latham Island Residents

Figure 6.3-2 presents the estimated mean toxic arsenic intakes for Receptors 2a and 2c consuming an average fish diet, based on the results of a survey of Dogrib communities that indicated fish consumption of 84 g/d for a typical adult. The dietary intakes by the Receptor 2 adult and child were based on the Receveur *et al.* (1996) survey, which features a high reliance on country foods and low intake of market foods.

The results plotted on Figure 6.3-2 indicate that the estimated toxic arsenic intake for the adult is marginally higher than the reported range of typical exposures for Canadian adults in the remediation case. The mean total arsenic intake for the child falls within the typical range for Canadian children. For Receptors 2a and 2c, arsenic intake due to market foods contributes about one-third of the overall toxic arsenic intake, while the terrestrial and aquatic pathways each also contribute about one-third of the intake.

FIGURE 6.3-2
ESTIMATED MEAN DAILY INTAKE OF TOXIC ARSENIC BY RECEPTOR 2 (LATHAM ISLAND) ADULT AND CHILD IN 2010



* Includes inorganic + toxic organic arsenic. Note all modelled market foods, 78% of fish and all other aquatic and terrestrial intakes were assumed to be in a toxic form. Typical range estimates are from Environment Canada (1993) and are an estimate of the inorganic arsenic only.

Fish and caribou consumption each account for between 25 to 26% of the intake as indicated on Table 6.3-2. The drinking water pathway accounts for 0.5 to 0.8% of the intake, while soil accounts for 1.1 to 3.1% of exposure.

The results of the sensitivity analyses performed for this receptor are presented in Table 6.3-4. Results show that arsenic exposures are marginally higher when individuals at Latham Island were assumed to consume water from Back Bay rather than from the municipal water supply (7.7×10^{-4} mg/(kg d) vs. 8.3×10^{-4} mg/(kg d)). In contrast, when individuals at Latham Island are assumed to consume fish from Baker Creek rather than from Back Bay, exposures for both the adult and child increase significantly. Results for the third sensitivity analysis where receptors were assumed to be consuming both water from Back Bay and fish from Baker Creek were virtually the same as results for the second sensitivity analysis, further demonstrating the importance of fish source compared to the limited importance of drinking water source on exposure for individuals living on Latham Island.

TABLE 6.3-4
RESULTS OF SENSITIVITY ANALYSIS FOR RECEPTORS 2a AND 2c
(LATHAM ISLAND)

Receptor	Analysis	Description	Total Arsenic Exposure (mg/(kg d))			
			5th	Mean	Median	95th
Latham Island – Adult	Base Case	Municipal Water; Fish from Back Bay	4.0×10^{-4}	7.7×10^{-4}	6.7×10^{-4}	1.5×10^{-3}
Latham Island – Child	Base Case	Municipal Water; Fish from Back Bay	7.0×10^{-4}	1.3×10^{-3}	1.1×10^{-3}	2.5×10^{-3}
Latham Island – Adult	1	Water from Back Bay	4.6×10^{-4}	8.3×10^{-4}	7.2×10^{-4}	1.5×10^{-3}
Latham Island – Child	1	Water from Back Bay	7.6×10^{-4}	1.4×10^{-3}	1.2×10^{-3}	2.6×10^{-3}
Latham Island – Adult	2	Fish from Baker Creek	1.5×10^{-3}	8.5×10^{-3}	5.3×10^{-3}	2.6×10^{-2}
Latham Island – Child	2	Fish from Baker Creek	2.3×10^{-3}	1.4×10^{-2}	8.9×10^{-3}	4.4×10^{-2}
Latham Island – Adult	3	Combination of 1 + 2	1.6×10^{-3}	8.5×10^{-3}	5.3×10^{-3}	2.6×10^{-2}
Latham Island – Child	3	Combination of 1 + 2	2.4×10^{-3}	1.4×10^{-2}	9.0×10^{-3}	4.4×10^{-2}
Latham Island – Adult	4	High Fish Intake	4.4×10^{-4}	9.4×10^{-4}	7.8×10^{-4}	2.0×10^{-3}
Latham Island – Child	4	High Fish Intake	7.7×10^{-4}	1.6×10^{-3}	1.3×10^{-3}	3.3×10^{-3}

An additional sensitivity analysis was conducted for the Latham Island receptor to explore the impact of high fish intake rates on arsenic exposure and risk under the remediation scenario. Results of this analysis are also shown on Table 6.3-4. This sensitivity analysis resulted in arsenic intakes that were higher than those presented in Table 6.3-2. This is not surprising as fish accounts for 25% of exposure. Although the high fish intake rate has a larger impact on exposure than does the source of drinking water, its impact is still considerably less than that of the source of fish (i.e. Baker Creek vs. Back Bay).

The final sensitivity analysis explored the arsenic exposures experienced by a toddler living on Latham Island under the remediation scenario. The toddler was anticipated to experience higher exposure than either the adult or child receptors, with a mean arsenic exposure of 1.8×10^{-3} mg/(kg d) and a 95th percentile exposure of 3.6×10^{-3} mg/(kg d).

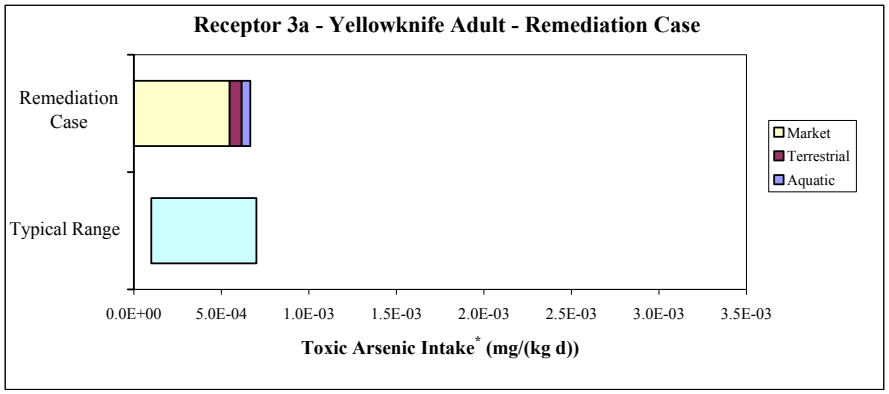
Receptors 3a and 3c – City of Yellowknife Residents

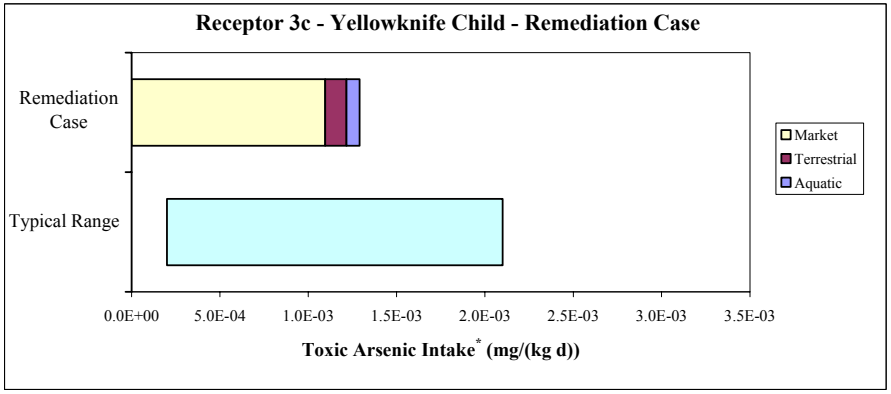
The estimated mean daily toxic arsenic intakes for Receptors 3a and 3c, who live in Yellowknife and have a diet similar to that of the typical Canadian, are presented in Figure 6.3-3. As evidenced in the figure, the contribution of arsenic from market foods dominates the arsenic intake (82% to 85%). The aquatic and terrestrial pathways are relatively minor contributors. The figure shows that the estimated intakes of total arsenic are within the typical range of arsenic exposures for Canadian adults and children.

Next to market foods, the principal pathways of arsenic intake by these adult and the child receptors are ingestion of caribou meat (4 to 5%), vegetation (garden produce and berries: 3 to 4%), fish (5 to 6%), water (0.5 to 0.9%), soil (0.5 to 1.3%), and grouse (0.4 to 0.5%) as summarized in Table 6.3-2. It has been assumed that caribou and grouse were obtained from Baker Creek and that garden produce and berries were obtained from the City of Yellowknife. Soil exposure was assumed to occur in their backyard. Also, Receptors 3a and 3c are assumed to obtain their fish and ducks from North Yellowknife Bay.

The results of the sensitivity analyses performed for this receptor are presented in Table 6.3-5. Results show that arsenic exposures are marginally higher when residents of the City of Yellowknife were assumed to consume berries from the Giant Mine site or water from Back Bay rather than from the municipal water supply. In contrast, when individuals in Yellowknife are assumed to consume fish from Baker Creek rather than from Back Bay, exposures for both the adult and child increase significantly as was seen for the previous receptors. Results for the fourth sensitivity analysis where receptors were assumed to be consuming berries from the Giant Mine site, fish from Baker Creek and water from Back Bay were virtually the same as results for the second sensitivity analysis, further demonstrating the importance of fish source compared to the limited importance of berry and drinking water sources on exposure and risk for residents of the City of Yellowknife.

FIGURE 6.3-3
ESTIMATED MEAN DAILY INTAKE OF TOXIC ARSENIC BY RECEPTOR 3 (CITY OF YELLOWKNIFE) ADULT AND CHILD IN 2010





* Includes inorganic + toxic organic arsenic. Note all modelled market foods, 78% of fish and all other aquatic and terrestrial intakes were assumed to be in a toxic form. Typical range estimates are from Environment Canada (1993) and are an estimate of the inorganic arsenic only.

TABLE 6.3-5
RESULTS OF SENSITIVITY ANALYSIS FOR RECEPTORS 3a AND 3c
(CITY OF YELLOWKNIFE)

Receptor	Analysis	Description	Total Arsenic Exposure (mg/(kg d))			
			5th	Mean	Median	95th
Yellowknife – Adult	Base Case	Berries from Yellowknife; Municipal Water; Fish from Yellowknife Bay	2.3×10^{-4}	6.7×10^{-4}	6.1×10^{-4}	1.3×10^{-3}
Yellowknife – Child	Base Case	Berries from Yellowknife; Municipal Water; Fish from Yellowknife Bay	6.2×10^{-4}	1.3×10^{-3}	1.2×10^{-3}	2.3×10^{-3}
Yellowknife – Adult	1	Berries from Giant Mine	2.7×10^{-4}	7.1×10^{-4}	6.7×10^{-4}	1.3×10^{-3}
Yellowknife – Child	1	Berries from Giant Mine	6.7×10^{-4}	1.4×10^{-3}	1.2×10^{-3}	2.4×10^{-3}
Yellowknife – Adult	2	Fish from Baker Creek	7.9×10^{-4}	4.2×10^{-3}	2.8×10^{-3}	1.3×10^{-2}
Yellowknife – Child	2	Fish from Baker Creek	1.5×10^{-3}	6.8×10^{-3}	4.7×10^{-3}	2.0×10^{-2}
Yellowknife – Adult	3	Water from Back Bay	2.9×10^{-4}	7.2×10^{-4}	6.7×10^{-4}	1.3×10^{-3}
Yellowknife – Child	3	Water from Back Bay	6.8×10^{-4}	1.4×10^{-3}	1.2×10^{-3}	2.4×10^{-3}
Yellowknife – Adult	4	Combination of 1 + 2 + 3	8.8×10^{-4}	4.3×10^{-3}	2.9×10^{-3}	1.3×10^{-2}
Yellowknife – Child	4	Combination of 1 + 2 + 3	1.6×10^{-3}	6.9×10^{-3}	4.8×10^{-3}	2.0×10^{-2}

The final sensitivity analysis explored the arsenic exposures experienced by a toddler living in Yellowknife under the remediation scenario. The toddler was anticipated to experience higher exposure than either the adult or child receptors, with a mean arsenic exposure of 2.0×10^{-3} mg/(kg d) and a 95th percentile exposure of 3.4×10^{-3} mg/(kg d).

Receptors 4a and 4c – Dettah Community Residents

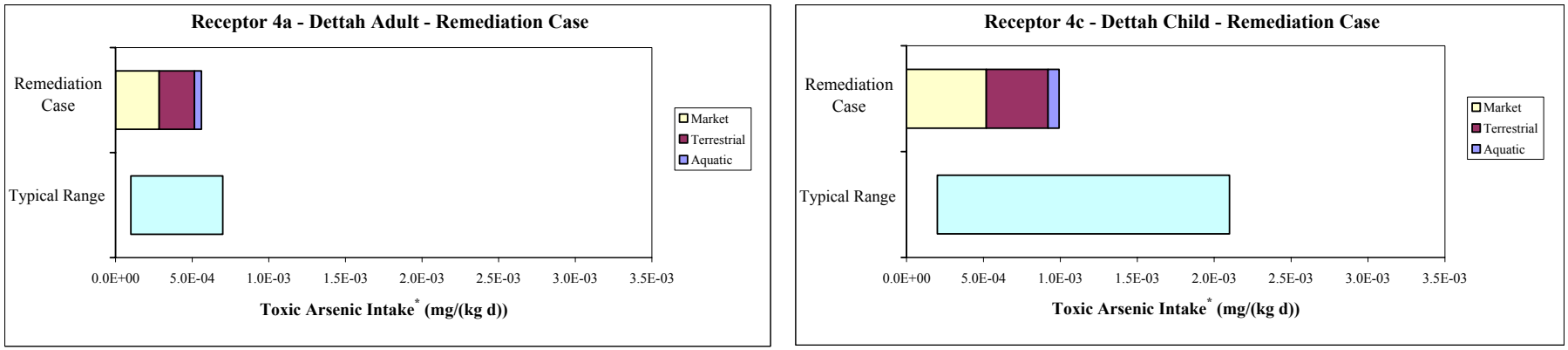
The estimated mean toxic arsenic intakes for Receptors 4a and 4c with an average fish diet are presented on Figure 6.3-4. These Receptors (4a and 4c) were assumed to obtain their drinking water from the municipal supply and their fish and ducks from South Yellowknife Bay. Garden produce, berries and wild game were assumed to be obtained from the area of the Dettah community.

The main pathway of arsenic intake by Receptors 4a and 4c is associated with consumption of market foods, followed by the terrestrial pathways and lastly the aquatic pathways, as demonstrated on Figure 6.3-4. Details of the contributions of all pathways are summarized in Table 6.3-2 for the remediation scenario. A review of these tables indicates that other than market foods, caribou (35%) and fish (7%) are the main contributors to the intakes by Receptors 4a and 4c.

In comparison to the range of arsenic intakes reported for Canadians eating a typical diet, the estimated intake of total arsenic for the adult and child falls within the range for the remediation scenario.

The results of the sensitivity analyses performed for this receptor are presented in Table 6.3-6. Results show that arsenic exposures are essentially the same for individuals in the Dettah Community whether they consume water from South Yellowknife Bay rather than from the municipal water supply or game from the Giant Mine site as opposed to game from the area around Dettah. Results for the third sensitivity analysis where receptors were assumed to consume both water from South Yellowknife Bay and game from Baker Creek were virtually the same as results for either sensitivity analysis. These results emphasize that the source of drinking water or local game is only of marginal importance for arsenic exposures for residents of the Dettah Community.

FIGURE 6.3-4
ESTIMATED MEAN DAILY INTAKE OF TOXIC ARSENIC BY RECEPTOR 4 (DETTAH COMMUNITY) ADULT AND CHILD IN 2010



* Includes inorganic + toxic organic arsenic. Note all modelled market foods, 78% of fish and all other aquatic and terrestrial intakes were assumed to be in a toxic form. Typical range estimates are from Environment Canada (1993) and are an estimate of the inorganic arsenic only.

TABLE 6.3-6
RESULTS OF SENSITIVITY ANALYSES FOR RECEPTORS 4a and 4c
(DETTAH COMMUNITY)

Receptor	Analysis	Description	Total Arsenic Exposure (mg/(kg d))			
			5th	Mean	Median	95th
Dettah – Adult	Base Case	Municipal Water; Game from Dettah	3.3×10^{-4}	5.6×10^{-4}	4.8×10^{-4}	1.1×10^{-3}
Dettah – Child	Base Case	Municipal Water; Game from Dettah	5.7×10^{-4}	1×10^{-3}	8.3×10^{-4}	1.95×10^{-3}
Dettah – Adult	1	Water from South Yellowknife Bay	3.3×10^{-4}	5.7×10^{-4}	4.9×10^{-4}	1.1×10^{-3}
Dettah – Child	1	Water from South Yellowknife Bay	5.7×10^{-4}	1.0×10^{-3}	8.4×10^{-4}	2.0×10^{-3}
Dettah – Adult	2	Game from Giant Mine	3.3×10^{-4}	5.7×10^{-4}	4.9×10^{-4}	1.1×10^{-3}
Dettah – Child	2	Game from Giant Mine	5.8×10^{-4}	1.0×10^{-3}	8.5×10^{-4}	2.0×10^{-3}
Dettah – Adult	3	Combination of 1 + 2	3.4×10^{-4}	5.8×10^{-4}	4.9×10^{-4}	1.1×10^{-3}
Dettah – Child	3	Combination of 1 + 2	5.8×10^{-4}	1.0×10^{-3}	8.6×10^{-4}	2.0×10^{-3}
Dettah – Adult	4	High Fish Intake	3.4×10^{-4}	5.6×10^{-4}	4.9×10^{-4}	9.7×10^{-4}
Dettah – Child	4	High Fish Intake	5.8×10^{-4}	9.8×10^{-4}	8.6×10^{-4}	1.8×10^{-3}

An additional sensitivity analysis was conducted for the Dettah Community receptor to explore the impact of high fish intake rates on arsenic exposure and risk under the remediation scenario. Results of this analysis are presented in Table 6.3-6. As seen on the table, the total arsenic intake estimates are similar to the other sensitivity cases. Overall, fish intake rates are of limited importance for arsenic exposure for residents of the Dettah Community.

The final sensitivity analysis explored the arsenic exposures experienced by a toddler living in Dettah under the remediation scenario. The toddler was anticipated to experience higher exposure than either the adult or child receptors, with a mean arsenic exposure of 1.3×10^{-3} mg/(kg d) and a 95th percentile exposure of 2.6×10^{-3} mg/(kg d).

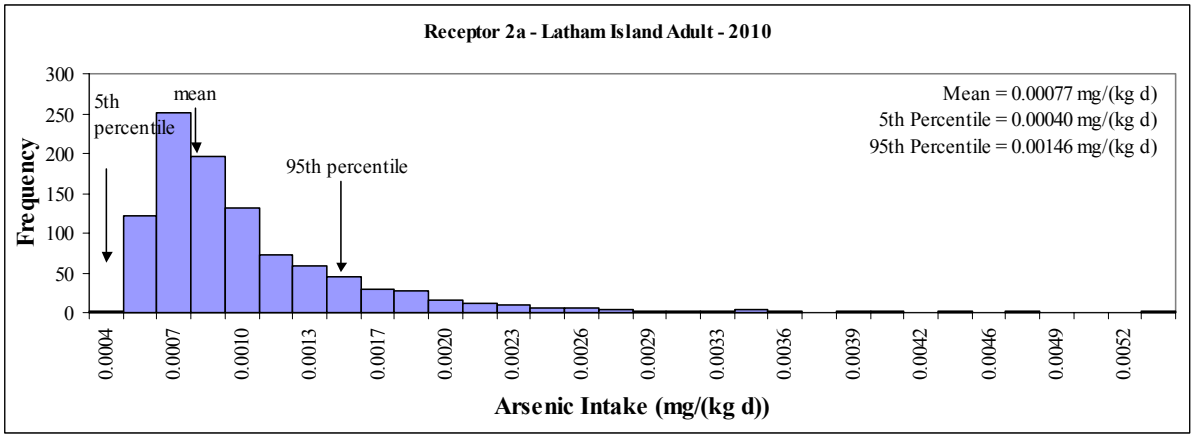
Uncertainty in Intake Estimates

Figure 6.3-5 presents a histogram plot of the output distributions of toxic arsenic intakes for the adult resident of Latham Island (Receptor 2a), generated from 1000 probabilistic simulation trials for the remediation scenario. Receptors 2a was selected as this receptor was predicted to

receive the second highest arsenic intake resulting from the Giant Mine activities of all the adults and children considered in the assessment. Receptor 1a, while predicted to incur the highest arsenic intake, only receives a small portion of the arsenic intake from local sources influenced by the Giant Mine site.

As seen from inspection of Figure 6.3-5, the histogram has a pronounced tail. This is attributable to the importance of certain exposure pathways contributing more to the intake (i.e. fish, market foods and caribou). The skewed distribution of the arsenic intake shows the influence of consuming larger amounts of caribou, fish and other traditional foods, in which the arsenic contents are assumed to have lognormal distribution characteristics compared to other receptors that consume larger amount of market foods, which are assumed to display more triangular, bell-shaped distributions.

FIGURE 6.3-5
FREQUENCY HISTOGRAM OF RECEPTOR 2a TOTAL ARSENIC INTAKES FOR REMEDIATION SCENARIO



6.3.5 Significance of Toxic Arsenic Exposure

The assessment of the daily intake of total (inorganic and toxic organic) arsenic by the four different hypothetical adult receptors showed that Receptors 1a and 2a result in the highest exposures of arsenic. Similarly, child receptors 1c and 2c received the highest exposures amongst the child receptors. For Receptor 1a and c, the estimated mean total daily intakes of arsenic, inclusive of market foods, were 0.0009 mg/(kg d) for the adult and 0.0016 mg/(kg d) for the child. While the intakes are above the range of typical intakes for Canadians living in communities that do not have local arsenic issues, they are similar to or below exposure levels in communities with elevated arsenic levels in the local environment. Market foods account for 40% to 82% of the arsenic intakes.

Figure 6.3-6 provides a schematic representation of the mean arsenic intake for the most exposed adult receptors (Receptors 1a and 2a) and provides a comparison to estimated exposure levels in communities with high arsenic levels, such as Wawa, Ontario (O'Connor Associates 2000), Deloro, Ontario (CANTOX 1999) and a community in Newfoundland (Environmental and Occupational Health Plus and SENES 2002). As seen from the figure, the predicted arsenic intakes for the Yellowknife area receptors are below those calculated for these communities.

6.3.6 Carcinogenic Risk of Toxic Arsenic Exposure

Toxic arsenic is known to have carcinogenic effects, as discussed in Section 5.2. Carcinogenic risk is expressed as the incremental incidence of developing cancer for a lifetime of exposure.

For this assessment, a composite person was used to capture the exposure over a lifetime (70 years of exposure) spanning a person's childhood and adult years. Equation (6-1) shows the method of calculating the carcinogenic risks.

$$Risk = Intake \times SF_o \qquad (6\text{-}1)$$

where:

Intake = weighted average intake {mg/(kg d)} over the person lifetime (obtained from pathways model)

SF_o = Slope Factor – oral pathway {1.2 $(mg/(kg\ d))^{-1}$} (see Section 5.2)

The estimate of a lifetime of exposure included consideration of the toddler, child and adult life stages over a total 70-year lifetime.

Table 6.3-7 shows the lifetime risk levels for increased cancer incidence calculated for the remediation scenario for the composite receptors at each of the receptor locations. The risk estimates summarized in the table are inclusive of all major oral exposure pathways including ingestion of water, food and contaminated soil and includes background exposure. The table also presents the risks associated with super market foods. A third column presents the risks from exposure with study area alone. These risks include exposure from background, as well as to incremental exposure from loadings at the Giant Mine site. From this assessment, it can be seen that the risk estimate associated with background plus the Giant Mine site inputs is highest for Receptor 2 at 6.1 in 10,000 for the remediation case and lowest for Receptor 3 at 1.6 in 10,000. The incremental risks for Receptor 1 (Giant Mine Townsite) and Receptor 4 (Dettah Community) are essentially the same, demonstrating that background levels in the area account for a substantial portion of the exposure.

FIGURE 6.3-6
COMPARISON OF ARSENIC INTAKES FROM STUDY AREA WITH THOSE OF OTHER COMMUNITIES WITH HIGH ARSENIC EXPOSURES

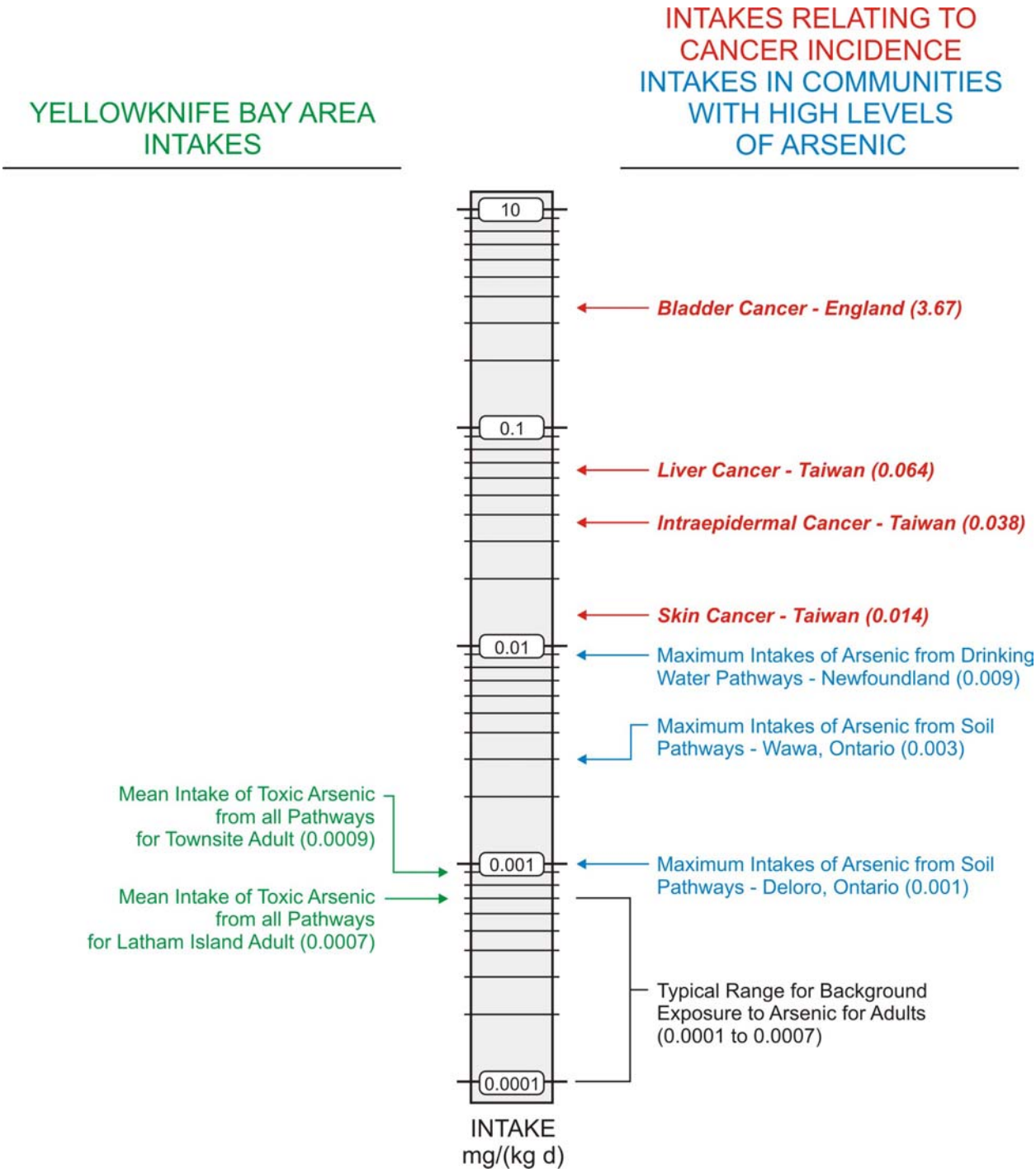


TABLE 6.3-7
ESTIMATED MEAN LIFETIME CARCINOGENIC RISK FOR COMPOSITE ADULT

Receptor[a]	Receptor Name	Remediation Case	Supermarket Foods		Other Arsenic Sources	
			Consumption of Supermarket Foods	% of Risk	Incremental Risk without Supermarket Foods	% of Risk
1	Giant Mine Townsite	1.2×10^{-3}	7.8×10^{-4}	65	4.2×10^{-4}	35
2	Latham Island	1.0×10^{-3}	3.9×10^{-4}	39	6.1×10^{-4}	61
3	City of Yellowknife Resident	9.2×10^{-4}	7.6×10^{-4}	82	1.6×10^{-4}	18
4	Dettah Community Resident	7.6×10^{-4}	3.9×10^{-4}	51	3.7×10^{-4}	49

Note: a - Composite individual encompassing 11 years as a child and 59 years as an adult.
Incremental risk without supermarket foods also includes background exposure.

6.3.7 Implications of the Assessment

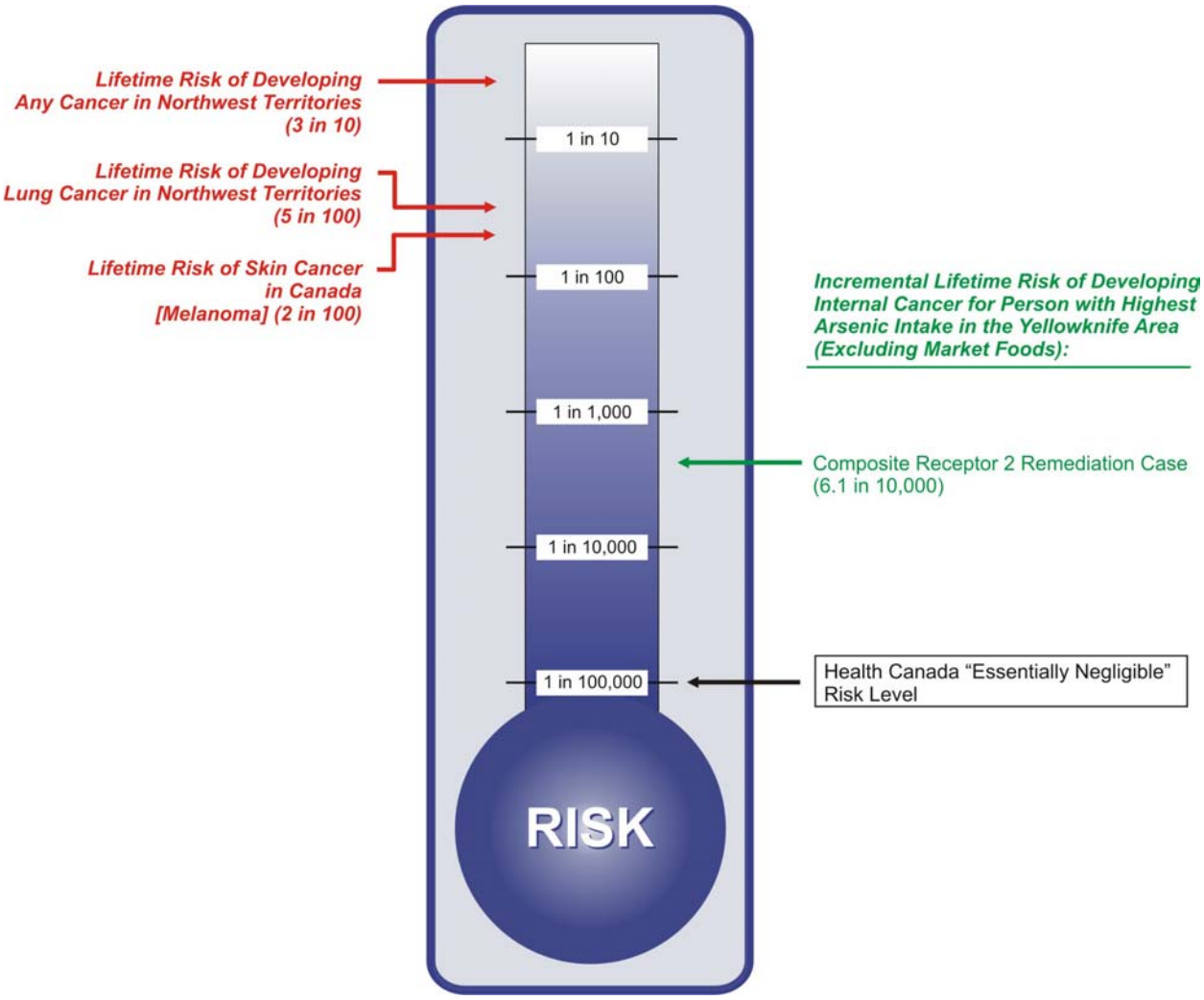
To provide a perspective on the risks levels provided in Table 6.3-7, risk levels can be calculated for the range of typical intakes for the Canadian population reported in Section 6.3. Thus, the risk level for a composite receptor, based on the ranges of typical intakes for a child and adult shown in Figures 6.3-1 to 6.3-4, ranges from 1.4 in 10,000 to 1.1 in 1,000. The risk levels associated with the remediated Giant Mine area as discussed in Section 6.3.6 (1.6 in 10,000 to 6.1 in 10,000) are within the risk levels associated with typical arsenic exposure across Canada.

In addition, studies conducted in communities with high arsenic levels employing either cross-section evaluations of the entire community (Chapels Cove, Newfoundland; arsenic in drinking water) or population samples (Wawa, Ontario and Deloro, Ontario; arsenic in soil) have failed to show elevated levels of skin cancer or other adverse effects associated with arsenic exposure. Although cross-sectional evaluations can only constitute an estimate of prevalence of skin cancer in these communities, which does not allow attribution of causation to any risk factor like arsenic, the prevalence can provide an indication for comparison with other communities to reveal unusual patterns of disease. The findings from the communities are not unexpected since the intakes are below intakes associated with elevated risks of skin cancer or other types of cancer (liver and bladder) that are related to arsenic exposure (see Figure 6.3-6).

Figure 6.3-7 provides a comparison of the predicted incremental risks for lifetime exposure to arsenic for Receptor 2 to other Canadian cancer statistics. It should be noted that the incremental lifetime risks include background exposure to arsenic in various media such as soils, water and traditional foods but does not include exposure to arsenic present in market foods. As seen in the

figure, the predicted cancer risks are below the lifetime incidence cancer rate of 3 in 10 for the Northwest Territories population (Canadian Cancer Statistics 2003) as well as being below the risks of developing lung cancer (5 in 100) or developing skin cancer in the Canadian population (2 in 100). These results suggest that the risk of developing cancer from total arsenic exposure would be 20-300 times lower than the overall cancer risk. While the incremental lifetime risk levels are above the Health Canada acceptable level of 1 in 100,000, the development of lung cancer from exposure to arsenic present on or released from the Giant Mine site will not be distinguishable in the Yellowknife population from other causes of cancer.

FIGURE 6.3-7
COMPARISON OF LIFETIME RISK OF ARSENIC EXPOSURE TO LIFETIME RISK OF CANCER



6.4 UNCERTAINTIES INVOLVED

The primary uncertainties in the risk assessment are associated with estimates of arsenic loadings and exposure concentrations as well as the toxicity data used to define the arsenic toxicity reference values for the ecological and human receptors.

Uncertainty in the toxicity data was not accounted for in this assessment. In principle, it should be possible to construct statistical distributions of toxicity data (e.g. the LC_{25}) and use these distributions in repeated estimates of the screening index values or hazard quotients. However, limitations in available toxicity data preclude incorporation of their uncertainty in the overall analyses.

There is uncertainty associated with the TRV's selected for this assessment. There is also uncertainty associated with the slope factor for evaluation of potential carcinogenic effects. A range of potential slope factors is discussed in Section 5.2 and a value of 1.2 $(mg/(kg\ d))^{-1}$ was chosen for the assessment. This resulted in a risk of 1.2 in 1,000 for the Giant Mine Townsite receptor (Table 6.3-7). With the use of a slope factor of 2.8 $(mg/(kg\ d))^{-1}$ as recommended by the FCSAP program, the potential carcinogenic risk is calculated as 2.6 in 1,000. As seen in Table 6.3-7, this risk includes background exposure and is still well below the overall cancer risk for the Canadian population.

There is uncertainty also in the form of the organic arsenic compounds in fish and whether the organic arsenic compounds are toxic. Similarly, the consideration that caribou-related arsenic is entirely inorganic is cautious and may overestimate the exposure via this source. Additionally, the estimates of the exposure commitment from market food sources is also subject to a large uncertainty. For example, the assumption that all of the arsenic was in the inorganic form leads to an overestimate of exposure. Section 7.5 and Table 7.5-1 provides a detailed discussion on the uncertainties in the assessment. Even given these uncertainties, it is our opinion that, the exposure estimates derived from this assessment are reasonable estimates of the risk presented by the arsenic in the local environment of the Yellowknife study area through the various exposure media.

The use of the 17% bioaccessibility factor for ingestion of soils is also an area of uncertainty. If the bioaccessibility factor is increased to 73% as indicated from some literature sources, the arsenic intakes to the various human receptors increase marginally as seen in the following table (Table 6.4-1). However, the overall conclusions of the assessment do not change.

TABLE 6.4-1
DOSE IMPLICATION OF INCREASE IN BIOACCESSIBILITY

Remediation Case

Receptor	With Bioaccessiblity Set to 17% Total (mg/kg/d)				With Bioaccessiblity Set to 73.1% Total (mg/kg/d)			
	5th	**Mean**	**Median**	**95th**	**5th**	**Mean**	**Median**	**95th**
1a.Townsite - adult	$3.7x10^{-4}$	$8.8x10^{-4}$	$8.1x10^{-4}$	$1.6x10^{-3}$	$4.2x10^{-4}$	$9.4x10^{-4}$	$8.5x10^{-4}$	$1.7x10^{-3}$
1c.Townsite - child	$8.4x10^{-4}$	$1.6x10^{-3}$	$1.5x10^{-3}$	$2.7x10^{-3}$	$1.1x10^{-3}$	$1.9x10^{-3}$	$1.8x10^{-3}$	$3.1x10^{-3}$
2a.Latham Is. - adult	$4.0x10^{-4}$	$7.7x10^{-4}$	$6.7x10^{-4}$	$1.5x10^{-3}$	$4.8x10^{-4}$	$9.8x10^{-4}$	$8.3x10^{-4}$	$2.1x10^{-3}$
2c.Latham Is. - child	$7.0x10^{-4}$	$1.3x10^{-3}$	$1.1x10^{-3}$	$2.5x10^{-3}$	$8.8x10^{-4}$	$1.7x10^{-3}$	$1.5x10^{-3}$	$3.4x10^{-3}$
3a.Yellowknife - adult	$2.3x10^{-4}$	$6.7x10^{-4}$	$6.1x10^{-4}$	$1.3x10^{-3}$	$2.7x10^{-4}$	$7.0x10^{-4}$	$6.4x10^{-4}$	$1.4x10^{-3}$
3c.Yellowknife - child	$6.2x10^{-4}$	$1.3x10^{-3}$	$1.2x10^{-3}$	$2.3x10^{-3}$	$6.5x10^{-4}$	$1.4x10^{-3}$	$1.3x10^{-3}$	$2.4x10^{-3}$
4a.Dettah - adult	$3.3x10^{-4}$	$5.6x10^{-4}$	$4.8x10^{-4}$	$1.1x10^{-3}$	$3.5x10^{-4}$	$6.8x10^{-4}$	$5.6x10^{-4}$	$1.4x10^{-3}$
4c.Dettah - child	$5.7x10^{-4}$	$1.0x10^{-3}$	$8.3x10^{-4}$	$2.0x10^{-3}$	$6.4x10^{-4}$	$1.2x10^{-3}$	$9.9x10^{-4}$	$2.3x10^{-3}$

7.0 SUMMARY AND CONCLUSIONS

A Tier 2 risk assessment was carried out for the Giant Mine site based on current plans for remediation of contaminated surface areas and management of arsenic trioxide dust stored in vaults in the underground mine workings. Details of the proposed remediation plan are presented in a separate report. In brief, the remediation work will involve: removal of surface structures; decontamination of areas with elevated arsenic and hydrocarbon levels; infilling of some of the surface pits; rerouting of parts of Baker Creek; recontouring and covering of the tailings management areas; freezing of the arsenic trioxide dust vaults; and construction of a new mine water treatment plant with effluent discharge to Back Bay.

To assess the resulting risks following completion of the remediation activities, a remediation scenario was defined. The arsenic load to Baker Creek from all sources is estimated to total 480 kg/y, while an additional arsenic load of 210 kg/y is estimated to enter Back Bay directly. The latter load includes 140 kg/y from the mine water treatment plant, which was assumed to be discharged directly to Back Bay, and 70 kg/y of direct mine site runoff to Back Bay. A separate assessment was undertaken for a "walk-away" or "do-nothing" scenario. The arsenic loads associated with this scenario are substantially higher than the remediation scenario evaluated and the results of the "do-nothing" assessment are presented in Appendix F. The following discussion pertains only to the proposed remediation scenario.

The assessment considered risks to both ecological species and people from exposure to arsenic via several pathways. The exposure analysis was undertaken using a lake dispersion and pathways model that permitted the many complexities of the assessment to be carried out in a rigorous manner. Measured arsenic levels in environmental media were used in the exposure assessment when such data were available. If measured levels were not available, predictive modelling was used, particularly to evaluate the effects of changes in arsenic loads to the aquatic environment. The basic steps involved in the assessment process included:

- characterization of arsenic releases to the environment;
- assessment of arsenic uptake by aquatic plants, benthic invertebrates and fish;
- characterization of arsenic levels in soils and sediments in the study area;
- characterization of arsenic levels in the atmospheric environment;
- characterization of arsenic levels in terrestrial vegetation and animals;
- selection of representative ecological species from all trophic levels;
- selection of human receptors with a representative range of diets and exposure pathways;
- identification of exposure pathways for ecological species and human receptors;
- modelling of arsenic dispersion in Back Bay and Yellowknife Bay, including prediction of water column and sediment concentrations;

- pathways analysis of arsenic levels in the environment and transfer along the food chain (e.g. uptake by aquatic plants and transfer to animals and birds that consume these plants);
- identification of toxicity reference values for ecological species and humans; and
- characterization of the risks from total arsenic intake via all major exposure pathways.

To account for uncertainties in many of the factors considered in the assessment, the pathways analysis was performed probabilistically. Each simulation was performed 1,000 times to produce a distribution of model output values. The results of these simulations were then analyzed statistically to estimate expected mean values, minimum (5^{th} percentile) values and maximum (95^{th} percentile) values.

The ecological assessment considered aquatic receptors from various trophic levels (e.g. phytoplankton, zooplankton, benthic invertebrates, predator fish and bottom-feeder fish). The aquatic ecosystem assessment considered exposure to arsenic present in the Baker Creek drainage, as well as in Back Bay and Yellowknife Bay. Ecological impacts were characterized by the value of a screening index (generally considered to be 1), which is calculated as the ratio of the predicted arsenic concentration in water to the toxicity reference value for the aquatic species under consideration.

In addition, detailed pathways modelling was conducted for several terrestrial receptors to estimate their potential exposure to arsenic present in these water bodies and the study area soils and terrestrial vegetation. The terrestrial receptors identified for inclusion in the assessment included three duck species (mallard, merganser and scaup), hare, spruce grouse, moose, mink, muskrat, bear and wolf. Caribou were also included in the assessment, as they comprise a significant portion of the local diet and are known to winter in the study area. Ecological impacts were determined through a comparison to the Lowest Observable Adverse Effects Level (LOAEL) toxicity reference value (TRV) obtained from literature data.

An assessment of the potential implications to human health from arsenic exposure for the remediation scenario was also considered for human receptors (an adult and child) living at four different locations, and with differing lifestyle characteristics. The pathways modelling was conducted to provide estimates of the potential exposures of both adults and children living in the study area to arsenic intakes from drinking water, ingestion of soil, eating vegetables and berries and eating fish and wild game, which were assumed to be exposed to arsenic present in the study area. A range of dietary characteristics was assumed for the human receptors to provide a range of exposure estimates. Results from a dietary survey of Dene living in the Yellowknife area, as well as from a regional survey were used to determine dietary characteristics for the receptors. Other exposure information, such as drinking water consumption was obtained from data on the general Canadian population. Various sensitivity

analyses were performed to investigate the impact of varying the geographical source of the exposure pathways, the impact of upper bound estimates of fish intake rates, and the impact of arsenic exposure on toddlers.

In human populations, the most sensitive endpoint of arsenic exposure is considered to be cancer. Therefore, in this study, lifetime carcinogenic risk was estimated.

7.1 WATER QUALITY PREDICTIONS

Water quality predictions for Baker Creek, Back Bay and Yellowknife Bay are summarized in Table 7.1-1. Predictions are compared to the CCME water quality guidelines for the protection of aquatic life and for drinking water.

TABLE 7.1-1
COMPARISON OF WATER QUALITY PREDICTIONS TO ARSENIC GUIDELINES

	CCME Arsenic Guideline for Protection of Freshwater Aquatic Life (5 µg/L)				Proposed Canadian Arsenic Guideline for Drinking Water (5 µg/L)			
	Baker Creek	Back Bay	North YK Bay	South YK Bay	Baker Creek	Back Bay	North YK Bay	South YK Bay
Remediation Case	x	✔	✔	✔	x	✔	✔	✔

Notes: x - Indicates exceedance of the appropriate guideline for both the predicted mean and 95th percentile values.
✔ - Indicates that predicted arsenic concentration in water is below the appropriate guideline.

7.2 SEDIMENT QUALITY PREDICTIONS

Arsenic levels in sediments in Baker Creek, Back Bay and Yellowknife Bay have been shown to be elevated above background due to historic activities. Sediment quality predictions showed decreasing trends with time in all water bodies for the remediation case. However, arsenic levels in sediment remain elevated. Only in South Yellowknife Bay is the sediment arsenic concentration predicted to approach background conditions by 2100.

7.3 ECOLOGICAL ASSESSMENT

A summary of the results of the ecological assessment for all aquatic receptors and scenarios is shown in Table 7.3-1. The water quality predictions (see above) indicated that the arsenic level in Baker Creek is expected to consistently exceed the CCME guideline for the protection of aquatic life. Thus, a more detailed examination of the impacts on the aquatic environment was undertaken, with consideration of the specific receptors expected to be found in the local area. The results of this assessment showed that there is the potential at the 95th percentile that the predicted arsenic levels could affect bottom-feeding fish in Baker Creek when a comparison is

made to EC_{25} toxicity reference values. No adverse effects to aquatic species are anticipated in Back Bay or Yellowknife Bay under the remediation case. A sensitivity analysis was carried out using a lower toxicity reference value (EC_{10}); the comparison indicated that there would be no adverse effects in aquatic species in Back Bay or Yellowknife Bay. However, potential adverse effects are predicted in predatory and bottom feeder fish in Baker Creek. Biological surveys in 2002 indicate the presence of predatory and bottom feeder fish in Baker Creek both upstream and downstream of the mine workings indicating that the TRVs used in this assessment may be conservative. More biological sampling is needed to provide a weight of evidence that adverse effects are not occurring in aquatic fish populations in Baker Creek.

TABLE 7.3-1
RESULTS OF ECOLOGICAL ASSESSMENT FOR AQUATIC RECEPTORS USING EC_{25} TOXICITY REFERENCE VALUES

	Aquatic Plants				Benthic Invertebrates			
	Baker Creek	**Back Bay**	**North YB**	**South YB**	**Baker Creek**	**Back Bay**	**North YB**	**South YB**
Remediation Case	✓	✓	✓	✓	✓	✓	✓	✓
	Predatory Fish				**Bottom Feeder Fish**			
	Baker Creek	**Back Bay**	**North YB**	**South YB**	**Baker Creek**	**Back Bay**	**North YB**	**South YB**
Remediation Case	✓	✓	✓	✓	x	✓	✓	✓

Note:

x - Indicates exceedance of toxicity reference value for aquatic species at the predicted 95th percentile values.

✓ - indicates that negative effects are not expected.

YB - Yellowknife Bay.

A summary of the results of the ecological assessment for terrestrial animals is provided in Table 7.3-2 and for aquatic birds (ducks) in Table 7.3-3. There were a number of uncertain components in the terrestrial risk assessment. Cautious assumptions were adopted whenever the uncertainties could not be resolved. For example, for the terrestrial receptors, it was assumed that while in the study area, they spend all their time on the Giant Mine site and/or in the lower reach of Baker Creek (i.e., in the areas with the highest arsenic content). It was also assumed that the species obtain 100% of their food and water from the Baker Creek watershed (all species) or Back Bay (ducks only) while in the study area (e.g.,10% of the year for caribou, 50% of the year for the duck species and 100% of a year for hare). Additionally, the arsenic present in these media was assumed to be entirely transferred into the species. While the species were assumed to also consume either soil or sediment, only 17% of the arsenic in these media was assessed as being biologically available based on data from extraction test work. The estimated daily intake of arsenic by terrestrial species was compared to the Lowest Observable Adverse

Effects Level (LOAEL). The LOAEL is the lowest concentration where an effect can be seen in laboratory testing.

TABLE 7.3-2
RESULTS OF ECOLOGICAL ASSESSMENT FOR TERRESTRIAL RECEPTORS

	Bear	Caribou	Hare	Grouse	Moose	Mink	Muskrat	Wolf
Remediation Case	✓	✓	x	✓	✓	x	x	✓

Notes:
x - indicates exceedance of appropriate toxicity reference value for the predicted mean and 95th percentile values.
✓ - indicates that negative impacts are not expected.

TABLE 7.3-3
RESULTS OF ECOLOGICAL ASSESSMENT FOR DUCKS

	Mallard		Merganser		Scaup	
	Baker Creek	Back Bay	Baker Creek	Back Bay	Baker Creek	Back Bay
Remediation Case	✓	✓	✓	✓	✓	✓

Notes:
x - indicates exceedance of appropriate toxicity reference value for the predicted mean and 95th percentile values.
✓ - indicates that negative impacts are not expected.

The results of the risk assessment showed that, with two exceptions, the estimated arsenic intakes for terrestrial species were below toxicity reference values in the remediation scenario. Estimated arsenic intakes from all sources for bear, caribou, grouse and wolf were predicted to be well below toxicity reference values for these species (Table 7.3-2). Likewise, the arsenic intakes predicted for the three duck species were well below the toxicity reference values (Table 7.3-3).

The first exception was that of hare in the vicinity of Baker Creek, where toxicity reference values were predicted to be exceeded at the mean and 95th percentile levels. The major source of arsenic for hare is terrestrial vegetation. Measured arsenic levels, representative of current conditions, were used in the assessment. While the remediation plan provides for removal of contaminated soils with arsenic content of greater than 340 mg/kg, the cautious assumption was made that arsenic levels in terrestrial vegetation would not change.

The second exception is that of terrestrial species living in the aquatic environment (mink and muskrat) on Baker Creek, where toxicity reference values were predicted to be exceeded at both the mean and 95th percentile levels. The major sources of arsenic intake by mink and muskrat are related to the elevated levels of arsenic in the creek water, creek sediments and aquatic plants. Arsenic levels in the sediments in Baker Creek are elevated due to historical discharges. Arsenic levels in surface runoff in Baker Creek upstream and downstream of the mine site are also elevated due to historic releases to the atmospheric and aquatic environments from the mine. Post-remediation arsenic loadings to Baker Creek from the Giant Mine site will be significantly reduced, but upstream inputs will continue.

To address the question of possible effects on these terrestrial species, a field investigation was undertaken on Baker Creek to establish current conditions. The biological studies that were carried out specifically targeted the muskrat population on Baker Creek. Twelve active burrows were discovered (downstream of the mine workings) that supported an estimated population of between 66 and 197 animals (Jacques Whitford 2003). Subsequently to this study a number of muskrat were trapped both upstream and downstream of the mine workings and arsenic analyses were carried out on the muscle, organs (liver and kidney) and tail of the muskrats. The muskrat tissue data indicated that the mine workings had an effect on the concentration of arsenic measured in the muskrats. Muskrats collected from downstream locations had on average approximately two times higher concentrations than the ones collected upstream. The analyses also showed that the kidney had the highest concentration of arsenic and the muscle had the lowest arsenic concentration. However, the concentrations found in the organs were below levels that have been reported to be toxic in other animals. Based on the toxicity studies and field evidence from the biological survey which indicates that there are active dens that support a substantial population of muskrat on Baker Creek, it is unlikely that the presence of arsenic in the sediments of Baker Creek is causing serious adverse effects on populations of small terrestrial mammals that have a significant aquatic based diet.

7.4 HUMAN HEALTH ASSESSMENT

A detailed quantitative risk assessment was conducted that examined the exposure to arsenic through the aquatic and terrestrial pathways, with consideration of both direct exposure (e.g. drinking water) and indirect (e.g. consumption of fish, berries, etc.) pathways of exposure. Representative receptors were identified at several locations in the Yellowknife study area. Dietary survey data reported for the study area were used to capture the consumption patterns of the local populations. To encompass the differences in exposure patterns in a lifetime, both an adult and a child (aged 5 to 11 years) were considered at most locations. In addition, the source of drinking water and food items was varied amongst the receptors as indicated on Table 7.4-1 to take into account differences in arsenic levels in these dietary components in the study area.

The results show that Receptors 1 (at Giant Mine Townsite) were predicted to have the highest arsenic intakes, followed by Receptor 2 (on Latham Island). The main contributor to the exposure was market foods; however, the arsenic intakes by these receptors are also a direct reflection of their assumed dietary characteristics. For example, Receptor 2 was assumed to obtain fish and ducks from Back Bay, small and large game from the Baker Creek watershed and garden produce and berries from Latham Island. This receptor was assumed to rely heavily on country foods. In comparison, Receptor 1 was assigned a much higher market food diet but to also obtain small and large game from the Baker Creek watershed.

The estimated total mean toxic arsenic intakes for the eight receptors (four adults and four children) and the incremental lifetime cancer risks for composite person (11 years as a child and 59 years as an adult) are summarized on the bottom portion of Table 7.4.-1. It is noted that the incremental lifetime risks include background from various media such as water, soil, traditional food; however, it does not include risks associated with consumption of supermarket foods. From the table it can be seen that the predicted arsenic intakes are:

- within the range of risk levels associated with typical exposure of the general Canadian population living in communities across Canada;
- well below incidence rates of cancer in the Northwest Territories; and,
- similar to exposure levels in communities with elevated arsenic levels in the local environment. Communities such as Deloro and Wawa in Ontario and other communities in Newfoundland with similar high levels of arsenic do not report elevated prevalence of cancers related to arsenic exposure.

In summary, given that the cancer risk predictions are intended to over-estimate cancer risk, measurable elevation of cancer prevalence would not be expected.

TABLE 7.4-1
RESULTS OF HUMAN HEALTH ASSESSMENT

Diet	Receptor 1a and c Giant Mine Townsite		Receptor 2a and c Latham Island		Receptor 3a and c City of Yellowknife		Receptor 4a and c Dettah Community	
Dietary Component								
Drinking Water	Municipal Supply		Municipal Supply		Municipal Supply		Municipal Supply	
Soil	Giant Townsite		Latham Island		City of Yellowknife		Dettah Community	
Garden Produce	Giant Townsite		Latham Island		City of Yellowknife		Dettah Community	
Berries	Giant Mine Site		Latham Island		City of Yellowknife		Dettah Community	
Large Game	Baker Creek		Baker Creek		Baker Creek		Dettah Community	
Small Game	Baker Creek		Baker Creek		Baker Creek		Dettah Community	
Ducks	Baker Creek/Back Bay		Back Bay		North Yellowknife Bay		South Yellowknife Bay	
Fish	Back Bay		Back Bay		North Yellowknife Bay		South Yellowknife Bay	
Medicinal Teas	-		Giant Mine Site		-		Dettah Community	
Supermarket Foods	Imported		Imported		Imported		Imported	
Estimated Mean Toxic Arsenic Intakes (mg/(kg d))								
	Adult	Child	Adult	Child	Adult	Child	Adult	Child
Remediation Case	<u>0.00088</u>	0.0016	<u>0.00077</u>	0.0013	0.00067	0.0013	0.00056	0.001
Estimated Mean Incremental Lifetime Carcinogenic Risk Including Background (Excluding Market Foods)								
	Composite Person		**Composite Person**		**Composite Person**		**Composite Person**	
Remediation Case	4.2 in 10,000		6.1 in 10,000		1.6 in 10,000		3.7 in 10,000	

<u>Notes</u>:

Mean arsenic intakes included contributions from toxic arsenic forms and from market foods.

<u>Underline</u> indicates that estimated mean intake exceeds the typical range of intakes for the general Canadian population, i.e. 0.0001 to 0.0007 mg/(kg d) for adults and 0.0002 to 0.0021 mg/(kg d) for children aged 5 to 11 years old.

Composite person encompasses 11 years as a child and 59 years as an adult

Sensitivity analyses were performed to explore the importance of the geographical source of drinking water and food items, the impact that upper bound fish intake rates would have on exposure, and the exposures that would be experienced by the toddler sensitive life stage. The geographical source of fish was shown to have the largest impact on arsenic intake with fish taken from Baker Creek having a significant potential effect on arsenic exposure and risk. Exposure assessment employing the upper bound fish intake rate had little impact on overall arsenic intake and risk.

Analyses using the toddler indicated higher exposure due to the higher intake to body weight ratio. Soil intake by a toddler represented only 5% of the total estimated arsenic intake. The majority of the arsenic intake for a toddler came from ingestion of supermarket food, caribou, fish, backyard produce, and berries.

7.5 SUMMARY OF RISK ASSESSMENT AND UNCERTAINTIES

In summary, the results of the ecological and human health risk assessments of the proposed remediation plan for the Giant Mine site show that:

- Aquatic plants and fish in Back Bay and Yellowknife Bay are not at risk of adverse effects for arsenic releases as high as about 690 kg/year to Back Bay. In Baker Creek, the predicted arsenic release of 480 kg/year under the remediation case is expected to remain above the CCME water quality guideline of 5 µg/L for protection of aquatic life and may result in potential adverse effects in fish in Baker Creek. Additional clean-up of Baker Creek sediments would reduce these risks.
- Further studies of Baker Creek sediments are underway and should provide stronger evidence as to whether historical arsenic contamination in Baker Creek sediment has caused a reduction in benthic community diversity. Recovery of this system is expected to take a long time. The diversity of benthic communities in parts of Back Bay and North Yellowknife Bay may also be affected by existing levels but this situation will gradually improve as sediments with elevated arsenic levels are buried over time and covered with cleaner material.
- The assessment indicates that some small terrestrial animals (e.g. mink and muskrat) in the Baker Creek watershed are potentially at risk under the proposed remediation scenario. The primary sources of arsenic intake by these species are related to the elevated arsenic levels in Baker Creek sediments, aquatic plants and surface drainage, all of which are linked to historical contamination in the watershed. Field investigations carried out in 2003 and 2004 however, show that there is a good population of muskrat in the downstream reach of the creek. While the muskrat at downstream locations were found to contain higher levels of arsenic in muscle and body organs than muskrat from upstream locations, the tissue arsenic levels were below levels found to have toxic effects

in other animals. Taken as a whole, these results suggest that the arsenic present in Baker Creek does not appear to have an adverse effect on the muskrat population.

- People living in the study area are unlikely to be at risk of adverse effects from arsenic exposure, even though arsenic levels in the area are higher than found in other communities in Canada. The estimated total intakes for Yellowknife area residents generally fall within the range of typical arsenic intakes for other Canadians. Estimated cancer risks arising from Giant Mine arsenic are well below the risks associated with other causes of cancer.

The uncertainties discussed in each of the above sections need to be taken into consideration when these results are put to use. Table 7.5-1 summarizes the major assumptions adopted for the ecological and human health risk assessments. Each assumption was reviewed to determine whether it was likely to lead to under-estimation or over-estimation of risks. The resulting table allows the overall effect of these assumptions to be examined. It is clear that the majority of assumptions lead to "over-estimation" of risks. This is appropriate because the risk assessment, while using as many realistic assumptions as possible, still has uncertainty. On balance the assumptions tend to be cautious, however, there are two assumptions that are of unknown effect in that they could lead to over- or under-estimation of risk. The first assumption relates to the use of the bioaccessibility of the sediments being the same as soil. From a human health perspective, the soil pathway is relatively minor and as such it is unlikely that the estimated risk estimates would change. For the terrestrial animals, soil represents a larger fraction of exposure; however, given that the estimated intakes are well below the TRV for all animals that consume soil with the exception of the hare, the results would not be different. For the hare, vegetation represents the highest pathway of exposure. However, the mean intake is slightly above the TRV and thus this assumption may influence the results for the hare. Also, it is not known whether the TRVs derived for wildlife based on laboratory test studies are over- or under-estimated.

TABLE 7.5-1
SUMMARY OF UNCERTAINTIES IN ASSESSMENT OF ECOLOGICAL AND HUMAN HEALTH RISKS

Assumption	Effect of Assumption			
	Possibly Leads to Under-estimation of Risks	**Leads to Neither Over- nor Under-estimation**	**Likely Leads to Over-Estimation of Risks**	**Could Lead to Over or Under-Estimation**
Arsenic Sources				
Estimates of Arsenic Releases from Giant Mine			x	
Estimates of Arsenic in Water, Soils, Sediments		x		
Estimates of Arsenic in Market Foods		x		
Arsenic Transport and Fate				
Mass Transfer Coefficients - Exchange between water column and sediment calibrated against measured levels		x		
Historic Loads to Area - Not known with certainty but estimated in part through model calibration		x		
Arsenic Intake by Ecological Receptors				
Residence Time of Aquatic Species - Assumed to be in each water body 100% of time - Fish - Benthos and Aquatic Plants		 x	x	
Aquatic Toxicity Reference Values - Based on Laboratory Toxicity Testing			x	
Dietary and Feeding Characteristics of Terrestrial Species - Based on Literature Information		x		
Exposure of Terrestrial Species - Assumed while in the study area to obtain all food and water from Baker Creek - Ducks assumed to spend 100% of whole time in study area on each waterbody			x x	

TABLE 7.5-1 (Cont'd)
SUMMARY OF UNCERTAINTIES IN ASSESSMENT OF ECOLOGICAL AND HUMAN HEALTH RISKS

Assumption	Effect of Assumption			
	Possibly Leads to Under-estimation of Risks	**Leads to Neither Over- nor Under-estimation**	**Likely Leads to Over-Estimation of Risks**	**Could Lead to Over or Under-Estimation**
Bioaccessibility - Assumed arsenic bioaccessibility measured in sediments is the same as for soils				x
Terrestrial Toxicity Reference Values - Based on Laboratory Toxicity Testing[1]				x
Arsenic Intake by Human Receptors				
Residency Time - Assumed to be present for a full 70-year lifetime at each location and to be exposed at maximum conditions			x	
Soil Ingestion for Humans - Assumed soil ingestion constant for whole year			x	
Backyard Garden Produce - Assumed to occur every day for whole year. Amount of produce grown based on literature studies			x	
Drinking Water Intakes - Assumed to be on municipal supply every day for 70 years		x		
Dietary Intake Rates of Food			x	
Local Meat Sources - Assumed that all arsenic is in toxic inorganic form			x	

TABLE 7.5-1 (Cont'd)
SUMMARY OF UNCERTAINTIES IN ASSESSMENT OF ECOLOGICAL AND HUMAN HEALTH RISKS

Assumption	Effect of Assumption			
	Possibly Leads to Under-estimation of Risks	**Leads to Neither Over- nor Under-estimation**	**Likely Leads to Over-Estimation of Risks**	**Could Lead to Over or Under-Estimation**
Local Fish Sources - Assumed 3% of total arsenic is in inorganic form		x		
- Assumed 78% of organic arsenic is in toxic form[3]			x	
Arsenic Toxicity Reference Values - Oral cancer slope factor based on Health Canada 2004 assessment for the Drinking Water Guideline[4]			x	

Notes:

1. From a human health perspective, the soil pathway is relatively minor and as such it is unlikely that the estimated risk estimates would change. For the terrestrial animals, soil represents a larger fraction of exposure; however, given that the estimated intakes are well below the TRV for all animals that consume soil with the exception of the hare, the findings would not be different.
2. It is unknown whether the toxicity reference values derived from laboratory studies on mice are directly applicable to the wildlife in question.
3. Additional research carried out on fish in Yellowknife Bay indicated that 3% of the total arsenic is in the inorganic form. However, the analytical method used was unable to specify non-toxic organic forms. The results of the test indicate that as much as 78% of the organic arsenic could potentially be toxic.
4. The derivation of risks using the 2004 Health Canada slope factor is cautious since it is based on upper bound estimates of exposure. However, there are other slope factors provided by the U.S. EPA and other Health Canada documents that are more restrictive.

REFERENCES

Agency for Toxic Substances and Disease Registry (ATSDR) 2000. *Toxicological Profile for Arsenic (Update).* U.S. Department of Health and Human Services.

Akaitcho Study 2000. *Risk Characterization of Arsenic Exposure from Consumption of Berries in the Akaitcho Territory.* Draft Final Report. March.

American Industrial Health Council (AIHC) 1992. *Improving Risk Characterization*. Proceedings of a Symposium, 26-27 September, 1991. American Industrial Health Council, Washington, DC.

Birge, W.J., J.E. Hudson, J.A. Black and A.G. Wasterman 1979a. *Embryo-larval Bioassays on Inorganic Coal Elements and In Situ Biomonitoring of Coal-waste Effluents.* In: Surface mining and fish/wildlife needs in the eastern United States, D.E. Samuel, J.R. Stauffer, C.H. Hocutt, and W.T. Mason Jr., eds. pp. 97-104. Fish and Wildlife Service, Office of Biological Services. FWS/OBS-78/81. U.S. Government Printing Office, Washington, D.C. {cited in CCME 1999, arsenic}.

Birge, W.J., J.A. Black, and A.G. Westerman 1979b. *Evaluation of Aquatic Pollutants Using Fish and Amphibian Eggs as Bioassay Organisms.* In: S.W. Nielsen, G. Migaki, and D.G. Scarpelli (Eds.), Symp. Animals Monitors Environ. Pollut., 1977, Storrs, CT 12: 108-118. {cited in U.S. EPA AQUIRE, arsenic, aluminum and lead, molybdenum}.

Borgmann, U., R. Cove and C. Loveridge 1980. *Effects of Metal on the Biomass Production Kinetics of Freshwater Copepods.* Can. J. Fish. Aquat. Sci. 37: 567-575. {cited in CCME 1999}.

Boucher, M., M.S. Evans, P. Simon, G. Lowe and D. Muir 1997. *Fort Resolution Domestic Fishing Zone Contaminants Research.* From Synopsis of Research Conducted under the 1995-1997 Northern Contaminants Program. Indian and Northern Affairs Canada: 74.

Braune, B.M., Environment Canada, Canadian Wildlife Service, National Wildlife Research Centre, Malone Associates 1997. *Contaminants in Waterfowl from Northern Canada. From Synopsis of Research Conducted Under the 1995/96 and 1996/97 Northern Contaminants Program.* Indian and Northern Affairs Canada: 155 – 159.

Byron W.R., G.W. Bierbower, J.B., Brouwer, *et al.* 1967. *Pathologic changes in rats and dogs from two-year feeding of sodium arsenite or sodium arsenate*. Toxicol Appl Pharmacol 10: 132-147.

Cadwell, R.D., B. Parkhurst, W. Warren-Hicks and J. Volosin 1993. *Aquatic Ecological Risk Assessment and Clean-up Goals for Metals Arising from Mining Operations.* pp 61-72. In Bender, E.S. and F.A. Jones (eds.) Applications of Ecological Risk Assessment to Hazardous Waste Site Remediation. Water Environment Federation, Alexandria, Virginia.

Canada North Environmental Services Limited Partnership (CanNorth) 2000. *Hatchet Lake Dietary Survey – Field Report*. March.

Canadian Council of Ministers of the Environment (CCME) 2002. *Canadian Environmental Quality Guidelines*.

Canadian Council of Ministers of the Environment (CCME) 1999. *Canadian Environmental Quality Guidelines.*

Canadian Council of Ministers of the Environment (CCME) 1997. *A Framework for Ecological Risk Assessment: Technical Appendices*. March.

Canadian Council of Ministers of the Environment (CCME) 1996. *A Framework for Ecological Risk Assessment: General Guidance*. March.

Cantor K.P. 2001. *Invited Commentary: Arsenic and Cancer of the Urinary Tract*. Am. J. Epidemiol. 2001 153: 419-421.

CANTOX Environmental Inc. 1999. *Deloro Village Exposure Assessment and Health Risk Characterization for Arsenic and Other Metals*. February 2, 1999.

Chan, L.. 2003. *Yellowknives Dene First Nation Land and Environment Committee's Medicinal Plants Study Report 2002-2003.* Prepared by School of Dietetics and Human Nutrition, McGill University. Prepared for Northern Contaminants Program, DIAND. May.

Cohen, S.M., L.L. Arnold, E. Uzvolgyi, M. Cano, M. St. John, S. Yamamoto, X. Lu, and X. Chris Le 2002. *Possible Role of Dimethylarsinous Acid in Dimethylarsinic Acid-Induced Urothelial Toxicity and Regeneration in the Rat*. Chem. Res. Toxicol. 2002, 15, 1150-1157. April.

Dabeka, R.W., A.D. McKenzie, G.M.A. Lacroix, C. Cleroux, S. Bowe, R.A. Graham, H.B.S. Conacher and P. Verdier 1993. *Survey of Arsenic in Total Diet Food Composites and Estimation of the Dietary intake of Arsenic by Canadian Adults and Children*. Journal of AOAC International. Vol 76. No. 1. pp14-25.

Davey, E., D. Maxwell and G. Stephens 1998. *Arsenic Levels in Berries and Soils from the Yellowknives Dene First Nation Traditional Territory*. From Synopsis of Research Conducted under the 1998-1999 Northern Contaminants Program. Indian and Northern Affairs Canada: 81-85.

Department of Indian Affairs and Northern Development (DIAND) 2004. *Arsenic Concentration and Speciation in Fish from Back Bay near Yellowknife, NT*. Toxicology Centre, University of Saskatchewan. Draft report. June 11, 2004. pp. 1-30.

Dillon Consulting Limited 2004. *Biological Sampling at Baker Creek 2003 – Technical Report Final.* Indian and Northern Affairs Canada. April 2004. 03-2049.

Dillon Consulting Limited 2002a. *Biological Sampling at Baker Creek – Summary Report.* Department of Indian and Northern Affairs Development. April 2002. 01-9670-3000.

Dillon Consulting Limited 2002b. *Biological Sampling at Baker Creek 2002 –Technical Report-Final.* Prepared for Indian and Northern Affairs Canada. 02-0465. November.

Dillon Consulting Limited 2002c. *Biological Sampling at Baker Creek 2002 – Technical Report Final.* Department of Indian and Northern Affairs Development. November 2002. 02-0465.

Elkin, B.T. and C. MacDonald 2000. *Heavy Metal and Radionuclide Contaminants in Caribou*. From Synopsis of Research Conducted under the 1998-1999 Northern Contaminants Program. Indian and Northern Affairs Canada: 169 – 173.

Elkin, B.T., C. MacDonald, J. Nishi, D. Panayi, M.A.D. Ferguson 1999. *Metal and Radionuclide Effects in Caribou* (*Rangifer tarandus*). From Synopsis of Research Conducted under the 1999-2000 Northern Contaminants Program. Indian and Northern Affairs Canada: 167 – 171.

Elkin, B.T., C. MacDonald, J. Nishi, D. Panayi, M.A.D. Ferguson 1998. *Metal and Radionuclide Accumulation and Effects in Caribou* (*Rangifer tarandus*). From Synopsis of Research Conducted under the 1998-1999 Northern Contaminants Program. Indian and Northern Affairs Canada: 97 – 102.

Environment Canada 1997. *Environmental Assessments of Priority Substances Under the Canadian Environmental Protection Act.* Guidance Manual Version 1.0. EPS 2/CC/3E Chemicals Evaluation Division Commercial Chemicals Evaluation Branch Environment Canada. March.

Environment Canada 1993. *Canadian Environmental Protection Act: Priority Substances List Assessment Report - Arsenic*.

Environmental and Occupational Health +Plus and SENES 2002. *Arsenic in Drinking Water in Newfoundland: History Response and Analysis of Potential Impacts.* July.

Evans, M.S., D. Muir, M. Whittle, G.A. Stern, L. Lockhart, G. Low, M. Boucher, B. Parlee and T. Heron 2001. *Spatial and Long-term Trends in Organic Contaminants and Metals in Fish Species Important to the Commercial, Sports and Domestic Fisheries of Great Slave Lake and the Slave River Ecosystem.* Synopsis of Research conducted under the 2000-2001 Northern Contaminants Program. Indian and Northern Affairs Canada.

Falk, M.R., M.D. Miller and S.J.M. Kostiuk 1973. *Data Supplement to: Biological Effects of Mining Wastes in the Northwest Territories.* Environment Canada Fisheries and Marine Service.

Federal-Provincial-Territorial Committee on Drinking Water 2004. *Arsenic in Drinking Water. Document issued for Public Comment, May 17, 2004.*

Francesconi, K.A. and J.S. Edmonds. 1987. *The Identification of Arsenobetaine as the Sole Water-Soluble Arsenic Constituent of the Tail Muscle of the Western King Prawn Penaeus latisulcatus.* Comp. Biochem Physiol.

Gamberg, M. and M. Palmer 1998. *Contaminants in Yukon Moose and Caribou – 1998*. From Synopsis of Research Conducted under the 1998-1999 Northern Contaminants Program. Indian and Northern Affairs Canada: 111 – 116.

Golder Associates 2004. *Chemical Analysis of Muskrat from the Giant Mine Site*. Raw Data. September.

Hanaoka, K., T. Fujita, M. Matsuura, S. Tagawa and T. Kaise. 1987. *Identification of Arsenobetaine as a Major Arsenic Compound in Muscle of Two Demersal Sharks, Shortnose Dogfish Squalus brevirostris and Starspotted Shark Mustelus manazo.* Comp. Biochem. Physiol. B. 86(4)681-2.

Health Canada 2004. *Arsenic in Drinking Water. Document for Public Comment.* Prepared by the Federal-Provincial-Territorial Committee on Drinking Water. Comment period ends May 17, 2005. November.

Health Canada 2003. *Federal Contaminated Site Risk Assessment in Canada – Part I: Guidance on Human Health Screening Level Risk Assessment (SLRA).* Version 1.1. October.

Health Canada 2002. *Toxicological Reference Doses for Trace Elements.* Food Directorate. Last updated in August 20 , 2002.

Health Canada 1994. *Human Health Risk Assessment for Priority Substances*. Canadian Environmental Protection Act.

HydroQual Consultants Incorporated and Gore & Storrie Limited 1989. *Water Quality Study of Yellowknife Bay: 1987 – 1988*.

International Programme for Chemical Safety (IPCS) 2000. *Short List Definitions. Harmonization Project (Terminology)*. Exposure Assessment Planning Workgroup. (http://www.ipcsharmonize.org/).

Jackson, F.J. 1998. *Yellowknife – Back Bay Summer Water Quality Monitoring Program (September 1992 to June 1995).* Water Resources Division. Indian and Northern Affairs Canada. April.

Jackson, F.J., C.N. Lafontaine and J. Klaverkamp 1996. *Yellowknife – Back Bay Study on Metal and Trace Element Contamination of Water, Sediment and Fish*. Department of Indian and Northern Affairs Canada. November.

Jacques Whitford 2003. *The Presence of Muskrats (ondatra zibethicus in Baker Creek Northwest Territories*. Prepared for Indian and Northern Affairs Canada.

Jenner, Henk A., and Joke P.M. Janssen-Mommen 1993. *Duckweed Lemna minor as a Tool for Testing Toxicity of Coal Residues and Polluted Sediments*. Arch. Environ. Contam. Toxicol. 25(1): 3-11. {cited in U.S. EPA AQUIRE}.

Kennedy, D., M. Boucher, B. Elkin *et al*. 1998. *Metals and Organic Contaminants in Beaver and Muskrat in the Slave River Delta Area, NWT*. From Synopsis of Research Conducted under the 1998-1999 Northern Contaminants Program. Indian and Northern Affairs Canada: 127 – 132.

Koch, I. 1998. *Arsenic and Antimony Species in the Terrestrial Environment.* Thesis. University of Waterloo.

Koch, I., L. Wang, C.A. Ollson, W.R. Cullen and K.J.Reimer 2000. *The Predominance of Inorganic Arsenic Species in Plants from Yellowknife, Northwest Territories, Canada.* Environmental Science and Technology. 34(1): 22-26.

Kohlmeyer, U., J. Kuballa and E. Jantzen. 2002. *Simultaneous Separation of 17 Inorganic and Organic Arsenic Compounds in Marine Biota by Means of High Performance Liquid Chromatography/Inductively Coupled Plasma Mass Spectrometry*. Rapid Commun. Mass. Spectrom. 16(10)964-74.

Liber, K. and S.L. White Sobey 2000. *Toxicity of Uranium, Molybdenum, Nickel, and Arsenic to Hyalella azteca and Chironomus Tentans in Water-only and Spiked-sediment Toxicity Tests.* Toxicology Centre, University of Saskatchewan.

Lorax Environmental 1999. *Assessment of Metal Speciation in Baker Creek, Yellowknife.* Report to Indian and Northern Affairs Canada. Water Resources Division.

MacDonald, D.D., B.L. Charlish, M.L. Haines and K. Brydges 1994. *Approach to the Assessment of Sediment Quality in Florida Coastal Waters: Volume 3 Supporting Documentation: Biological Effects Database for Sediment*. Florida Department of Environmental Protection, Tallahassee, FL. {Cited in: Toxicological Benchmarks for Screening Contaminants of Potential Concern for Effects on Sediment-Associated Biota: 1997 Revision, U.S. Department of Energy}.

Mace, I.S. 1998. *A Study of Arsenic Contamination from the Royal Oak Giant Mine, Yellowknife, Northwest Territories*, M.Eng. Thesis. May.

Maher, W.A. 1985. *The Presence of Arsenobetaine in Marine Animals*. Comp. Biochem. Physiol. C. 80(1):199-201.

Massachusetts Department of Environmental Protection (MADEP). 2002. *Technical Update: Calculation of Enhanced Soil Ingestion Rate.* Office of Research and Standards, MADEP, Boston, MA. Internet: http://www.state.ma.us/dep/ors/files/Soiling.doc

Meacher, D.M. et al. 2002. *Estimation of Multimedia Inorganic Arsenic Intake in the U.S. Population.* In: Human and Ecological Risk Assessment, Vol. 8, No.7, pp. 1697-1721.

Moore, J.W., S.J. Wheeler and D.J. Sutherland 1978. *The Effects of Metal Mines on Aquatic Ecosystems in the Northwest Territories II.* Giant Yellowknife Mines Limited. Report EPS 5-NW-78-9. June.

Morales, K.H., Ryan, L., Kuo, T.L., Wu, M.M., and Chen, C.J. 2000. *Risk of Internal Cancers from Arsenic in Drinking Water.* Environ. Health Perspect., 108: 655-661.

Mudroch, A., S.R. Joshi, D. Sutherland, P. Murdoch and K.M. Dickson 1989. *Geochemistry of Sediments in the Back Bay and Yellowknife Bay of the Great Slave Lake.* Enviro. Geol. Sci. Vol. 14, No. 1.

National Environmental Policy Institute (NEPI) 2000. *Assessing the Bioavailability of Metals in Soil for Use in Human Health Risk Assessments*. Bioavailability Policy Report Phase II: Metals Task Force Report. Summer.

O'Connor Associates 2000. *Assessment of Human Health Risks Posed by Arsenic Contamination in and Near Wawa, Ontario*. June.

Ollson, C.A. 1999. *Arsenic Concentration of the Terrestrial and Freshwater Environment Impacted by Gold Mining Operations Yellowknife, Northwest Territories.* Thesis. Royal Military College of Canada. November.

Ontario Ministry of the Environment (MOE) 2001. *Soil Investigation and Human Health Risk Assessment Report* for the Rodney St. Community, Port Colborne. October.

Puls, R. 1994. *Mineral Levels in Animal Health: Diagnostic Data.* 2nd Edition.Sherpa Internationa, Clearbrook, BC, Canada.

Receveur, O., A. Ing, L. Chan, H. Kuhnlein 1998. *Recovery of Yellowknives Dene Dietary Survey.* Centre for Indigenous Peoples' Nutrition and Environment (CINE). September.

Receveur, O., M. Boulay, C. Mills, W. Carpenter and H. Kuhnlein 1996. *Variance in Food Use in Dene/Metis Communities.* Centre for Indigenous Peoples' Nutrition and Environment (CINE). October.

Richardson, G.M. 1997. *Compendium of Canadian Human Exposure Factors for Risk Assessment.* O'Connor Associates.

Sahtu Dene Council 1998. Sahtu Caribou/Moose Sampling Program. From Synopsis of Research Conducted under the 1998-1999 Northern Contaminants Program. Indian and Northern Affairs Canada: 191 – 197.

Sample, B.E., D.M. Opresko, G.W. Suter II 1996. *Toxicological Benchmarks for Wildlife: 1996 Revision*. June 1996.

Schoof, R.A. et al. 1999. *A Market Basket Survey of Inorganic Arsenic in Food.* Food and Chemical Toxicology 37 (1999) 839 846. www.elsevier.com/locate/foodchemtox

Schroeder, H.A. and M. Mitchener. 1971. *Toxic Effects of Trace Elements on the Reproduction of Mice and Rats*. Arch. Environ. Health. 23: 102-106.

SENES Consultants Limited (SENES) 2003. *Screening Level Human Health Risk Assessment for Giant Mine*. Prepared for DIAND. November.

SENES Consultants Limited (SENES) 1985. *Probabilistic Model Development for the Assessment of the Long Term Effects of Uranium Mill Tailings in Canada – Phase II.* Prepared for the National Uranium Tailings Program, Canada Centre for Mineral and Energy Technology. Energy, Mines and Resources Canada.

Suner, M.A, V. Devesa, M.J. Clemente, D. Velez, R. Montoro, I. Urieta, M. Jalon and M.L. Macho. 2002. *Organoarsenical Species Contents in Fresh and Processed Seafood Products.* J. Agric. Food Chem 50(4):924-32.

Suter, G.W. II. 1991. *Screening Level Risk Assessment of Off-site Ecological Effects in Surface Waters Downstream of the U.S. Department of Energy Oak Ridge Reservation.* ORNL/ER-8. Oak Ridge National Laboratory, Oak Ridge, Tennessee.

Suter and Tsao 1996. *Toxicological Benchmarks for Screening Potential Contaminants of Concern for Effects on Aquatic Biota: 1996 Revision*. U.S. Department of Energy. June.

Thompson, P.A., J.A. Kurias and S.S. Mihok. 2002. *Using Benthic Invertebrates and Sediment Chemistry Monitoring Data for Estimating Lowest Effect/Severe Effect Levels at Uranium Mines In Canada*. Canadian Nuclear Safety Commission - Working Benchmarks. 2002 in preparation.

Tseng, W.P. 1977. *Effects and Dose-Response Relationships of Skin Cancer and Blackfoot Disease with Arsenic.* Environ. Health Perspect. 19: 109-119.

Tseng, W.P., H.M. Chu, S.W. How, *et al.* 1968. *Prevalence of Skin Cancer in an Endemic Area of Chronic Arsenicism in Taiwan.* J. Natl. Cancer Inst. 40: 453-463.

United States Environmental Protection Agency (U.S. EPA) 1997. *Exposure Factors Handbook.* Volume I to III – General Factors. EPA/600/P-95/002.

United States Environmental Protection Agency (U.S. EPA) 1992. *Framework for Ecological Risk Analysis.* EPA/630/R-92-001. Washington DC.

United States Environmental Protection Agency (U.S. EPA) 1988. *Arsenic, inorganic (CASRN 7440-38-2)*. Integrated Risk Information System (IRIS). File first on-line in February 1988. Internet: http://www.epa.gov/iris/subst/0278.htm

United States Fish and Wildlife Service (USFWS) 1964. *Pesticide-Wildlife Studies, 1963: A Review of Fish and Wildlife Services Investigations During the Calendar Year.* FWS Circular 199.

World Health Organization (WHO) 1996. *Guidelines for Drinking Water Quality.* 2nd edition. Vol. 2. Health criteria and other supporting information. Geneva. Internet: http//www.who.int/water_sanitation_health/dwq2vl/en/index1.html (Accessed Oct. 8, 2004)

Yost L.J., *et al.* 2004. *Estimation of Dietary Intake of Inorganic Arsenic in U.S. Children.* In: Human and Ecological Risk Assessment; 10: 473-483, 2004.

Yost L.J., Schoof, RA, Aucoin R. 1998. *Intake Of Inorganic Arsenic In The North American Diet*. In: Human and Ecological Risk Assessment; 4(1): 137-152.

APPENDIX A

YELLOWKNIFE AREA ENVIRONMENTAL DATA

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APPENDIX A YELLOWKNIFE AREA ENVIRONMENTAL DATA

Numerous investigations have been carried out to measure arsenic levels in components of the aquatic and terrestrial environments in the Yellowknife area in the past several decades. This appendix provides a compilation of data on measured arsenic levels in soils, garden produce, terrestrial vegetation, aquatic plants and fish. Where data permitted, site-specific transfer factors were calculated and are also presented here. These include soil-to-terrestrial vegetation, water-to-aquatic vegetation and water-to-fish muscle. Data on arsenic levels in Baker Creek, Back Bay and Yellowknife Bay water and sediments are provided in Appendix B. Much of the data included in this appendix was used to characterize source inputs to the pathways analysis of arsenic intakes by terrestrial species and people living in the study area.

A1.0 MEASURED ARSENIC CONCENTRATIONS IN SOIL

Wildlife and humans may be exposed to arsenic in soil as a result of the inadvertent ingestion of soil or as a result of uptake by vegetation which is subsequently consumed. Arsenic levels in soil in the Yellowknife area are high due to the surface exposure of naturally mineralized geologic material and due to past mining activities. Consequently, several investigations have been carried out over the years to characterize levels in the study area.

Table A1-1 summarizes measured soil concentrations for the Yellowknife, Con Mine and Dettah areas taken from literature sources. Arsenic concentrations measured on Latham Island and the N'Dilo community soils were available from a spreadsheet ('Locations Taiga General Export.xls') provided by Dr. Mark Richardson, Risklogic Scientific Services Inc., Ottawa. Table A1-2 contains the analytical results of arsenic measured in soil from the Giant Mine site and control sites in the City of Yellowknife (contained in a document prepared by EBA Engineering Consultants (1998) 'Ebaarsenic98.xls').

The data presented on Tables A1-1 and A1-2 were evaluated to determine summary statistics of the arsenic levels in soils at each of the four human receptor locations (i.e. the Giant Mine townsite; in the N'Dilo community on the north end of Latham Island; in the City of Yellowknife excluding Latham Island; and, in the Dettah community on the east shore of Yellowknife Bay). The distributions of the soil arsenic data were found to be characteristic of lognormal distributions hence, summary statistics were estimated for the geometric mean and geometric standard deviation. The summary statistics provided in Table A1-3 for each receptor location were input to the pathways analysis of soil arsenic intake by the respective receptors.

Summary statistics are also provided in Table A1-3 for the Giant Mine site. As expected, arsenic levels are much higher on the mine site than at the other locations included in the table.

TABLE A1-1
MEASURED ARSENIC LEVELS IN YELLOWKNIFE AREA SOIL

Yellowknife Area			
Soil Sample	**Location**	**Reference**	**Concentration (mg/kg (dw))**
29012	Yellowknife City, NW corner, Bristol Pit	ESG (2000b)	2.5
29014	Yellowknife City, SW end middle, Bristol Pit	ESG (2000b)	3.4
29018	Yellowknife City, Lions Comm Park	ESG (2000b)	3.5
29022	Yellowknife City, Lamoureaux Parkette	ESG (2000b)	3.8
29013	Yellowknife City, NE corner, Bristol Pit	ESG (2000b)	3.9
	natural background in City of Yellowknife	RMCC (1999)	4 to 70
29016	Yellowknife City, Trails End Park	ESG (2000b)	4.3
29017	Yellowknife City, Lions Comm Park	ESG (2000b)	4.4
19	Yellowknife, from ConMine	Risklogic (2002)	4.4
29019	Yellowknife City, Forrest Drive Park	ESG (2000b)	4.9
	Long Lake Beach	RMCC (1999)	5
4243	Yellowknife	ConMine (1987)	5
4261	Yellowknife	ConMine (1987)	5
6922	Yellowknife	ConMine (1987)	5
7038	Yellowknife	ConMine (1987)	5
7050	Yellowknife	ConMine (1987)	5
7056	Yellowknife	ConMine (1987)	5
7061	Yellowknife	ConMine (1987)	5
7075	Yellowknife	ConMine (1987)	5
7080	Yellowknife	ConMine (1987)	5
7083	Yellowknife	ConMine (1987)	5
12014	Yellowknife	ConMine (1987)	5
12016	Yellowknife	ConMine (1987)	5
12018	Yellowknife	ConMine (1987)	5
12020	Yellowknife	ConMine (1987)	5
12027	Yellowknife	ConMine (1987)	5

TABLE A1-1 (Cont'd)
MEASURED ARSENIC LEVELS IN YELLOWKNIFE AREA SOIL

Yellowknife Area			
Soil Sample	**Location**	**Reference**	**Concentration (mg/kg (dw))**
12039	Yellowknife	ConMine (1987)	5
12040	Yellowknife	ConMine (1987)	5
12042	Yellowknife	ConMine (1987)	5
12045	Yellowknife	ConMine (1987)	5
	natural background in City of Yellowknife	ESG (2000b)	5 to 150
	perimeter of tailings ponds	RMCC (1999)	5 to 1165
29020	Yellowknife City, Forrest Drive Park	ESG (2000b)	5.4
29021	Yellowknife City, Borealis Co-op Park	ESG (2000b)	5.6
	Stout Road Park	RMCC (1999)	7
29186	N'Dilo, older playground, 0-15 cm depth	ESG (2000b)	7
01AS027	N'Dilo	YASRC	7.1
29024	Yellowknife City, Northlands Parkette	ESG (2000b)	7.1
	north of Giant Mine (Ingraham Trail)	ESG (2000a)	7.1 to 127
9	Yellowknife, from ConMine (1999)	Risklogic (2002)	7.3
29079	Yellowknife City, School Draw Park	ESG (2000b)	7.6
5	Yellowknife, from ConMine (1999)	Risklogic (2002)	7.9
	Horton Drive Park	RMCC (1999)	8
29080	Yellowknife City, Doornbos Park	ESG (2000b)	8
29015	Latham Island Community Park	ESG (2000b)	8.6
4273	Yellowknife	ConMine (1987)	10
4281	Yellowknife	ConMine (1987)	10
7011	Yellowknife	ConMine (1987)	10
7053	Yellowknife	ConMine (1987)	10
7092	Yellowknife	ConMine (1987)	10
7250	Yellowknife	ConMine (1987)	10
7256	Yellowknife	ConMine (1987)	10
258	Yellowknife	ConMine (1987)	10

TABLE A1-1 (Cont'd)
MEASURED ARSENIC LEVELS IN YELLOWKNIFE AREA SOIL

Yellowknife Area			
Soil Sample	**Location**	**Reference**	**Concentration (mg/kg (dw))**
12011	Yellowknife	ConMine (1987)	10
12017	Yellowknife	ConMine (1987)	10
12021	Yellowknife	ConMine (1987)	10
12022	Yellowknife	ConMine (1987)	10
12023	Yellowknife	ConMine (1987)	10
12033	Yellowknife	ConMine (1987)	10
12035	Yellowknife	ConMine (1987)	10
12046	Yellowknife	ConMine (1987)	10
	Frame Lake Beach	RMCC (1999)	11
	playground of William McDonald School	RMCC (1999)	11
26	Yellowknife, from ConMine (1999)	Risklogic (2002)	11
28	Yellowknife, from ConMine (1999)	Risklogic (2002)	11
29079	Yellowknife City, Doornbos Park	ESG (2000b)	11.6
01AS026	N'Dilo	YASRC	11.8
Garden 9	Yellowknife City, Old Town YK	ESG (2001b)	12
21	Yellowknife, from ConMine (1999)	Risklogic (2002)	12
27	Yellowknife, from ConMine (1999)	Risklogic (2002)	13
01AS032	Latham Island	YASRC	13.3
4226	Yellowknife	ConMine (1987)	15
4245	Yellowknife	ConMine (1987)	15
4247	Yellowknife	ConMine (1987)	15
4251	Yellowknife	ConMine (1987)	15
4271	Yellowknife	ConMine (1987)	15
6929	Yellowknife	ConMine (1987)	15
7015	Yellowknife	ConMine (1987)	15
7019	Yellowknife	ConMine (1987)	15

TABLE A1-1 (Cont'd)
MEASURED ARSENIC LEVELS IN YELLOWKNIFE AREA SOIL

Yellowknife Area			
Soil Sample	**Location**	**Reference**	**Concentration (mg/kg (dw))**
7026	Yellowknife	ConMine (1987)	15
7042	Yellowknife	ConMine (1987)	15
7058	Yellowknife	ConMine (1987)	15
7069	Yellowknife	ConMine (1987)	15
7081	Yellowknife	ConMine (1987)	15
7082	Yellowknife	ConMine (1987)	15
7255	Yellowknife	ConMine (1987)	15
7267	Yellowknife	ConMine (1987)	15
32	Yellowknife	Risklogic (2002)	16
01AS031	Latham Island	YASRC	18.1
29074	marina next to building 0-10 cm	ESG (2001a)	19
	Giant Mine Townsite	ESG (2000b)	19 to 1850
01AS029	Latham Island	YASRC	19.8
14	Yellowknife	Risklogic (2002)	20
25	Yellowknife	Risklogic (2002)	20
4236	Yellowknife	ConMine (1987)	20
4275	Yellowknife	ConMine (1987)	20
4278	Yellowknife	ConMine (1987)	20
6915	Yellowknife	ConMine (1987)	20
7009	Yellowknife	ConMine (1987)	20
7020	Yellowknife	ConMine (1987)	20
7024	Yellowknife	ConMine (1987)	20
7027	Yellowknife	ConMine (1987)	20
7030	Yellowknife	ConMine (1987)	20
7031	Yellowknife	ConMine (1987)	20
7033	Yellowknife	ConMine (1987)	20

TABLE A1-1 (Cont'd)
MEASURED ARSENIC LEVELS IN YELLOWKNIFE AREA SOIL

Yellowknife Area			
Soil Sample	**Location**	**Reference**	**Concentration (mg/kg (dw))**
7057	Yellowknife	ConMine (1987)	20
7059	Yellowknife	ConMine (1987)	20
7077	Yellowknife	ConMine (1987)	20
7097	Yellowknife	ConMine (1987)	20
7271	Yellowknife	ConMine (1987)	20
14	Yellowknife	Risklogic (2002)	20
25	Yellowknife	Risklogic (2002)	20
29069/70	Giant Mine Townsite - crushed rock fill	ESG (2000a)	22.5
4282	Yellowknife	ConMine (1987)	25
6917	Yellowknife	ConMine (1987)	25
7018	Yellowknife	ConMine (1987)	25
7021	Yellowknife	ConMine (1987)	25
7036	Yellowknife	ConMine (1987)	25
7045	Yellowknife	ConMine (1987)	25
7073	Yellowknife	ConMine (1987)	25
7090	Yellowknife	ConMine (1987)	25
7270	Yellowknife	ConMine (1987)	25
12034	Yellowknife	ConMine (1987)	25
12036	Yellowknife	ConMine (1987)	25
6	Yellowknife	Risklogic (2002)	26
8	Yellowknife	Risklogic (2002)	29
Garden 5	Latham Island	ESG (2001b)	30
4249	Yellowknife	ConMine (1987)	30
4283	Yellowknife	ConMine (1987)	30
7034	Yellowknife	ConMine (1987)	30

TABLE A1-1 (Cont'd)
MEASURED ARSENIC LEVELS IN YELLOWKNIFE AREA SOIL

Yellowknife Area			
Soil Sample	**Location**	**Reference**	**Concentration (mg/kg (dw))**
7044	Yellowknife	ConMine (1987)	30
7251	Yellowknife	ConMine (1987)	30
7259	Yellowknife	ConMine (1987)	30
12013	Yellowknife	ConMine (1987)	30
	City of Yellowknife	median, RMCC (1999)	31
12	Yellowknife, from ConMine (1999)	Risklogic (2002)	31
	natural background in City of Yellowknife	median, RMCC (1999)	32
29068	Giant Mine Townsite - lawn	ESG (2000a)	32
29	Yellowknife, from ConMine (1999)	Risklogic (2002)	33
7	Yellowknife, from ConMine (1999)	Risklogic (2002)	33
29182	N'Dilo, North of boat launch (200 lot), 0-15 cm	ESG (2000b)	34
15	Yellowknife, from ConMine (1999)	ESG (200b)	35
4240	Yellowknife	ConMine (1987)	35
4277	Yellowknife	ConMine (1987)	35
6920	Yellowknife	ConMine (1987)	35
6930	Yellowknife	ConMine (1987)	35
7051	Yellowknife	ConMine (1987)	35
7064	Yellowknife	ConMine (1987)	35
7086	Yellowknife	ConMine (1987)	35
7253	Yellowknife	ConMine (1987)	35
7273	Yellowknife	ConMine (1987)	35
Garden 6	Yellowknife City, Downtown YK	ESG (2001b)	35
29078	Yellowknife City, Niven Lake Residential	ESG (2000b)	35
29067	Giant Mine Townsite - lawn	ESG (2000a)	36
PSO23-2300-01	Areas 7 (Townsite Road)	Golder (2001)	36.1

TABLE A1-1 (Cont'd)
MEASURED ARSENIC LEVELS IN YELLOWKNIFE AREA SOIL

Yellowknife Area			
Soil Sample	**Location**	**Reference**	**Concentration (mg/kg (dw))**
29062	Giant Mine Townsite - crushed rock fill	ESG (2000a)	37
16	Con Mine 1999 Latham Island,	Risklogic (2002)	37
2	Yellowknife, from ConMine (1999)	Risklogic (2002)	37
24	Yellowknife, from ConMine (1999)	Risklogic (2002)	38
4250	Yellowknife	ConMine (1987)	40
6913	Yellowknife	ConMine (1987)	40
6931	Yellowknife	ConMine (1987)	40
7047	Yellowknife	ConMine (1987)	40
7063	Yellowknife	ConMine (1987)	40
7249	Yellowknife	ConMine (1987)	40
12041	Yellowknife	ConMine (1987)	40
29061	Giant Mine Townsite - lawn	ESG (2000a)	40
01AS024	N'Dilo	YASRC	40.1
41	Yellowknife, from ConMine (1999)	Risklogic (2002)	41
30	Yellowknife, from ConMine (1999)	Risklogic (2002)	42
29075	marina between boats 0-10 cm	ESG (2001a)	43
29054	Giant Mine Townsite - lawn	ESG (2000a)	45
4227	Yellowknife	ConMine (1987)	45
6914	Yellowknife	ConMine (1987)	45
6926	Yellowknife	ConMine (1987)	45
6928	Yellowknife	ConMine (1987)	45
7078	Yellowknife	ConMine (1987)	45
7087	Yellowknife	ConMine (1987)	45
7099	Yellowknife	ConMine (1987)	45
7252	Yellowknife	ConMine (1987)	45
7262	Yellowknife	ConMine (1987)	45

TABLE A1-1 (Cont'd)
MEASURED ARSENIC LEVELS IN YELLOWKNIFE AREA SOIL

Yellowknife Area			
Soil Sample	**Location**	**Reference**	**Concentration (mg/kg (dw))**
7272	Yellowknife	ConMine (1987)	45
12025	Yellowknife	ConMine (1987)	45
12037	Yellowknife	ConMine (1987)	45
	Giant Mine Townsite	ESG (2001a)	48 to 3277
01AS030	Latham Island	YASRC	49.9
7022	Yellowknife	ConMine (1987)	50
7048	Yellowknife	ConMine (1987)	50
12044	Yellowknife	ConMine (1987)	50
01AS025	N'Dilo	YASRC	50.4
17	Yellowknife, from ConMine (1999)	Risklogic (2002)	51
29184	N'Dilo, Turner Road, 6 lot, 0-10 cm	ESG (2000b)	52
20	Yellowknife, from ConMine (1999)	ESG (2000b)	53.5
4246	Yellowknife	ConMine (1987)	55
4255	Yellowknife	ConMine (1987)	55
4258	Yellowknife	ConMine (1987)	55
6918	Yellowknife	ConMine (1987)	55
7008	Yellowknife	ConMine (1987)	55
7043	Yellowknife	ConMine (1987)	55
7062	Yellowknife	ConMine (1987)	55
7091	Yellowknife	ConMine (1987)	55
12012	Yellowknife	ConMine (1987)	55
12026	Yellowknife	ConMine (1987)	55
29076	marina forested area 3-7 cm	ESG (2001a)	56
Garden 10	Latham Island	ESG (2001b)	56
3	Yellowknife, from ConMine (1999)	Risklogic (2002)	56
4	Yellowknife, from ConMine (1999)	Risklogic (2002)	58

TABLE A1-1 (Cont'd)
MEASURED ARSENIC LEVELS IN YELLOWKNIFE AREA SOIL

Yellowknife Area			
Soil Sample	**Location**	**Reference**	**Concentration (mg/kg (dw))**
4234	Yellowknife	ConMine (1987)	60
4257	Yellowknife	ConMine (1987)	60
7010	Yellowknife	ConMine (1987)	60
7055	Yellowknife	ConMine (1987)	60
7248	Yellowknife	ConMine (1987)	60
12024	Yellowknife	ConMine (1987)	60
12028	Yellowknife	ConMine (1987)	60
12031	Yellowknife	ConMine (1987)	60
12032	Yellowknife	ConMine (1987)	60
1	Yellowknife, from ConMine (1999)	Risklogic (2002)	61
29077	Giant Mine Townsite - forrested area (cruising club)	ESG (2000b)	64
4232	Yellowknife	ConMine (1987)	65
4260	Yellowknife	ConMine (1987)	65
6924	Yellowknife	ConMine (1987)	65
4268	Yellowknife	ConMine (1987)	70
12043	Yellowknife	ConMine (1987)	70
29059/60	Giant Mine Townsite - lawn	ESG (2000a)	70
13	Yellowknife, from ConMine (1999)	Risklogic (2002)	74
7095	Yellowknife	ConMine (1987)	75
7261	Yellowknife	ConMine (1987)	75
12029	Yellowknife	ConMine (1987)	75
PSO27-2300-04	Areas 6 (Townsite)	Golder (2001)	77.6
4279	Yellowknife	ConMine (1987)	80
7014	Yellowknife	ConMine (1987)	80
7023	Yellowknife	ConMine (1987)	80
7065	Yellowknife	ConMine (1987)	80

TABLE A1-1 (Cont'd)
MEASURED ARSENIC LEVELS IN YELLOWKNIFE AREA SOIL

Yellowknife Area			
Soil Sample	**Location**	**Reference**	**Concentration (mg/kg (dw))**
7254	Yellowknife	ConMine (1987)	80
7265	Yellowknife	ConMine (1987)	80
29137	Giant Mine Townsite - S end Garden 1	ESG (2001b)	81
7085	Yellowknife	ConMine (1987)	85
01AS028	N'Dilo	YASRC	88.6
01AS023	N'Dilo	YASRC	89.3
7025	Yellowknife	ConMine (1987)	90
7084	Yellowknife	ConMine (1987)	90
7260	Yellowknife	ConMine (1987)	90
7028	Yellowknife	ConMine (1987)	95
7037	Yellowknife	ConMine (1987)	95
7049	Yellowknife	ConMine (1987)	95
7096	Yellowknife	ConMine (1987)	95
01AS016	N'Dilo,	YASRC	96.6
7012	Yellowknife	ConMine (1987)	100
	open field of Kasteel Drive	RMCC (1999)	102
10	Yellowknife, from ConMine (1999)	Risklogic (2002)	105
6921	Yellowknife	ConMine (1987)	105
7266	Yellowknife	ConMine (1987)	105
23	Yellowknife, from ConMine (1999)	Risklogic (2002)	107
	Department of Indian and Northern Affairs Water Resources Laboratory	RMCC (1999)	107
7041	Yellowknife	ConMine (1987)	110
7068	Yellowknife	ConMine (1987)	110
7098	Yellowknife	ConMine (1987)	110
01AS021	N'Dilo	YASRC	110
PSO22-2300-02	Areas 7 (Townsite Road)	Golder (2001)	110

TABLE A1-1 (Cont'd)
MEASURED ARSENIC LEVELS IN YELLOWKNIFE AREA SOIL

Yellowknife Area			
Soil Sample	**Location**	**Reference**	**Concentration (mg/kg (dw))**
4233	Yellowknife	ConMine (1987)	115
4256	Yellowknife	ConMine (1987)	115
7040	Yellowknife	ConMine (1987)	120
01AS020	N'Dilo	YASRC	123
6925	Yellowknife	ConMine (1987)	125
4223	Yellowknife	ConMine (1987)	130
4238	Yellowknife	ConMine (1987)	130
4286	Yellowknife	ConMine (1987)	130
01AS022	N'Dilo	YASRC	131
29023	Giant Mine Townsite - sandbox, playground	ESG (2000b)	132
	Old Airport Road	RMCC (1999)	133
31	Yellowknife, from ConMine (1999)	Risklogic (2002)	133
01AS015	N'Dilo	YASRC	138
4241	Yellowknife	ConMine (1987)	140
4269	Yellowknife	ConMine (1987)	140
6919	Yellowknife	ConMine (1987)	140
7071	Yellowknife	ConMine (1987)	140
7093	Yellowknife	ConMine (1987)	140
12030	Yellowknife	ConMine (1987)	140
29066	Giant Mine Townsite - sand	ESG (2000a)	145
18	Yellowknife	Risklogic (2002)	148
	parking lot of the Department of Fisheries and Oceans warehouse on McDonald Drive	RMCC (1999)	148
01AS019	N'Dilo	YASRC	160
4288	Yellowknife	ConMine (1987)	150
4289	Yellowknife	ConMine (1987)	150
7079	Yellowknife	ConMine (1987)	150

TABLE A1-1 (Cont'd)
MEASURED ARSENIC LEVELS IN YELLOWKNIFE AREA SOIL

Yellowknife Area			
Soil Sample	**Location**	**Reference**	**Concentration (mg/kg (dw))**
4224	Yellowknife	ConMine (1987)	155
4266	Yellowknife	ConMine (1987)	160
4272	Yellowknife	ConMine (1987)	160
7088	Yellowknife	ConMine (1987)	160
7263	Yellowknife	ConMine (1987)	160
7074	Yellowknife	ConMine (1987)	165
7060	Yellowknife	ConMine (1987)	170
7094	Yellowknife	ConMine (1987)	170
7268	Yellowknife	ConMine (1987)	170
29144	Giant Mine Townsite - Across from house Garden 1	ESG (2001b)	174
4252	Yellowknife	ConMine (1987)	175
01AS018	N'Dilo	YASRC	175
4242	Yellowknife	ConMine (1987)	180
4262	Yellowknife	ConMine (1987)	180
4248	Yellowknife	ConMine (1987)	185
4229	Yellowknife	ConMine (1987)	190
29178	N'Dilo, north point, NE side, 0-10 cm	ESG (2000b)	194
7070	Yellowknife	ConMine (1987)	195
01AS017	N'Dilo	YASRC	199
7039	Yellowknife	ConMine (1987)	200
4264	Yellowknife	ConMine (1987)	205
7264	Yellowknife	ConMine (1987)	205
7066	Yellowknife	ConMine (1987)	210
4280	Yellowknife	ConMine (1987)	220
29055	Giant Mine Townsite - lawn	ESG (2000a)	222
29183	N'Dilo, SW end, between 99 lot and road, 0-7 cm	ESG (2000b)	226
4274	Yellowknife	ConMine (1987)	230

TABLE A1-1 (Cont'd)
MEASURED ARSENIC LEVELS IN YELLOWKNIFE AREA SOIL

Yellowknife Area			
Soil Sample	**Location**	**Reference**	**Concentration (mg/kg (dw))**
4267	Yellowknife	ConMine (1987)	240
4284	Yellowknife	ConMine (1987)	240
4263	Yellowknife	ConMine (1987)	250
7046	Yellowknife	ConMine (1987)	250
7100	Yellowknife	ConMine (1987)	260
4253	Yellowknife	ConMine (1987)	265
6923	Yellowknife	ConMine (1987)	270
7016	Yellowknife	ConMine (1987)	270
7269	Yellowknife	ConMine (1987)	295
	reasonable upper limit (90^{th} percentile) for background in Yellowknife	Risklogic (2002)	300
7072	Yellowknife	ConMine (1987)	310
4244	Yellowknife	ConMine (1987)	320
4235	Yellowknife	ConMine (1987)	340
7257	Yellowknife	ConMine (1987)	345
29143	Giant Mine Townsite - Next to house Garden 1	ESG (2001b)	351
7089	Yellowknife	ConMine (1987)	360
29187	N'Dilo, S border, part of 909 lot, 2-10 cm depth	ESG (2000b)	360
29057	Giant Mine Townsite - lawn	ESG (2000a)	366
4239	Yellowknife	ConMine (1987)	380
4265	Yellowknife	ConMine (1987)	425
29072	marina pumphouse 0-10 cm depth	ESG (2001a)	471
4230	Yellowknife	ConMine (1987)	485
4237	Yellowknife	ConMine (1987)	485
4285	Yellowknife	ConMine (1987)	490
4270	Yellowknife	ConMine (1987)	880
PSO-28-2300-01	Areas 6 (Townsite)	Golder (2001)	891

TABLE A1-1 (Cont'd)
MEASURED ARSENIC LEVELS IN YELLOWKNIFE AREA SOIL

Yellowknife Area			
Soil Sample	**Location**	**Reference**	**Concentration (mg/kg (dw))**
7076	Yellowknife	ConMine (1987)	905
4276	Yellowknife	ConMine (1987)	1190
29056	Giant Mine Townsite - crushed rock fill	ESG (2000a)	1204
29064	Giant Mine Townsite - crushed rock fill	ESG (2000a)	1490
29063	Giant Mine Townsite - crushed rock fill	ESG (2000a)	1500
PSO24-2300-02	Areas 6 (Townsite)	Golder (2001)	1750
29053	Giant Mine Townsite - crushed rock fill	ESG (2000a)	1800
29058	Giant Mine Townsite - crushed rock fill	ESG (2000a)	1850
PSO24-2300-01	Areas 6 (Townsite)	Golder (2001)	3280
PSO25-2300-01	Areas 6 (Townsite)	Golder (2001)	3700
PSO24-2300-03	Areas 6 (Townsite)	Golder (2001)	4900
PSO27-2300-02	Areas 6 (Townsite)	Golder (2001)	6330
PSO-28-2300-02	Areas 6 (Townsite)	Golder (2001)	7120
PSO27-2300-01	Areas 6 (Townsite)	Golder (2001)	8380
PSO27-2300-03	Areas 6 (Townsite)	Golder (2001)	16600
Dettah Receptor Area			
	Location	**Notes**	**Concentration (ppm (dw))**
29195	Latham Island, W side of main road, N part	ESG (2000b)	7.2
29188	North of dock, behind firewall	ESG (2000b)	8.4
29192	Playground behind school	ESG (2000b)	11
29189	SW part, between LL and MM,	ESG (2000b)	19
29190	SW part, between LL and MM,	ESG (2000b)	28
29193	NE part, between S and R	ESG (2000b)	39
29191	SE part near W and dock at cul-de-sac	ESG (2000b)	116
29194	on hill near K and J	ESG (2000b)	144

Note: mg/kg dw = parts per million dry weight which is equivalent to µg/g (dw) or mg/kg (dw).

TABLE A1-2
MEASURED ARSENIC LEVELS IN GIANT MINE SITE SOIL

Giant Mine Property			
Soil Sample	**Location**	**Reference**	**Concentration (mg/kg (dw))**
PSO-11-2200-03	Area 1 (Mill)	Golder (2001)	5.2
S1	GM Bhnd A2 Opn Pit	EBA (1998)	22
PSO-35-2300-02	Area 8 (S. Tailings Impoundment)	Golder (2001)	22.4
G49	Downstream of Dam #3 near site S27	EBA (1998)	24
S15	GM brock quarry	EBA (1998)	25
PSO-40-2500-01	Akaitcho Area	Golder (2001)	35.9
G47	Kam Lake Road - in the bush near the Telsat Base	EBA (1998)	40
PSO-39-2500-01	Akaitcho Area	Golder (2001)	41.7
PSO-06-2200-02	Areas 2,3,4 (West of TPR)	Golder (2001)	50.2
PSO-32-2300-002	Area 5 (Propane Tanks)	Golder (2001)	50.3
PSO-10-2200-01	Areas 2,3,4 (West of TPR)	Golder (2001)	51.5
S4	Gm Bwtn A1 & A2 opn pts	EBA (1998)	52
S5	GM Bhnd A Shaft	EBA (1998)	53
G1	Near akaitcho warehouse	EBA (1998)	60
PSO-41-2500-01	Akaitcho Area	Golder (2001)	71.7
PSO-05-2200-01	Areas 2,3,4 (West of TPR)	Golder (2001)	81.6
G40	Approximately 200 yards north of the old PCB building near S19	EBA (1998)	100
PSO-17-2200-03	Area 1 (Mill)	Golder (2001)	110
PSO-35-2300-01	Area 8 (S. Tailings Impoundment)	Golder (2001)	113
S26	GM near N pond	EBA (1998)	114
S8	GM Bhnd A1 opn pit	EBA (1998)	120
PSO-01-2200-02	Area 1 (Mill)	Golder (2001)	130
S89	GM B shaft	EBA (1998)	135
S14	GM bhnd #7 dam	EBA (1998)	140
S23	GM Carbon Columns	EBA (1998)	140
PSO-21-2500-04	Area 1 (Mill)	Golder (2001)	143
PSO-30-2300-01	Area 5 (Propane Tanks)	Golder (2001)	146

TABLE A1-2 (Cont'd)
MEASURED ARSENIC LEVELS IN GIANT MINE SITE SOIL

Giant Mine Property			
Soil Sample	**Location**	**Reference**	**Concentration (mg/kg (dw))**
G10	C dry bunker C fuel storage tanks S37	EBA (1998)	160
S34	GM akaitcho	EBA (1998)	160
PSO-17-2200-02	Area 1 (Mill)	Golder (2001)	165
S32	GM hzrdus wstdmp ste	EBA (1998)	190
PSO-02-2200-01	Area 1 (Mill)	Golder (2001)	192
PSO-20-2200-01	Area 1 (Mill)	Golder (2001)	199
S20	GM TRP plant site	EBA (1998)	200
S33	GM akaitcho	EBA (1998)	210
S17	GM Ingh tr (bhnd mill)	EBA (1998)	220
PSO-31-2300-02	Area 5 (Propane Tanks)	Golder (2001)	221
PSO-37-2300-03	Area 8 (S. Tailings Impoundment)	Golder (2001)	227
PSO-21-2500-03	Area 1 (Mill)	Golder (2001)	247
PSO-09-2200-02	Areas 2,3,4 (West of TPR)	Golder (2001)	249
S11	GM back yard	EBA (1998)	260
S65	GM fuel strg-lmbr yard	EBA (1998)	270
S38	GM fuel tnks acr C dry	EBA (1998)	270
S7	GM Tnks Rd A Shft	EBA (1998)	280
S10	GM gowganda yard	EBA (1998)	290
G37	North side of MEG garage close to oil containment sump for boiler fuel tank near S58	EBA (1998)	300
PSO-37-2300-01	Area 8 (S. Tailings Impoundment)	Golder (2001)	307
G43	Between Legislative Assembly Building parking lot and Frame Lake Shoreline	EBA (1998)	310
PSO-36-2300-01	Area 8 (S. Tailings Impoundment)	Golder (2001)	313
S18	GM UBC Quarry	EBA (1998)	320
G15	Mill bunker C fuel storage tank near S67	EBA (1998)	320
S9	GM Near A1 opn pit	EBA (1998)	330
G12	Electrical shop area near S50	EBA (1998)	350
PSO-06-2200-01	Areas 2,3,4 (West of TPR)	Golder (2001)	351

TABLE A1-2 (Cont'd)
MEASURED ARSENIC LEVELS IN GIANT MINE SITE SOIL

Giant Mine Property			
Soil Sample	**Location**	**Reference**	**Concentration (mg/kg (dw))**
S13	GM near pit shop	EBA (1998)	370
S77	GM refinery sump	EBA (1998)	390
S22	GM Polishing Pond	EBA (1998)	400
G42	Gowganda scrap yard near S12	EBA (1998)	400
S16	GM bhnd S pond	EBA (1998)	420
S68	GM fuel strg-lmbr yard	EBA (1998)	430
S47	GM near hoist room	EBA (1998)	430
S21	GM TRP plant site	EBA (1998)	430
PSO-32-2300-01	Area 5 (Propane Tanks)	Golder (2001)	437
G2	Below east side of Akaitcho bunkhouses near S33	EBA (1998)	450
PSO-37-2300-04	Area 8 (S. Tailings Impoundment)	Golder (2001)	480
G30	On bank alongside old Prelude Lake Hwy west of AC Kiln building west of G25	EBA (1998)	490
PSO-31-2300-01	Area 5 (Propane Tanks)	Golder (2001)	516
S28	GM near dam #3	EBA (1998)	530
S81	GM stack	EBA (1998)	530
G16	Lumber storage area near S70	EBA (1998)	540
S25	GM pocket lake area	EBA (1998)	560
PSO-04-2200-02	Areas 2,3,4 (West of TPR)	Golder (2001)	562
S29	GM near NW pond	EBA (1998)	590
G38	mill reagent storage area quonset hut yard near S74	EBA (1998)	600
G26	On bank immediately to the west of the AC kiln building - south end of bank	EBA (1998)	600
PSO-13-2500-04	Area 1 (Mill)	Golder (2001)	628
S80	GM BC Roaster	EBA (1998)	650
PSO-13-2500-02	Area 1 (Mill)	Golder (2001)	652
PSO-12-2200-03	Area 1 (Mill)	Golder (2001)	679
G18	B shaft area near S91	EBA (1998)	700
S44	GM near C dry	EBA (1998)	730

TABLE A1-2 (Cont'd)
MEASURED ARSENIC LEVELS IN GIANT MINE SITE SOIL

Giant Mine Property			
Soil Sample	**Location**	**Reference**	**Concentration (mg/kg (dw))**
PSO-16-2200-01	Area 1 (Mill)	Golder (2001)	753
S31	GM bhnd NW pond	EBA (1998)	760
PSO-15-2200-02	Area 1 (Mill)	Golder (2001)	782
PSO-15-2200-02	Area 1 (Mill)	Golder (2001)	802
PSO-13-2500-03	Area 1 (Mill)	Golder (2001)	804
S59	GM BC screenhouse	EBA (1998)	810
S76	GM refinery sump	EBA (1998)	810
PSO-09-2200-01	Areas 2,3,4 (West of TPR)	Golder (2001)	824
PSO-01-2200-01	Area 1 (Mill)	Golder (2001)	841
PSO-04-2200-01	Areas 2,3,4 (West of TPR)	Golder (2001)	873
PSO-18-2200-03	Area 1 (Mill)	Golder (2001)	919
G32	On bank alongside old Prelude Lake Hwy west of yard between the mill and the roaster near rse hse	EBA (1998)	920
G36	Southeast side of MEG garage - approximately 50 feet from the garage near S57	EBA (1998)	920
G14	Fuel storage in lumber yard near S66	EBA (1998)	940
PSO-14-2200-02	Area 1 (Mill)	Golder (2001)	943
S67	GM fuel strg-lmbr yard	EBA (1998)	990
S37	GM fuel tnks acr C dry	EBA (1998)	1020
PSO-08-2200-01	Areas 2,3,4 (West of TPR)	Golder (2001)	1060
S51	GM warehouse	EBA (1998)	1060
S61	GM BC Refinery	EBA (1998)	1090
S90	GM B shaft	EBA (1998)	1100
G33	On bank alongside old Prelude Lake Hwy west of mill (solution sump area)	EBA (1998)	1110
PSO-12-2200-03	Area 1 (Mill)	Golder (2001)	1120
S50	GM new electrical shop	EBA (1998)	1130
S36	GM near admin bldg	EBA (1998)	1180
S79	GM roaster	EBA (1998)	1180
S62	GM refinery	EBA (1998)	1210

TABLE A1-2 (Cont'd)
MEASURED ARSENIC LEVELS IN GIANT MINE SITE SOIL

Giant Mine Property			
Soil Sample	**Location**	**Reference**	**Concentration (mg/kg (dw))**
S91	GM B shaft	EBA (1998)	1230
PSO-08-2200-02	Areas 2,3,4 (West of TPR)	Golder (2001)	1270
PSO-15-2200-03	Area 1 (Mill)	Golder (2001)	1320
PSO-18-2200-03	Area 1 (Mill)	Golder (2001)	1330
G25	On bank immediately to the west of the AC kiln building - north end of bank	EBA (1998)	1350
S84	GM baghouse	EBA (1998)	1360
PSO-18-2200-02	Area 1 (Mill)	Golder (2001)	1390
S66	GM fuel strg-lmbr yard	EBA (1998)	1430
G39	Lumber storage yard near S71	EBA (1998)	1450
G31	On bank alongside old Prelude Lake Hwy west of AC Kiln building west of G26	EBA (1998)	1460
S85	GM stack	EBA (1998)	1490
G29	On bank alongside old Prelude Lake Hwy west of AC Kiln building near raise house	EBA (1998)	1500
PSO-17-2200-01	Area 1 (Mill)	Golder (2001)	1600
S70	GM lumber yard	EBA (1998)	1600
G27	On bank alongside old Prelude Lake Hwy west of baghouse building	EBA (1998)	1600
PSO-14-2200-03	Area 1 (Mill)	Golder (2001)	1630
S69	GM fuel strg-lmbr yard	EBA (1998)	1640
S52	GM BC acrs from Crushr	EBA (1998)	1660
S82	GM stack	EBA (1998)	1680
S48	GM hoist-c shaft	EBA (1998)	1710
PSO-19-2200-01	Area 1 (Mill)	Golder (2001)	1720
S78	GM behind AC	EBA (1998)	1720
S41	GM near C dry	EBA (1998)	1720
PSO-18-2200-04	Area 1 (Mill)	Golder (2001)	1740
S12	GM gowganda yard	EBA (1998)	1770
PSO-15-2200-01	Area 1 (Mill)	Golder (2001)	1830
PSO-08-2200-03	Areas 2,3,4 (West of TPR)	Golder (2001)	1880

TABLE A1-2 (Cont'd)
MEASURED ARSENIC LEVELS IN GIANT MINE SITE SOIL

Giant Mine Property			
Soil Sample	**Location**	**Reference**	**Concentration (mg/kg (dw))**
PSO-43-0900-01	Areas 2,3,4 (West of TPR)	Golder (2001)	1890
PSO-13-2500-01	Area 1 (Mill)	Golder (2001)	1970
PSO-07-2200-02	Areas 2,3,4 (West of TPR)	Golder (2001)	1980
PSO-18-2200-02	Area 1 (Mill)	Golder (2001)	1990
G8	C dry oil storage tank inside berm on north side	EBA (1998)	2000
G5	Fuelling station across from C dry - obvious oil staining on ground	EBA (1998)	2000
S6	GM Near Townsite	EBA (1998)	2000
S45	GM Rail	EBA (1998)	2000
G28	On bank alongside old Prelude Lake Hwy west of roaster building	EBA (1998)	2000
S27	GM N pond to GSL	EBA (1998)	2020
G7	C dry oil storage tank outside berm on north side bear S40	EBA (1998)	2050
G50	Downstream of Dam #1 near site S24	EBA (1998)	2050
S88	GM stack	EBA (1998)	2100
G3	NW pond hazardous dump site near S32	EBA (1998)	2100
S86	GM stack	EBA (1998)	2120
29203	assay office	ESG (2000b)	2125
PSO-07-2200-03	Areas 2,3,4 (West of TPR)	Golder (2001)	2130
S57	GM MEG	EBA (1998)	2140
G6	30 feet west from site G5 outside of obvious staining	EBA (1998)	2200
PSO-43-0900-03	Areas 2,3,4 (West of TPR)	Golder (2001)	2210
PSO-14-2200-03	Area 1 (Mill)	Golder (2001)	2240
S35	GM near 22B Dam	EBA (1998)	2240
G9	Near site S39	EBA (1998)	2250
PSO-15-2200-03	Area 1 (Mill)	Golder (2001)	2260
29205	Reagent warehouse	ESG (2000b)	2278
S43	GM diesel fuel storage	EBA (1998)	2320
S72	GM lumber yard stack	EBA (1998)	2320

TABLE A1-2 (Cont'd)
MEASURED ARSENIC LEVELS IN GIANT MINE SITE SOIL

Giant Mine Property			
Soil Sample	**Location**	**Reference**	**Concentration (mg/kg (dw))**
S75	GM mill yard	EBA (1998)	2340
PSO-13-2500-01A	Area 1 (Mill)	Golder (2001)	2370
S2	GM Bhnd A2 Opn Pit	EBA (1998)	2380
PSO-11-2200-01	Area 1 (Mill)	Golder (2001)	2440
S39	GM main gate area	EBA (1998)	2440
PSO-43-0900-02	Areas 2,3,4 (West of TPR)	Golder (2001)	2480
PSO-11-2200-02	Area 1 (Mill)	Golder (2001)	2520
PSO-18-2200-04	Area 1 (Mill)	Golder (2001)	2540
PSO-44-0900-01	Areas 2,3,4 (West of TPR)	Golder (2001)	2590
PSO-44-0900-02	Areas 2,3,4 (West of TPR)	Golder (2001)	2620
S74	GM Quonset Hut	EBA (1998)	2680
G41	Approximately 100 yards south of the old PCB building near S19	EBA (1998)	2700
PSO-44-0900-03	Areas 2,3,4 (West of TPR)	Golder (2001)	2740
S73	GM Quonset Hut	EBA (1998)	2800
PSO-03-2200-01	Areas 2,3,4 (West of TPR)	Golder (2001)	2850
S42	GM diesel fuel storage	EBA (1998)	2880
S46	GM old elec shop area	EBA (1998)	2900
PSO-21-2500-02	Area 1 (Mill)	Golder (2001)	2910
PSO-07-2200-01	Areas 2,3,4 (West of TPR)	Golder (2001)	2920
PSO-37-2300-02	Area 8 (S. Tailings Impoundment)	Golder (2001)	2930
S30	GM ICG Tank Farm	EBA (1998)	2940
G24	15 feet east of cottrel building under stack flues	EBA (1998)	3100
S24	GM B3 opn pit area	EBA (1998)	3100
S19	GM old PCB strg bldg	EBA (1998)	3180
G22	100 feet north of baghouse near roadway	EBA (1998)	3200
S3	GM Near Yacht Club	EBA (1998)	3240
S60	GM Southwest mill	EBA (1998)	3260

TABLE A1-2 (Cont'd)
MEASURED ARSENIC LEVELS IN GIANT MINE SITE SOIL

Giant Mine Property			
Soil Sample	**Location**	**Reference**	**Concentration (mg/kg (dw))**
S56	GM BC- Crusher	EBA (1998)	3280
G20	Approximately 30 feet west of the base of the roaster stack towards HCD plant	EBA (1998)	3500
G34	Open pit Crusher - 50 feet south of transfer conveyor to headframe near S53	EBA (1998)	3500
G17	Roaster stack area near S87	EBA (1998)	3500
PSO-19-2200-02	Area 1 (Mill)	Golder (2001)	3640
PSO-21-2500-01A	Area 1 (Mill)	Golder (2001)	3650
PSO-12-2200-01	Area 1 (Mill)	Golder (2001)	3760
G23	20 feet north of baghouse	EBA (1998)	4000
PSO-21-2500-01	Area 1 (Mill)	Golder (2001)	4280
S40	GM near catchb C dry	EBA (1998)	4400
G4	NW pond hazardous dump site near S32	EBA (1998)	4500
G11	SE corner of C dry mechanical shopnear S46	EBA (1998)	4500
PSO-14-2200-02	Area 1 (Mill)	Golder (2001)	4890
S55	GM crusher	EBA (1998)	5040
29221	Next to main office parking lot	ESG (2000b)	5144
S71	GM lumber yard	EBA (1998)	5280
S64	GM fuel strg-lmbr yard	EBA (1998)	5360
29213	Under overhead conveyer	ESG (2000b)	5462
S53	GM open pit crusher	EBA (1998)	5680
G35	Open pit crusher - 20 feetsouth of open pit crusher building near S53	EBA (1998)	5700
S58	GM MEG	EBA (1998)	6000
PSO-14-2200-01	Area 1 (Mill)	Golder (2001)	6160
PSO-12-2200-02	Area 1 (Mill)	Golder (2001)	6880
G19	Approximately 30 feet north of the base of the roaster stack near the As silo	EBA (1998)	7700
S49	GM crusher	EBA (1998)	7920
G13	Northwest side of crushing plant near S54	EBA (1998)	8000
29208	Next to baghouse	ESG (2000b)	8158

TABLE A1-2 (Cont'd)
MEASURED ARSENIC LEVELS IN GIANT MINE SITE SOIL

Giant Mine Property			
Soil Sample	**Location**	**Reference**	**Concentration (mg/kg (dw))**
S87	GM stack	EBA (1998)	8400
PSO-37-2300-02	Area 8 (S. Tailings Impoundment)	Golder (2001)	8560
S54	GM open pit crusher	EBA (1998)	9760
PSO-14-2200-01	Area 1 (Mill)	Golder (2001)	9880
S63	GM assay lab	EBA (1998)	11800
PSO-12-2200-02	Area 1 (Mill)	Golder (2001)	14200
S83	GM stack	EBA (1998)	14200
G21	In swampy area immediately to the east of the roaster stack, near S83	EBA (1998)	15800
29211	East of Mill	ESG (2000b)	21500
29206	Next to Roaster	ESG (2000b)	87000

Note: from EBA Engineering Consultants Limited (1998).
mg/kg dw = parts per million, dry weight which is equivalent to µg/g (dw) or mg/kg (dw).

While human exposure to soils on the mine site is currently limited to mine workers, wildlife may be exposed for longer durations. To reduce human and wildlife exposure in the future, the site remediation provides for removal of soils containing arsenic concentrations greater than the Northwest Territories industrial criteria of 340 mg/kg arsenic. Currently, approximately 420 ha of the mine site is covered with soil containing less than 340 mg (As)/kg (soil), 26 ha is covered with soil containing greater than 340 mg (As)/kg (soil) and 95.6 ha is covered with tailings. For risk assessment purposes, the arsenic level on the Giant Mine site was set equal to the upper bound of 340 mg/kg. The mean value across the site after remediation would be expected to be considerable less than 340 mg/kg. Therefore, the use of an arsenic concentration of 340 mg/kg is believed to provide a conservative overestimate of the likely exposure for wildlife to arsenic from the inadvertent ingestion of soil.

TABLE A1-3
DERIVED PROBABILITY DISTRIBUTIONS FOR
SOIL CONCENTRATIONS BY RECEPTOR LOCATION (mg/kg)

Location	Summary of Measured Data					Lognormal Distribution Specifiers			
	# of Samples	Minimum	Maximum	Geo Mean	Geo Std Dev	GM	GSD	Minimum	Maximum
Latham Island	24	7	780	106	3.6	106	3.6	8.2	1374
Yellowknife City	282	2.5	1190	42	3.8	42	3.8	2.9	607
Dettah	8	7.2	144	26	3.1	26	3.1	2.7	250
Townsite	41	19	16600	353	6.7	340 [a]	-		
Giant Mine	226	5.2	87000	998	4.4	340 [a]	-	-	-

Note: GM – Geometric Mean
Geo Std Dev, GSD – Geometric Standard Deviation
Min and max for the lognormal distribution were set equal to 2 GSD from the GM
a - see discussion in text

A2.0 MEASURED ARSENIC LEVELS IN GARDEN PRODUCE AND SOILS

ESG (2001b) studied arsenic concentrations in soil and vegetables from gardens around the Yellowknife area. Table A2-1 presents all the soil concentrations measured in the gardens.

TABLE A2-1
MEASURED SOIL CONCENTRATIONS IN GARDENS AROUND YELLOWKNIFE

Soil Sample	Description	Total arsenic (ppm) assumed dw	Reference
Garden 1	S end Giant Mine town site	81	ESG 2001b
Garden 1	next to house Giant Mine town site	351	ESG 2001b
Garden 1	across from house Giant Mine town site	174	ESG 2001b
Garden 2	E end Range Lake	17	ESG 2001b
Garden 2	Center Range Lake	22	ESG 2001b
Garden 2	W end Range Lake	44	ESG 2001b
Garden 3	Rat Lake	24	ESG 2001b
Garden 4	Con Mine town site	55	ESG 2001b
Garden 5	Latham Island	30	ESG 2001b
Garden 6	Downtown YK	35	ESG 2001b
Garden 7	Range Lake	29	ESG 2001b
Garden 8	Rat Lake	27	ESG 2001b
Garden 9	Old Town YK	12	ESG 2001b
Garden 10	Latham Island	56	ESG 2001b

Note: mg/kg dw = parts per million dry weight which is equivalent to µg/g (dw) or mg/kg (dw).

Table A2-2 presents the measured arsenic concentrations in vegetables from the gardens included in ESG (2001b) for above-ground vegetables. If the above-ground portion of a below-ground vegetable (i.e., beet greens) was measured, then the results are included in Table A2-2. Table A2-3 presents the arsenic concentrations measured in below-ground vegetables from ESG (2001b).

TABLE A2-2
MEASURED ABOVE-GROUND VEGETABLE CONCENTRATIONS IN GARDENS AROUND YELLOWKNIFE

Above-Ground Vegetation			Total arsenic (mg/kg ww)	Reference
Garden 1	Beet greens	*Beta vulgaris var. crassa*	0.29	ESG 2001b
Garden 1	Rhubarb Stalks	*Rheum rhababarum*	0.05	ESG 2001b
Garden 1	Saskatoon berries	*Amelanchier alnifolia*	0.44	ESG 2001b
Garden 2	Beet greens	*Beta vulgaris var. crassa*	0.18	ESG 2001b
Garden 2	Onion tops	*Allium cepa*	0.15	ESG 2001b
Garden 2	Leaf lettuce	*Lactuca sativa*	0.06	ESG 2001b
Garden 2	Romaine lettuce	*Lactuca sativa*	0.13	ESG 2001b
Garden 2	Celery leaves	*Apium graveolens var. dulce*	0.29	ESG 2001b
Garden 2	Celery	*Apium graveolens var. dulce*	0.05	ESG 2001b
Garden 2	Rhubarb Stalks	*Rhem rhababarum*	0.014	ESG 2001b
Garden 2	Peas	*Pisum sativum*	< 0.02	ESG 2001b
Garden 2	Beans	*Phaseolus vulgaris*	0.016	ESG 2001b
Garden 3	Beet greens	*Beta vulgaris var. crassa*	0.13	ESG 2001b
Garden 3	Leaf lettuce	*Lactuca sativa*	0.27	ESG 2001b
Garden 3	Romaine lettuce	*Lactuca sativa*	0.12	ESG 2001b
Garden 3	Tomatoes	*Lycopersicon esculentum*	0.009	ESG 2001b
Garden 4	Kohlrabi	*Brassica oleracea var. gongylodes*	0.044	ESG 2001b
Garden 4	Red cabbage	*Brassica oleracea var. capitata*	0.09	ESG 2001b
Garden 4	Kale	*Brassica oleracea var. acephala*	0.16	ESG 2001b
Garden 4	Peas	*Pisum sativum*	< 0.02	ESG 2001b
Garden 4	Broad beans	*Phaseolus vulgaris*	0.018	ESG 2001b
Garden 5	Swiss chard	*Beta vulgaris*	0.09	ESG 2001b
Garden 5	Italian parsley	*Petroselinum crispum var. neapolitanum*	0.1	ESG 2001b
Garden 5	Oregano	*Origanum sp.*	0.23	ESG 2001b
Garden 5	Rhubarb stalks	*Rheum rhababarum*	< 0.01	ESG 2001b
Garden 8	Beet greens	*Beta vulgaris var. crassa*	0.1	ESG 2001b

TABLE A2-2 (Cont'd)
MEASURED ABOVE-GROUND VEGETABLE CONCENTRATIONS IN GARDENS AROUND YELLOWKNIFE

Above-Ground Vegetation			Total arsenic (mg/kg ww)	Reference
Garden 8	Onion tops	*Allium cepa*	0.18	ESG 2001b
Garden 8	Garlic tops	*Allium sativum*	0.11	ESG 2001b
Garden 8	White cabbage	*Brassica oleracea var. capitata*	0.033	ESG 2001b
Garden 8	Dill	*Anethum graveolens*	0.07	ESG 2001b
Garden 8	Swiss chard	*Beta vulgaris*	0.06	ESG 2001b
Garden 8	Rhubarb Stalks	*Rheum rhababarum*	0.015	ESG 2001b
Garden 8	Pin cherries	*Prunus pensylvanica*	0.09	ESG 2001b
Garden 8	Saskatoon berries	*Amelanchier alnifolia*	0.15	ESG 2001b
Garden 9	White cabbage	*Brassica oleracea var. capitata*	< 0.01	ESG 2001b
Garden 9	Broccoli	*Brassica oleracea cymosa*	< 0.02	ESG 2001b
Garden 9	Peas	*Pisum sativum*	0.036	ESG 2001b
Garden 9	Zucchini	*Curcurbita pepo*	< 0.005	ESG 2001b
Garden 10	Leaf lettuce	*Lactuca sativa*	0.08	ESG 2001b
Garden 10	Rhubarb stalks	*Rheum rhababarum*	0.014	ESG 2001b
Garden 10	Beans	*Phaseolus vulgaris*	0.026	ESG 2001b

Note: mg/kg ww = parts per million, wet weight which is equivalent to µg/g (ww) or mg/kg (ww).

TABLE A2-3
MEASURED BELOW-GROUND VEGETABLE CONCENTRATIONS IN GARDENS AROUND YELLOWKNIFE

Below-Ground Vegetation			Total arsenic (mg/kg ww)	Reference
Garden 1	Radish	*Raphanus sativus*	0.17	ESG 2001b
Garden 2	Carrots	*Daucus carota*	0.034	ESG 2001b
Garden 2	Red potatoes	*Solanum tuberosum*	0.034	ESG 2001b
Garden 2	Onion (peeled)	*Allium cepa*	0.041	ESG 2001b
Garden 2	Beets	*Beta vulgaris var. crassa*	0.02	ESG 2001b
Garden 3	Beets	*Beta vulgaris var. crassa*	0.19	ESG 2001b
Garden 4	Red potatoes	*Solanum tuberosum*	0.026	ESG 2001b
Garden 5	Carrots	*Daucus carota*	0.05	ESG 2001b
Garden 6	Carrots	*Daucus carota*	0.06	ESG 2001b

TABLE A2-3 (Cont'd)
MEASURED BELOW-GROUND VEGETABLE CONCENTRATIONS IN GARDENS AROUND YELLOWKNIFE

Below-Ground Vegetation			Total arsenic (mg/kg ww)	Reference
Garden 6	White potatoes	*Solanum tuberosum*	< 0.03	ESG 2001b
Garden 7	Carrots	*Daucus carota*	0.037	ESG 2001b
Garden 7	White potatoes	*Solanum tuberosum*	< 0.02	ESG 2001b
Garden 8	Carrots	*Daucus carota*	0.02	ESG 2001b
Garden 8	Red potatoes	*Solanum tuberosum*	< 0.02	ESG 2001b
Garden 8	Onion (peeled)	*Allium cepa*	0.017	ESG 2001b
Garden 8	Garlic bulb	*Allium sativum*	< 0.03	ESG 2001b
Garden 8	Beets	*Beta vulgaris var. crassa*	0.034	ESG 2001b
Garden 9	Red potatoes	*Solanum tuberosum*	0.02	ESG 2001b
Garden 10	Carrots	*Daucus carota*	0.07	ESG 2001b
Garden 10	White potatoes	*Solanum tuberosum*	0.07	ESG 2001b
Garden 10	Red potatoes	*Solanum tuberosum*	0.06	ESG 2001b

Note: mg/kg ww = parts per million, wet weight which is equivalent to µg/g (ww) or mg/kg (ww).

Above- and below-ground vegetable concentrations can be input directly into the pathways model to estimate exposures to people living at these sites. Vegetable concentrations measured in gardens around the Yellowknife area were found to be statistically similar, with the exception of Garden 1 (Old Giant Mine Townsite) and 3 (Rat Lake area). Therefore, for all receptor locations except the Old Giant Mine Townsite, a beta distribution was assumed to characterize the above- and below-ground vegetable concentrations. For the Old Giant Mine Townsite a lognormal distribution was used. These distributions are shown in Table A2-4.

TABLE A2-4
DERIVED PROBABILITY DISTRIBUTIONS FOR ABOVE- AND BELOW-GROUND VEGETATION CONCENTRATIONS

	Average (ppm ww)	Distribution
Receptor Locations Except Old Giant Mine Townsite		
Above-ground vegetation	0.077	β(0.0, 0.515, 0.836, 4.735)
Below-ground vegetation	0.034	β(0.0, 0.134, 1.923, 5.670)
Receptors at the Old Giant Mine Townsite		
Above-ground vegetation	0.55	LN(0.28,3.2,0.05,5.6)
Below-ground vegetation	0.24	LN(0.19,1.95,0.05,0.72)

Note: Beta distribution -β(minimum, maximum, alpha, beta).
Lognormal distribution – LN(geometric mean, geometric standard deviation, minimum, maximum).

Site-specific transfer factors were developed using the data from the garden survey (ESG 2001b). Data from two gardens were not included in the analysis because of statistically different arsenic in soil concentrations (Garden 1 at the Old Giant Mine townsite) or vegetable concentrations (Garden 3 in the Rat Lake area).

The transfer factor for above-ground vegetables was used to model the concentration in browse and summer forage. Transfer factors were calculated assuming soil is the principal source of arsenic using the following equation (A-1):

$$TF_{soil-to-veg} = \frac{C_{veg}}{C_{soil}} \qquad \text{(A-1)}$$

where:

$TF_{soil\text{-}to\text{-}veg}$ = arsenic transfer factor from soil-to-vegetation ((mg/kg ww)/(mg/kg dw))
C_{veg} = concentration of vegetation in wet (fresh) weight (mg/kg ww)
C_{soil} = concentration of soil in dry weight (mg/kg dw)

For the calculation of site-specific transfer factors from the ESG (2001b) data, samples collected from Gardens 1 and 3 were not considered. However, the transfer factors derived for all other gardens were grouped together. A summary of these data is presented in Table A2-5.

TABLE A2-5
DERIVED SOIL-TO-GARDEN VEGETATION
TRANSFER FACTORS FOR ARSENIC

		Beta Distribution Specifiers			
Terrestrial vegetation	**# of Samples**	**Minimum**	**Maximum**	**Alpha**	**Beta**
Above-ground	34	0.0	0.018	0.772	4.634
Below-ground	19	0.0	0.003	2.821	6.040

Note: Gardens 1 and 3 not included.

A3.0 MEASURED ARSENIC LEVELS IN TERRESTRIAL VEGETATION

Arsenic level measurements on terrestrial mosses, mushrooms and lichens collected from the Con Mine and Giant Mine site have been reported by Koch *et al.* (2000). Results of the individual measurements are provided on Table A3-1 attached; summary statistics are presented on Table A3-2 below.

TABLE A3-1
MEASURED ARSENIC LEVELS IN TERRESTRIAL VEGETATION

Yellowknife Area			
Common Name	**Scientific Name**	**Concentration (mg/kg dw)**	**Reference**
not specified	-	nd to 3	background levels, Koch *et al.* (2000)
Birch	*Betula* sp.	526	Hutchinson *et al.* (1982)
Black Spruce	*Picea mariana*	8.7	washed foliage, Hutchinson *et al.* (1982)
Black Spruce	*Picea mariana*	8.9	unwashed foliage, Hutchinson *et al.* (1982)
Black Spruce	*Picea mariana*	101	washed foliage, Hutchinson *et al.* (1982)
Black Spruce	*Picea mariana*	123	unwashed foliage, Hutchinson *et al.* (1982)
Grass	not specified	12.6 to 89.4	O'Toole *et al.* (1972)
Moss	*Drepanocladus* sp.	490	Koch *et al.* (2000)
Moss	*Drepanocladus* sp.	880	Koch *et al.* (2000)
Moss	*Drepanocladus* sp.	1220	Koch *et al.* (2000)
Moss	*Fumaria hygrometrica*	1130	Koch *et al.* (2000)
Moss	*Fumaria hygrometrica*	1900	Koch *et al.* (2000)
Moss	*Pohlia* sp.	1310	Koch *et al.* (2000)
Moss	*Drepanocladus* sp.	770	Koch *et al.* (2000)
Lichen	*Cladonia* sp.	14.3	Koch *et al.* (2000)
Lichen	*Cladonia* sp.	29	Koch *et al.* (2000)
Lichen	*Cladonia* sp.	520	Koch *et al.* (2000)
Lichen	*Cladonia* sp.	38	Koch *et al.* (2000)
Lichen	*Cladonia* sp.	49	Koch *et al.* (2000)
Lichen	*Cladonia* sp.	55	Koch *et al.* (2000)
Lichen	*Cladonia* sp.	2300	Koch *et al.* (2000)
Rose bushes	not specified	15.4 to 138	O'Toole *et al.* (1972)

TABLE A3-1 (Cont'd)
MEASURED ARSENIC LEVELS IN TERRESTRIAL VEGETATION

Con Mine Area			
Common Name	**Scientific Name**	**Concentration (mg/kg dw)**	**Reference**
Grasses and shrubs	not specified	53	median, Koch *et al.* (2000)
Moss	*Drepanocladus* sp.	770	Koch *et al.* (2000)
Red raspberry	*Rubus idaeus*	83.6	Koch *et al.* (2000)
Squirreltail Barley	*Hordeum jubatum*	3.6	Koch *et al.* (2000)
Squirreltail Barley	*Hordeum jubatum*	8.0	Koch *et al.* (2000)
Squirreltail Barley	*Hordeum jubatum*	74.1	Koch *et al.* (2000)

Note: mg/kg dw = parts per million, dry weight, which is equivalent to µg/g (dw) or mg/kg (dw).

The mosses generally contain the highest levels of arsenic, although the highest individual measurement was recorded on a lichen sample. The terrestrial moss observed from the Giant Mine Tailings Pond was found to have the highest level of arsenic. It was noted that the lichen sample collected at the same location displayed the highest arsenic level of all lichen samples. This is the same region where the mushroom, *Lycoperdon* sp., was found to have the highest level of arsenic out of all mushrooms.

TABLE A3-2
SUMMARY STATISTICS OF TERRESTRIAL VEGETATION ARSENIC LEVELS (mg/kg (dw))

Terrestrial Vegetation	# of Samples	Minimum	Maximum	Geo Mean	Average	Std. Dev.
Moss	7	490	1900	1018.8	1100	452
Lichen	9	6.4	2300	55.7	336.4	754
Mushroom	5	8.3	1010	70	295.6	434

Additional sampling on the contaminant levels in mushrooms in the Northwest Territories has been completed and was reported on by Obst *et al.* (2000). Samples were collected from an area of 50 km x 200 km centered around the City of Yellowknife. The total arsenic levels in fungi measured ranged from 0.2 to 494.0 mg/kg dry weight (dw). Arsenic speciation indicated that in two samples of fungi, organic arsenic compounds represented more than 99.75% of total arsenic.

Arsenic speciation measurements carried out by Koch *et al.* (2000) demonstrated that the mosses, contain only inorganic arsenic (As(III) and As(V)) with exception to one species, *Drepanocladus* sp. The lichens mostly comprised of As(III) and As(V) together making up 62 to 93% of the total extracted arsenic. Arsenobetaine was found for the first time in lichens. Arsenobetaine existed in the mushrooms amounting to 62 to 88% of the total arsenic extracted.

Measured arsenic concentrations in berries were also available for the Yellowknife area. A summary of the relevant data is presented in Table A3-3. When available for a receptor location, site-specific berry concentrations were used in the pathways analysis. In the absence of data for a receptor location, berry concentrations were calculated based on the air concentration. Table A3-4 presents the probability distributions derived from the berry concentrations at each receptor location from the data summarized in Table A3-3.

TABLE A3-3
MEASURED ARSENIC LEVELS IN BERRIES (mg/kg wet weight)

Area	# of samples	Minimum (mg/kg)	Maximum (mg/kg)	Geo Mean (mg/kg)	Geo Std Dev	Avg (mg/kg)	Avg Std Dev	Berries
Yellowknife City	7	0.024	0.203	0.083	1.97	0.099	0.059	Raspberry, gooseberry, cranberry
Giant Mine	6	0.050	1.913	0.284	3.3	0.520	0.696	Raspberry, gooseberry, cranberry, rose hip
Joliffe Island (S. of Latham Island)	3	0.077	0.161	0.116	1.32	0.122	0.042	Raspberry, gooseberry, blueberry
Dettah Road	2	0.039	0.040	0.039	1.01	0.040	0.001	Gooseberry, rose hip

Note: The data presented in this table is summarized from Davey *et al.* 1998 and associated papers.

TABLE A3-4
DERIVED PROBABILITY DISTRIBUTIONS FOR BERRY CONCENTRATIONS BY RECEPTOR LOCATION (mg/kg (ww))

Location	Summary of Measured Data					Lognormal Distribution Specifiers			
	# of samples	Minimum	Maximum	Geo Mean	Geo Std Dev	GM	GSD	Minimum	Maximum
Latham Island	3	0.077	0.161	0.116	1.5	0.116	1.5	0.052	0.26
Yellowknife City	7	0.024	0.203	0.083	2.0	0.083	2.0	0.021	0.33
Dettah	2	0.039	0.040	0.039	1.0	0.039	2.0 *	0.010	0.16
Giant Mine	6	0.050	1.913	0.284	3.3	0.284	3.3	0.026	3.09

Note: GM – Geometric Mean.
Geo Std Dev, GSD – Geometric Standard Deviation.
* The GSD for Yellowknife City was used for the Dettah Road data.
Min and max for the lognormal distribution were set equal to 2 GSD from the GM.

A4.0 MEASURED ARSENIC LEVELS IN AQUATIC VEGETATION

A range of 150 to 3700 ppm arsenic in aquatic plants was reported for Kam, Grace, Keg, Likely and Chitty Lakes (Wagemann *et al.* 1978, cited by RMCC 1999). Koch *et al.* (2000) reported that submergent plants (e.g. *Myriophyllum* sp.) collected during an August sampling campaign predominantly contained arsenic in the form of arsenate (As(V)). Arsenate is the predominant form of arsenic in Yellowknife waters. Table A4-1 presents the measured data reported by Koch *et al.* (2000) and Dillon (2002a).

Site-specific transfer factors for aquatic vegetation were calculated using data presented in Dillon (2002a). Water concentrations measured downstream of Giant Mine (Dillon 2002a) were not typical of average exposure (0.06 mg/L measured by Dillon (2002a) vs. 0.18 mg/L from 1999 data). Therefore, for the calculation of site-specific transfer factors for water-to-aquatic vegetation, an average water concentration downstream of the Giant Mine site was substituted for the levels measured by Dillon (2002a). Upstream water and aquatic vegetation concentrations were not considered. Transfer factors were calculated following equation (A-2):

$$TF_{water-to-aq\ veg} = \frac{C_{aq\ veg} \times (1 - mc)}{C_{water}} \qquad \text{(A-2)}$$

where:

$TF_{water\text{-}to\text{-}fish}$ = water-to-aquatic vegetation transfer factor for arsenic [m^3/g (ww) = L/kg (ww)]
$C_{aq\ veg}$ = arsenic concentration in aquatic vegetation [mg/kg (dw)]
mc = assumed moisture content [0.8] to convert from (dw) to (ww)
C_{water} = arsenic concentration in water [0.178 mg/L]

A summary of site-specific transfer factors calculated for water-to-aquatic vegetation is provided in Table A4-2.

TABLE A4-1
MEASURED ARSENIC LEVELS IN AQUATIC VEGETATION

Yellowknife Area				
Common Name	**Scientific Name**	**Conc. (mg/kg dw)**	**Water Body**	**Reference**
Bur reed	*Sparganium augustifolium*	2.5	Yellowknife Bay	Koch *et al.* (2000)
Common cattail	*Typha latifolia*	0.52	Niven Lake	Koch *et al.* (2000)
Common cattail	*Typha latifolia*	5	Baker Creek	Koch *et al.* (2000)
Duckweed	*Lemna minor*	28	Niven Lake	Koch *et al.* (2000)
Eurasian watermilfoil	*Myriophyllum sp.*	17.4	Niven Lake	Koch *et al.* (2000)
Eurasian watermilfoil	*Myriophyllum* sp.	39	Niven Lake	Koch *et al.* (2000)
Eurasian watermilfoil	*Myriophyllum* sp.	78	Niven Lake	Koch *et al.* (2000)
Nodding bur marigold or Nodding beggar-ticks	*Bidens cernua*	100	Niven Lake	Koch *et al.* (2000)
Richardson's pondweed	*Potamogetan richardsonii*	20	Yellowknife Bay	Koch *et al.* (2000)
Scouring rush, or rough horsetail	*Equisetum hyemale*	65.5	Baker Creek, downstream	Dillon (2002a)
Scouring rush, or rough horsetail	*Equisetum hyemale*	74.2	Baker Creek, downstream	Dillon (2002a)
Scouring rush, or rough horsetail	*Equisetum hyemale*	55.5	Baker Creek, downstream	Dillon (2002a)
Scouring rush, or rough horsetail	*Equisetum hyemale*	85.1	Baker Creek, downstream	Dillon (2002a)
Scouring rush, or rough horsetail	*Equisetum hyemale*	17.6	Baker Creek, downstream	Dillon (2002a)
Scouring rush, or rough horsetail	*Equisetum hyemale*	76.3	Baker Creek, downstream	Dillon (2002a)
Water horsetail	*Equisetum fluviatile*	48	Yellowknife Bay at Baker Creek Outlet	Koch *et al.* (2000)

TABLE A4-1 (Cont'd)
MEASURED ARSENIC LEVELS IN AQUATIC VEGETATION

Con Mine Area				
Common Name	**Scientific Name**	**Conc. (mg/kg dw)**	**Water Body**	**Reference**
Water horsetail	*Equisetum fluviatile*	260	Baker Creek	Koch *et al.* (2000)
Water sedge	*Carex aquatilis*	40	Baker Creek, upstream	Dillon (2002a)
Water sedge	*Carex aquatilis*	98.6	Baker Creek, upstream	Dillon (2002a)
Water sedge	*Carex aquatilis*	17.6	Baker Creek, upstream	Dillon (2002a)
Water sedge	*Carex aquatilis*	103	Baker Creek, upstream	Dillon (2002a)
Water sedge	*Carex aquatilis*	61.8	Baker Creek, upstream	Dillon (2002a)
Water sedge	*Carex aquatilis*	32.4	Baker Creek, upstream	Dillon (2002a)
Bur reed	*Sparganium sp.*	28	ConMine Lakes	Dushenko *et al.* (1995)
Bur reed	*Sparganium sp.*	133	ConMine Lakes	Dushenko *et al.* (1995)
Common cattail	*Typha latifolia*	3.8	ConMine Lakes	Koch *et al.* (2000)
Common cattail (shoots)	*Typha latifolia*	17.2	ConMine Lakes	Dushenko *et al.* (1995)
Common cattail (roots)	*Typha latifolia*	232	ConMine Lakes	Dushenko *et al.* (1995)
Eurasian watermilfoil	*Myriophyllum exalbescens*	143	ConMine Lakes	Dushenko *et al.* (1995)
Marsh arrowgrass (shoots)	*Triglochin palustre*	40	ConMine Lakes	Dushenko *et al.* (1995)
Marsh arrowgrass (roots)	*Triglochin palustre*	470	ConMine Lakes	Dushenko *et al.* (1995)
Sago pondweed (roots)	*Potamogetan pectinatus*	592	ConMine Lakes	Dushenko *et al.* (1995)
Sago pondweed (shoots)	*Potamogetan pectinatus*	751	ConMine Lakes	Dushenko *et al.* (1995)
Sago pondweed	*Potamogetan pectinatus*	1219	ConMine Lakes	Dushenko *et al.* (1995)
Sedges	*Carex sp.*	3.9	ConMine Lakes	Koch *et al.* (2000)
Sedges	*Carex sp.*	8.1	ConMine Lakes	Koch *et al.* (2000)

TABLE A4-1 (Cont'd)
MEASURED ARSENIC LEVELS IN AQUATIC VEGETATION

Con Mine Area				
Common Name	**Scientific Name**	**Conc. (mg/kg dw)**	**Water Body**	**Reference**
Sedges	*Carex sp.*	8.2	ConMine Lakes	Koch *et al.* (2000)
Sedges	*Carex sp.*	8.5	ConMine Lakes	Koch *et al.* (2000)
Sedges	*Carex sp.*	9.4	ConMine Lakes	Koch *et al.* (2000)
Sedges	*Carex sp.*	17.0	ConMine Lakes	Koch *et al.* (2000)
Sedges	*Carex sp.*	22.3	ConMine Lakes	Koch *et al.* (2000)
Sedges	*Carex sp.*	23.4	ConMine Lakes	Koch *et al.* (2000)
Sedges	*Carex sp.*	23.9	ConMine Lakes	Koch *et al.* (2000)
Sedges	*Carex sp.*	25.2	ConMine Lakes	Koch *et al.* (2000)
Sedges	*Carex sp.*	28.4	ConMine Lakes	Koch *et al.* (2000)
Sedges	*Carex sp.*	30.9	ConMine Lakes	Koch *et al.* (2000)
Sedges	*Carex sp.*	31.3	ConMine Lakes	Koch *et al.* (2000)
Sedges	*Carex sp.*	35.3	ConMine Lakes	Koch *et al.* (2000)
Sedges	*Carex sp.*	40.3	ConMine Lakes	Koch *et al.* (2000)
Sedges	*Carex sp.*	45.9	ConMine Lakes	Koch *et al.* (2000)
Sedges	*Carex sp.*	57.6	ConMine Lakes	Koch *et al.* (2000)
Sedges	*Carex sp.*	135.9	ConMine Lakes	Koch *et al.* (2000)
Shrubby Cinquefoil	*Potentilla fruticosa*	11.9	ConMine Lakes	Koch *et al.* (2000)
Ticklegrass	*Agrostis scabra*	53.1	ConMine Lakes	Koch *et al.* (2000)
Ticklegrass	*Agrostis scabra*	71.9	ConMine Lakes	Koch *et al.* (2000)
Water horsetail	*Equisetum fluviatile*	30	ConMine Lakes	Koch *et al.* (2000)
Water horsetail (shoots)	*Equisetum fluviatile*	34	ConMine Lakes	Dushenko *et al.* (1995)
Water horsetail (roots)	*Equisetum fluviatile*	352	ConMine Lakes	Dushenko *et al.* (1995)

Note: mg/kg dw = parts per million dry weight which is equivalent to µg/g (dw) or mg/kg (dw).

TABLE A4-2
DERIVED WATER-TO-AQUATIC VEGETATION TRANSFER FACTORS FOR ARSENIC (L/kg (ww))

	Summary of Measured Data					Lognormal Distribution Specifiers			
	# of Samples	**Minimum**	**Maximum**	**Geo Mean**	**Geo Std Dev**	**GM**	**GSD**	**Minimum**	**Maximum**
Aquatic vegetation	6	20	95	63	1.8	63	1.8	19.4	204.1

Note: GM – Geometric Mean.
Geo Std Dev, GSD – Geometric Standard Deviation.
The min and max for the lognormal distribution were set 2 GSD from the GM.

A5.0 MEASURED ARSENIC LEVELS IN FISH

Measured concentrations in fish were available for a number of different water bodies. Also, the arsenic concentrations in liver, kidney, and muscle were provided in the literature. Values presented for Baker Creek and Yellowknife Bay in Table A5-1 from Falk *et al.* (1973) were obtained in 1972. Values from Yellowknife Bay and Back Bay from Jackson *et al.* (1996) are from 1992-93. Resolution Bay data in Table A5-1 were collected in 1996 and are for muscle only. From Table A5-1 it can be seen that measured arsenic concentrations in muscle, liver and kidney of fish are generally consistent and that arsenic does not tend to accumulate preferentially in only one organ. For the pathways assessment therefore, it is assumed that whole body fish concentrations of arsenic are equivalent to the concentration in muscle only.

Site-specific transfer factors were calculated for fish using the data presented in Jackson *et al.* (1996). Transfer factors were calculated following equation (A-3):

$$TF_{water-to-fish} = \frac{C_{fish}}{C_{water}} \times \frac{1}{1000} \qquad \text{(A-3)}$$

where:

$TF_{water\text{-}to\text{-}fish}$	=	water-to-fish transfer factor for arsenic [m^3/g (ww)]
C_{fish}	=	arsenic concentration in fish [µg/g (ww)]
C_{water}	=	arsenic concentration in water [µg/L]
1000	=	conversion factor [L/m^3]

A summary of the derived water-to-fish transfer factors for arsenic from the data in Jackson *et al.* (1996) is presented in Table A5-2. The model used for this assessment considers only transfer factors from water-to-fish muscle.

Subsequent to the derivation of a transfer factor for fish based on Jackson *et al.* (1996) additional fish concentration data was received. The recent work conducted by the University of Saskatchewan (de Rosemond *et al.* 2004) showed similar levels in arsenic in fish that that observed by Jackson *et al.* (1996) and thus a re-evaluation of the transfer factor was not undertaken.

TABLE A5-1
MEASURED ARSENIC LEVELS IN FISH TISSUE

Baker Creek				
Fish	**Muscle Concentration (mg/kg (ww))**	**Liver Concentration (mg/kg (ww))**	**Kidney Concentration (mg/kg (ww))**	**Reference**
Long nose sucker	0.21	-	-	Falk *et al.* (1973)
Long nose sucker	0.26	-	-	Falk *et al.* (1973)
Long nose sucker	0.32	-	-	Falk *et al.* (1973)
Long nose sucker	0.32	-	-	Falk *et al.* (1973)
Northern pike	0.22	0.60	-	Falk *et al.* (1973)
Northern pike	0.33	-	-	Falk *et al.* (1973)
Northern pike	0.33	0.02	-	Falk *et al.* (1973)
Northern pike	0.37	0.87	-	Falk *et al.* (1973)
Whitefish	0.22	0.87	-	Falk *et al.* (1973)
Yellowknife Bay / Back Bay				
Fish	**Muscle Concentration (mg/kg (ww))**	**Liver Concentration (mg/kg (ww))**	**Kidney Concentration (mg/kg (ww))**	**Reference**
Northern Pike	0.03	0.05	0.06	Jackson *et al.* (1996)
Northern Pike	0.07	0.07	0.14	Jackson *et al.* (1996)
Northern Pike	0.07	0.27	0.14	Jackson *et al.* (1996)
Northern Pike	0.08	0.05	-	Jackson *et al.* (1996)
Northern Pike	0.08	0.1	-	Jackson *et al.* (1996)
Northern Pike	0.09	0.42	0.41	Jackson *et al.* (1996)
Northern Pike	0.09	0.55	0.26	Jackson *et al.* (1996)
Northern Pike	0.09	0.07	0.24	Jackson *et al.* (1996)
Northern Pike	0.09	0.09	0.05	Jackson *et al.* (1996)
Northern Pike	0.09	0.16	-	Jackson *et al.* (1996)
Northern Pike	0.09	0.1	0.09	Jackson *et al.* (1996)
Northern Pike	0.01	0.13	-	Jackson *et al.* (1996)
Northern Pike	0.11	0.07	0.09	Jackson *et al.* (1996)
Northern Pike	0.11	0.15	-	Jackson *et al.* (1996)
Northern Pike	0.11	0.11	-	Jackson *et al.* (1996)
Northern Pike	0.11	0.35	-	Jackson *et al.* (1996)

TABLE A5-1 (Cont'd)
MEASURED ARSENIC LEVELS IN FISH TISSUE

Yellowknife Bay / Back Bay				
Fish	**Muscle Concentration (mg/kg (ww))**	**Liver Concentration (mg/kg (ww))**	**Kidney Concentration (mg/kg (ww))**	**Reference**
Northern Pike	0.11	0.15	-	Jackson *et al.* (1996)
Northern Pike	0.11	0.12	0.13	Jackson *et al.* (1996)
Northern Pike	0.12	0.1	0.6	Jackson *et al.* (1996)
Northern Pike	0.12	0.19	-	Jackson *et al.* (1996)
Northern Pike	0.12	0.09	0.15	Jackson *et al.* (1996)
Northern Pike	0.12	0.08	-	Jackson *et al.* (1996)
Northern Pike	0.12	0.2	-	Jackson *et al.* (1996)
Northern Pike	0.12	0.15	-	Jackson *et al.* (1996)
Northern Pike	0.12	0.19	-	Jackson *et al.* (1996)
Northern Pike	0.12	0.2	-	Jackson *et al.* (1996)
Northern Pike	0.12	0.08	0.18	Jackson *et al.* (1996)
Northern Pike	0.12	0.64	0.09	Jackson *et al.* (1996)
Northern Pike	0.12	0.16	0.2	Jackson *et al.* (1996)
Northern Pike	0.13	0.69	0.71	Jackson *et al.* (1996)
Northern Pike	0.13	0.19	-	Jackson *et al.* (1996)
Northern Pike	0.13	0.13	-	Jackson *et al.* (1996)
Northern Pike	0.13	0.22	-	Jackson *et al.* (1996)
Northern Pike	0.14	0.15	0.27	Jackson *et al.* (1996)
Northern Pike	0.14	0.2	0.18	Jackson *et al.* (1996)
Northern Pike	0.14	0.05	0.15	Jackson *et al.* (1996)
Northern Pike	0.14	0.27	-	Jackson *et al.* (1996)
Northern Pike	0.14	0.14	-	Jackson *et al.* (1996)
Northern Pike	0.14	0.2	0.18	Jackson *et al.* (1996)
Northern Pike	0.14	0.64	0.69	Jackson *et al.* (1996)
Northern Pike	0.15	0.85	0.48	Jackson *et al.* (1996)
Northern Pike	0.15	0.38	0.37	Jackson *et al.* (1996)
Northern Pike	0.15	0.18	0.26	Jackson *et al.* (1996)
Northern Pike	0.15	0.19	-	Jackson *et al.* (1996)
Northern Pike	0.15	0.22	-	Jackson *et al.* (1996)
Northern Pike	0.15	0.25	0.26	Jackson *et al.* (1996)

TABLE A5-1 (Cont'd)
MEASURED ARSENIC LEVELS IN FISH TISSUE

Yellowknife Bay / Back Bay				
Fish	**Muscle Concentration (mg/kg (ww))**	**Liver Concentration (mg/kg (ww))**	**Kidney Concentration (mg/kg (ww))**	**Reference**
Northern Pike	0.15	0.73	0.14	Jackson *et al.* (1996)
Northern Pike	0.15	0.23	0.15	Jackson *et al.* (1996)
Northern Pike	0.16	0.45	0.73	Jackson *et al.* (1996)
Northern Pike	0.16	0.1	-	Jackson *et al.* (1996)
Northern Pike	0.16	0.47	0.14	Jackson *et al.* (1996)
Northern Pike	0.16	0.13	0.22	Jackson *et al.* (1996)
Northern Pike	0.16	0.09	0.09	Jackson *et al.* (1996)
Northern Pike	0.17	0.09	0.18	Jackson *et al.* (1996)
Northern Pike	0.17	0.23	-	Jackson *et al.* (1996)
Northern Pike	0.17	0.16	0.09	Jackson *et al.* (1996)
Northern Pike	0.18	0.18	0.21	Jackson *et al.* (1996)
Northern Pike	0.18	0.07	0.22	Jackson *et al.* (1996)
Northern Pike	0.18	0.07	-	Jackson *et al.* (1996)
Northern Pike	0.18	0.23	-	Jackson *et al.* (1996)
Northern Pike	0.18	0.54	0.25	Jackson *et al.* (1996)
Northern Pike	0.18	0.13	0.12	Jackson *et al.* (1996)
Northern Pike	0.19	0.08	0.27	Jackson *et al.* (1996)
Northern Pike	0.19	0.27	0.18	Jackson *et al.* (1996)
Northern Pike	0.19	0.26	-	Jackson *et al.* (1996)
Northern Pike	0.19	0.14	0.11	Jackson *et al.* (1996)
Northern Pike	0.19	0.24	0.1	Jackson *et al.* (1996)
Northern Pike	0.2	0.46	0.4	Jackson *et al.* (1996)
Northern Pike	0.2	0.45	-	Jackson *et al.* (1996)
Northern Pike	-	0.73	-	Falk *et al.* (1973)
Northern Pike	0.2	1.08	0.8	Jackson *et al.* (1996)
Northern Pike	0.2	0.09	0.09	Jackson *et al.* (1996)
Northern Pike	0.21	0.13	0.19	Jackson *et al.* (1996)
Northern Pike	0.21	0.13	0.21	Jackson *et al.* (1996)
Northern Pike	0.21	0.13	0.3	Jackson *et al.* (1996)
Northern Pike	0.21	0.16	0.1	Jackson *et al.* (1996)

TABLE A5-1 (Cont'd)
MEASURED ARSENIC LEVELS IN FISH TISSUE

Yellowknife Bay / Back Bay				
Fish	**Muscle Concentration (mg/kg (ww))**	**Liver Concentration (mg/kg (ww))**	**Kidney Concentration (mg/kg (ww))**	**Reference**
Northern Pike	0.21	0.24	-	Jackson *et al.* (1996)
Northern Pike	0.22	0.34	0.6	Jackson *et al.* (1996)
Northern Pike	0.22	0.07	0.13	Jackson *et al.* (1996)
Northern Pike	0.22	0.15	0.14	Jackson *et al.* (1996)
Northern Pike	0.22	0.11	0.15	Jackson *et al.* (1996)
Northern Pike	0.22	0.12	0.26	Jackson *et al.* (1996)
Northern Pike	0.22	0.08	0.11	Jackson *et al.* (1996)
Northern Pike	0.22	0.63	0.13	Jackson *et al.* (1996)
Northern Pike	0.22	0.11	0.14	Jackson *et al.* (1996)
Northern Pike	0.23	0.18	0.4	Jackson *et al.* (1996)
Northern Pike	0.24	0.08	-	Jackson *et al.* (1996)
Northern Pike	0.24	0.05	0.19	Jackson *et al.* (1996)
Northern Pike	0.24	0.85	0.3	Jackson *et al.* (1996)
Northern Pike	0.24	0.64	0.21	Jackson *et al.* (1996)
Northern Pike	0.25	0.36	-	Jackson *et al.* (1996)
Northern Pike	0.25	0.29	0.26	Jackson *et al.* (1996)
Northern Pike	0.26	0.58	0.29	Jackson *et al.* (1996)
Northern Pike	0.26	0.12	0.24	Jackson *et al.* (1996)
Northern Pike	0.29	0.08	0.67	Jackson *et al.* (1996)
Northern Pike	0.3	0.18	0.21	Jackson *et al.* (1996)
Northern Pike	0.3	0.13	0.29	Jackson *et al.* (1996)
Northern Pike	0.3	0.26	-	Jackson *et al.* (1996)
Northern Pike	0.3	0.12	0.17	Jackson *et al.* (1996)
Northern Pike	0.3	-	-	Falk *et al.* (1973)
Northern Pike	0.32	-	-	Falk *et al.* (1973)
Northern Pike	0.32	0.83	0.33	Jackson *et al.* (1996)
Northern Pike	0.32	0.14	0.16	Jackson *et al.* (1996)
Northern Pike	0.33	0.08	0.13	Jackson *et al.* (1996)
Northern Pike	0.35	0.31	-	Jackson *et al.* (1996)
Northern Pike	0.36	0.17	-	Jackson *et al.* (1996)

TABLE A5-1 (Cont'd)
MEASURED ARSENIC LEVELS IN FISH TISSUE

Yellowknife Bay / Back Bay				
Fish	**Muscle Concentration (mg/kg (ww))**	**Liver Concentration (mg/kg (ww))**	**Kidney Concentration (mg/kg (ww))**	**Reference**
Northern Pike	0.36	0.1	0.69	Jackson *et al.* (1996)
Northern Pike	0.43	0.15	0.35	Jackson *et al.* (1996)
Northern Pike	0.44	0.05	-	Jackson *et al.* (1996)
Northern Pike	0.46	0.13	0.76	Jackson *et al.* (1996)
Northern Pike	0.5	0.7	0.4	Jackson *et al.* (1996)
Northern Pike	0.57	0.58	0.2	Jackson *et al.* (1996)
Northern Pike	0.8	0.34	0.22	Jackson *et al.* (1996)
Lake Whitefish	0.02	0.06	0.05	Jackson *et al.* (1996)
Lake Whitefish	0.03	0.09	0.05	Jackson *et al.* (1996)
Lake Whitefish	0.03	0.07	0.05	Jackson *et al.* (1996)
Lake Whitefish	0.03	0.03	0.09	Jackson *et al.* (1996)
Lake Whitefish	0.04	0.1	0.19	Jackson *et al.* (1996)
Lake Whitefish	0.04	0.46	-	Jackson *et al.* (1996)
Lake Whitefish	0.04	0.08	0.71	Jackson *et al.* (1996)
Lake Whitefish	< 0.05	0.28	<0.05	Jackson *et al.* (1996)
Lake Whitefish	< 0.05	0.44	<0.05	Jackson *et al.* (1996)
Lake Whitefish	< 0.05	0.21	0.11	Jackson *et al.* (1996)
Lake Whitefish	< 0.05	0.47	0.44	Jackson *et al.* (1996)
Lake Whitefish	< 0.05	0.33	0.33	Jackson *et al.* (1996)
Lake Whitefish	< 0.05	0.1	0.14	Jackson *et al.* (1996)
Lake Whitefish	< 0.05	0.45	1.01	Jackson *et al.* (1996)
Lake Whitefish	< 0.05	0.18	0.5	Jackson *et al.* (1996)
Lake Whitefish	< 0.05	0.1	0.12	Jackson *et al.* (1996)
Lake Whitefish	0.05	0.4	0.9	Jackson *et al.* (1996)
Lake Whitefish	0.05	0.16	0.51	Jackson *et al.* (1996)
Lake Whitefish	0.05	0.2	0.15	Jackson *et al.* (1996)
Lake Whitefish	0.05	0.09	0.13	Jackson *et al.* (1996)
Lake Whitefish	0.05	0.7	0.07	Jackson *et al.* (1996)
Lake Whitefish	0.06	0.25	0.05	Jackson *et al.* (1996)
Lake Whitefish	0.06	0.24	0.5	Jackson *et al.* (1996)

TABLE A5-1 (Cont'd)
MEASURED ARSENIC LEVELS IN FISH TISSUE

Yellowknife Bay / Back Bay				
Fish	**Muscle Concentration (mg/kg (ww))**	**Liver Concentration (mg/kg (ww))**	**Kidney Concentration (mg/kg (ww))**	**Reference**
Lake Whitefish	0.06	0.53	0.5	Jackson *et al.* (1996)
Lake Whitefish	0.06	0.19	0.86	Jackson *et al.* (1996)
Lake Whitefish	0.06	0.09	-	Jackson *et al.* (1996)
Lake Whitefish	0.06	0.19	0.35	Jackson *et al.* (1996)
Lake Whitefish	0.06	0.09	0.19	Jackson *et al.* (1996)
Lake Whitefish	0.06	0.09	-	Jackson *et al.* (1996)
Lake Whitefish	0.06	0.17	0.06	Jackson *et al.* (1996)
Lake Whitefish	0.06	0.61	0.08	Jackson *et al.* (1996)
Lake Whitefish	0.06	0.85	0.16	Jackson *et al.* (1996)
Lake Whitefish	0.06	0.58	0.34	Jackson *et al.* (1996)
Lake Whitefish	0.06	0.69	0.46	Jackson *et al.* (1996)
Lake Whitefish	0.06	0.12	0.1	Jackson *et al.* (1996)
Lake Whitefish	0.07	0.38	0.28	Jackson *et al.* (1996)
Lake Whitefish	0.07	0.35	0.33	Jackson *et al.* (1996)
Lake Whitefish	0.07	0.24	0.78	Jackson *et al.* (1996)
Lake Whitefish	0.07	0.1	0.48	Jackson *et al.* (1996)
Lake Whitefish	0.07	0.52	0.13	Jackson *et al.* (1996)
Lake Whitefish	0.08	0.17	0.33	Jackson *et al.* (1996)
Lake Whitefish	0.08	0.22	<0.05	Jackson *et al.* (1996)
Lake Whitefish	0.08	0.26	0.23	Jackson *et al.* (1996)
Lake Whitefish	0.08	0.23	0.31	Jackson *et al.* (1996)
Lake Whitefish	0.08	0.05	-	Jackson *et al.* (1996)
Lake Whitefish	0.08	0.71	0.63	Jackson *et al.* (1996)
Lake Whitefish	0.08	0.25	1.05	Jackson *et al.* (1996)
Lake Whitefish	0.08	0.06	0.11	Jackson *et al.* (1996)
Lake Whitefish	0.08	0.21	0.2	Jackson *et al.* (1996)
Lake Whitefish	0.09	0.12	0.19	Jackson *et al.* (1996)
Lake Whitefish	0.09	0.13	0.18	Jackson *et al.* (1996)
Lake Whitefish	0.09	0.12	0.21	Jackson *et al.* (1996)
Lake Whitefish	0.09	1.07	0.16	Jackson *et al.* (1996)

TABLE A5-1 (Cont'd)
MEASURED ARSENIC LEVELS IN FISH TISSUE

Yellowknife Bay / Back Bay				
Fish	**Muscle Concentration (mg/kg (ww))**	**Liver Concentration (mg/kg (ww))**	**Kidney Concentration (mg/kg (ww))**	**Reference**
Lake Whitefish	0.09	0.77	0.18	Jackson *et al.* (1996)
Lake Whitefish	0.09	0.12	-	Jackson *et al.* (1996)
Lake Whitefish	0.1	0.15	0.06	Jackson *et al.* (1996)
Lake Whitefish	0.1	0.3	0.07	Jackson *et al.* (1996)
Lake Whitefish	0.1	0.19	0.88	Jackson *et al.* (1996)
Lake Whitefish	0.1	0.25	0.12	Jackson *et al.* (1996)
Lake Whitefish	0.1	0.27	0.06	Jackson *et al.* (1996)
Lake Whitefish	0.1	0.24	0.65	Jackson *et al.* (1996)
Lake Whitefish	0.1	0.15	0.28	Jackson *et al.* (1996)
Lake Whitefish	0.1	1.06	-	Jackson *et al.* (1996)
Lake Whitefish	0.1	1.11	1.35	Jackson *et al.* (1996)
Lake Whitefish	0.1	0.64	1	Jackson *et al.* (1996)
Lake Whitefish	0.1	0.11	0.51	Jackson *et al.* (1996)
Lake Whitefish	0.1	0.13	-	Jackson *et al.* (1996)
Lake Whitefish	0.11	0.17	0.2	Jackson *et al.* (1996)
Lake Whitefish	0.11	0.15	0.09	Jackson *et al.* (1996)
Lake Whitefish	0.11	0.19	0.11	Jackson *et al.* (1996)
Lake Whitefish	0.11	0.09	0.09	Jackson *et al.* (1996)
Lake Whitefish	0.11	0.34	0.38	Jackson *et al.* (1996)
Lake Whitefish	0.11	0.3	0.05	Jackson *et al.* (1996)
Lake Whitefish	0.11	0.59	0.11	Jackson *et al.* (1996)
Lake Whitefish	0.11	0.59	0.51	Jackson *et al.* (1996)
Lake Whitefish	0.12	0.23	0.12	Jackson *et al.* (1996)
Lake Whitefish	0.12	0.3	0.06	Jackson *et al.* (1996)
Lake Whitefish	0.12	0.32	0.39	Jackson *et al.* (1996)
Lake Whitefish	0.12	0.33	0.54	Jackson *et al.* (1996)
Lake Whitefish	0.12	0.57	0.59	Jackson *et al.* (1996)
Lake Whitefish	0.12	0.11	0.15	Jackson *et al.* (1996)
Lake Whitefish	0.13	0.2	0.57	Jackson *et al.* (1996)
Lake Whitefish	0.13	0.1	0.06	Jackson *et al.* (1996)

TABLE A5-1 (Cont'd)
MEASURED ARSENIC LEVELS IN FISH TISSUE

Yellowknife Bay / Back Bay				
Fish	**Muscle Concentration (mg/kg (ww))**	**Liver Concentration (mg/kg (ww))**	**Kidney Concentration (mg/kg (ww))**	**Reference**
Lake Whitefish	0.13	0.12	-	Jackson *et al.* (1996)
Lake Whitefish	0.13	0.34	-	Jackson *et al.* (1996)
Lake Whitefish	0.13	0.06	0.16	Jackson *et al.* (1996)
Lake Whitefish	0.13	0.63	0.25	Jackson *et al.* (1996)
Lake Whitefish	0.13	1.15	-	Jackson *et al.* (1996)
Lake Whitefish	0.13	0.77	0.53	Jackson *et al.* (1996)
Lake Whitefish	0.13	0.52	0.97	Jackson *et al.* (1996)
Lake Whitefish	0.14	0.13	-	Jackson *et al.* (1996)
Lake Whitefish	0.14	0.22	0.46	Jackson *et al.* (1996)
Lake Whitefish	0.14	0.08	0.19	Jackson *et al.* (1996)
Lake Whitefish	0.14	0.24	0.1	Jackson *et al.* (1996)
Lake Whitefish	0.14	0.07	-	Jackson *et al.* (1996)
Lake Whitefish	0.14	0.05	0.12	Jackson *et al.* (1996)
Lake Whitefish	0.14	0.07	-	Jackson *et al.* (1996)
Lake Whitefish	0.14	0.18	-	Jackson *et al.* (1996)
Lake Whitefish	0.14	0.76	1.31	Jackson *et al.* (1996)
Lake Whitefish	0.15	0.21	0.54	Jackson *et al.* (1996)
Lake Whitefish	0.15	0.15	0.11	Jackson *et al.* (1996)
Lake Whitefish	0.15	0.87	2.65	Jackson *et al.* (1996)
Lake Whitefish	0.15	1.12	0.97	Jackson *et al.* (1996)
Lake Whitefish	0.16	0.45	0.4	Jackson *et al.* (1996)
Lake Whitefish	0.16	0.34	0.19	Jackson *et al.* (1996)
Lake Whitefish	0.16	0.15	0.29	Jackson *et al.* (1996)
Lake Whitefish	0.16	0.19	0.07	Jackson *et al.* (1996)
Lake Whitefish	0.16	0.21	0.05	Jackson *et al.* (1996)
Lake Whitefish	0.16	0.08	<0.05	Jackson *et al.* (1996)
Lake Whitefish	0.16	0.1	0.11	Jackson *et al.* (1996)
Lake Whitefish	0.16	0.12	0.07	Jackson *et al.* (1996)
Lake Whitefish	0.16	0.67	-	Jackson *et al.* (1996)
Lake Whitefish	0.17	0.13	0.36	Jackson *et al.* (1996)

TABLE A5-1 (Cont'd)
MEASURED ARSENIC LEVELS IN FISH TISSUE

Yellowknife Bay / Back Bay				
Fish	**Muscle Concentration (mg/kg (ww))**	**Liver Concentration (mg/kg (ww))**	**Kidney Concentration (mg/kg (ww))**	**Reference**
Lake Whitefish	0.17	0.39	0.15	Jackson *et al.* (1996)
Lake Whitefish	0.17	0.33	0.36	Jackson *et al.* (1996)
Lake Whitefish	0.17	0.16	0.31	Jackson *et al.* (1996)
Lake Whitefish	0.17	0.27	0.24	Jackson *et al.* (1996)
Lake Whitefish	0.17	0.45	0.12	Jackson *et al.* (1996)
Lake Whitefish	0.17	0.24	0.13	Jackson *et al.* (1996)
Lake Whitefish	0.17	0.7	-	Jackson *et al.* (1996)
Lake Whitefish	0.18	0.39	0.28	Jackson *et al.* (1996)
Lake Whitefish	0.18	0.38	0.62	Jackson *et al.* (1996)
Lake Whitefish	0.18	1.17	0.18	Jackson *et al.* (1996)
Lake Whitefish	0.18	1.03	-	Jackson *et al.* (1996)
Lake Whitefish	0.18	1.13	0.32	Jackson *et al.* (1996)
Lake Whitefish	0.18	0.08	0.16	Jackson *et al.* (1996)
Lake Whitefish	0.19	0.37	0.46	Jackson *et al.* (1996)
Lake Whitefish	0.19	0.19	0.09	Jackson *et al.* (1996)
Lake Whitefish	0.19	0.62	0.25	Jackson *et al.* (1996)
Lake Whitefish	0.19	0.81	0.3	Jackson *et al.* (1996)
Lake Whitefish	0.19	0.75	-	Jackson *et al.* (1996)
Lake Whitefish	0.2	0.35	1.39	Jackson *et al.* (1996)
Lake Whitefish	0.2	0.37	0.36	Jackson *et al.* (1996)
Lake Whitefish	0.2	0.05	-	Jackson *et al.* (1996)
Lake Whitefish	0.2	0.85	0.28	Jackson *et al.* (1996)
Lake Whitefish	0.2	0.99	0.37	Jackson *et al.* (1996)
Lake Whitefish	0.2	0.86	-	Jackson *et al.* (1996)
Lake Whitefish	0.21	0.18	0.33	Jackson *et al.* (1996)
Lake Whitefish	0.21	0.19	0.56	Jackson *et al.* (1996)
Lake Whitefish	0.21	0.23	-	Jackson *et al.* (1996)
Lake Whitefish	0.21	0.05	0.11	Jackson *et al.* (1996)
Lake Whitefish	0.21	0.08	0.45	Jackson *et al.* (1996)
Lake Whitefish	0.21	0.67	1.14	Jackson *et al.* (1996)

TABLE A5-1 (Cont'd)
MEASURED ARSENIC LEVELS IN FISH TISSUE

Yellowknife Bay / Back Bay				
Fish	**Muscle Concentration (mg/kg (ww))**	**Liver Concentration (mg/kg (ww))**	**Kidney Concentration (mg/kg (ww))**	**Reference**
Lake Whitefish	0.21	0.58	0.23	Jackson *et al.* (1996)
Lake Whitefish	0.22	0.17	0.33	Jackson *et al.* (1996)
Lake Whitefish	0.22	0.23	0.13	Jackson *et al.* (1996)
Lake Whitefish	0.22	0.12	3.43	Jackson *et al.* (1996)
Lake Whitefish	0.22	0.07	0.13	Jackson *et al.* (1996)
Lake Whitefish	0.25	0.27	0.28	Jackson *et al.* (1996)
Lake Whitefish	0.25	0.13	-	Jackson *et al.* (1996)
Lake Whitefish	0.26	0.15	-	Jackson *et al.* (1996)
Lake Whitefish	0.27	0.18	0.26	Jackson *et al.* (1996)
Lake Whitefish	0.28	0.4	0.34	Jackson *et al.* (1996)
Lake Whitefish	0.28	0.05	0.08	Jackson *et al.* (1996)
Lake Whitefish	0.28	0.59	0.26	Jackson *et al.* (1996)
Lake Whitefish	0.28	0.94	1.48	Jackson *et al.* (1996)
Lake Whitefish	0.28	0.67	0.33	Jackson *et al.* (1996)
Lake Whitefish	0.29	0.29	0.58	Jackson *et al.* (1996)
Lake Whitefish	0.29	0.13	0.14	Jackson *et al.* (1996)
Lake Whitefish	0.29	0.09	0.08	Jackson *et al.* (1996)
Lake Whitefish	0.29	0.08	0.06	Jackson *et al.* (1996)
Lake Whitefish	0.3	0.31	0.38	Jackson *et al.* (1996)
Lake Whitefish	0.3	0.25	0.62	Jackson *et al.* (1996)
Lake Whitefish	0.3	0.13	0.26	Jackson *et al.* (1996)
Lake Whitefish	0.31	0.19	0.53	Jackson *et al.* (1996)
Lake Whitefish	0.31	0.42	0.28	Jackson *et al.* (1996)
Lake Whitefish	0.31	0.05	0.34	Jackson *et al.* (1996)
Lake Whitefish	0.31	0.05	0.21	Jackson *et al.* (1996)
Lake Whitefish	0.32	0.38	0.18	Jackson *et al.* (1996)
Lake Whitefish	0.32	0.2	0.24	Jackson *et al.* (1996)
Lake Whitefish	0.32	0.71	0.59	Jackson *et al.* (1996)
Lake Whitefish	0.33	0.35	0.36	Jackson *et al.* (1996)
Lake Whitefish	0.33	0.11	0.16	Jackson *et al.* (1996)

TABLE A5-1 (Cont'd)
MEASURED ARSENIC LEVELS IN FISH TISSUE

Yellowknife Bay / Back Bay				
Fish	**Muscle Concentration (mg/kg (ww))**	**Liver Concentration (mg/kg (ww))**	**Kidney Concentration (mg/kg (ww))**	**Reference**
Lake Whitefish	0.35	0.54	0.27	Jackson *et al.* (1996)
Lake Whitefish	0.35	0.05	0.13	Jackson *et al.* (1996)
Lake Whitefish	0.38	0.28	0.7	Jackson *et al.* (1996)
Lake Whitefish	0.38	0.21	<0.05	Jackson *et al.* (1996)
Lake Whitefish	0.38	0.08	-	Jackson *et al.* (1996)
Lake Whitefish	0.39	0.17	0.11	Jackson *et al.* (1996)
Lake Whitefish	0.39	0.26	0.41	Jackson *et al.* (1996)
Lake Whitefish	0.39	0.61	0.39	Jackson *et al.* (1996)
Lake Whitefish	0.41	0.21	0.72	Jackson *et al.* (1996)
Lake Whitefish	0.41	0.58	0.32	Jackson *et al.* (1996)
Lake Whitefish	0.42	0.19	0.34	Jackson *et al.* (1996)
Lake Whitefish	0.43	0.31	-	Jackson *et al.* (1996)
Lake Whitefish	0.43	0.26	0.13	Jackson *et al.* (1996)
Lake Whitefish	0.43	0.8	0.46	Jackson *et al.* (1996)
Lake Whitefish	0.44	0.45	0.08	Jackson *et al.* (1996)
Lake Whitefish	0.45	0.83	0.43	Jackson *et al.* (1996)
Lake Whitefish	0.47	0.56	0.32	Jackson *et al.* (1996)
Lake Whitefish	0.47	0.4	0.59	Jackson *et al.* (1996)
Lake Whitefish	0.5	0.17	0.1	Jackson *et al.* (1996)
Lake Whitefish	0.54	0.81	0.36	Jackson *et al.* (1996)
Lake Whitefish	0.55	0.84	0.52	Jackson *et al.* (1996)
Lake Whitefish	1.11	0.53	0.09	Jackson *et al.* (1996)
Whitefish	<0.2	<0.2	-	Falk *et al.* (1973)
Whitefish	<0.2	-	-	Falk *et al.* (1973)

TABLE A5-1 (Cont'd)
MEASURED ARSENIC LEVELS IN FISH TISSUE

Fish	Muscle Concentration (mg/kg (ww))	Reference
Burbot	0.11 ± 0.05	1995, Evans *et al*. (2001)
Burbot	0.12 ± 0.02	1999, Evans *et al*. (2001)
Burbot/ Loche	0.13	mean of 5 samples, Boucher *et al*. (1997)
Burbot	0.13 ± 0.05	1996, Evans *et al*. (2001)
Burbot	0.13 ± 0.05	1999, Evans *et al*. (2001)
Inconnu/ Connie	0.32	mean of 5 samples, Boucher *et al*. (1997)
Lake Trout	0.21 ± 0.07	1999, Evans *et al*. (2001)
Lake Trout	0.22 ± 0.05	1995, Evans *et al.* (2001)
Lake Trout	0.28 ± 0.12	1999, Evans *et al*. (2001)
Northern Pike	0.10 ± 0.03	1999, Evans *et al*. (2001)
Northern Pike/ Jack	0.15	mean of 5 samples from 1996, Boucher *et al.* (1997)
Northern Pike	0.15 ± 0.04	1996, Evans *et al*. (2001)
Northern Pike	0.16 ± 0.05	1999, Evans *et al*. (2001)
Walleye/ Pickerel	0.08	mean of 5 samples, Boucher *et al*. (1997)

Note: mg/kg (ww) = parts per million, wet weight, which is equivalent to µg/g (ww) or mg/kg (ww).

TABLE A5-2
DERIVED WATER-TO-FISH TRANSFER FACTORS FOR ARSENIC (m^3/g (ww))

Fish-tissue	# of Samples	Minimum	Maximum	Geometric Mean	Geo Std Dev	Average	Avg Std Dev
Lake whitefish – muscle	186	$6.9x10^{-7}$	$4.9x10^{-4}$	$5.2x10^{-5}$	2.8	$8.0x10^{-5}$	$7.4x10^{-5}$
Lake whitefish – liver	206	$2.2x10^{-6}$	$5.2x10^{-4}$	$9.2x10^{-5}$	2.4	$1.5x10^{-4}$	$1.2x10^{-5}$
Lake whitefish – kidney	165	$2.2x10^{-6}$	$1.2x10^{-3}$	$8.8x10^{-5}$	3.5	$1.5x10^{-4}$	$1.6x10^{-4}$
Northern pike – muscle	83	$1.3x10^{-6}$	$5.5x10^{-4}$	$5.5x10^{-5}$	3.7	$9.9x10^{-5}$	$8.9x10^{-5}$
Northern pike - liver	85	$2.2x10^{-6}$	$4.8x10^{-4}$	$5.6x10^{-5}$	5.0	$1.3x10^{-4}$	$1.1x10^{-4}$

Note: a – count is the number of fish samples taken. Six water samples in each of six locations were available for all calculations.

Table A5-3 presents the probability distribution used for the model.

TABLE A5-3
WATER-TO-FISH TRANSFER FACTORS USED IN THE ASSESSMENT

Receptor	Summary of Measured Data					Lognormal Distribution Specifiers			
	# of Samples	Minimum	Maximum	Geometric Mean	Geometric Std Dev	GM	GSD	Minimum	Maximum
Fish	186	$6.9x10^{-7}$	$4.9x10^{-4}$	$5.2x10^{-5}$	2.8	$5.2x10^{-5}$	2.8	$6.6x10^{-6}$	$4.1x10^{-4}$

Note: GM – Geometric Mean.
Geo Std Dev, GSD – Geometric Standard Deviation.
Min and max for the lognormal distribution were set equal to 2 GSD from the GM.

A6.0 MEASURED ARSENIC LEVELS IN SNAILS

Snails were collected from the Baker Creek Outlet and from Baker Creek near the mill area at the Giant Mine (Koch 1998). The samples were composited on the whole and shelled samples. A summary of the data is provided in Table A6-1. The arsenic levels in the whole and shelled snails were divided by the average measured water concentration for the two sample sites to derive the site-specific transfer factors reported on Table A6-1.

Speciation analysis of the snails indicated that 40 to 60% of the arsenic was in an organic form, tetramethylarsonium. The inorganic species of arsenic (As(III) and As(V)) were found to represent 25% to 40% of the extracted arsenic. These statistics are characteristic when observing freshwater species. Arsenobetaine and Aresenosugar X both were found within the snails at 0.6% and 3.5% respectively.

TABLE A6-1
ARSENIC LEVELS IN SNAILS AND DERIVED WATER-TO-SNAIL TRANSFER FACTORS

Species of Snail	Moisture Content (%)	Measured Arsenic Concentration (mg/kg dry weight)	Estimated Arsenic Concentration (mg/kg fresh weight)	Average Arsenic concentration in water (mg/kg)	Transfer Factor (based on fresh weight)
Stagnicola sp., Shelled	81	82	15.6	0.17	97
Stagnicola sp., whole	69	83	25.7		151
Averages	75	82.5	21	0.17	124

Note:
mg/kg = parts per million which is equivalent to µg/g or mg/kg or mg/L.

A7.0 MEASURED ARSENIC LEVELS IN BENTHIC INVERTEBRATES

Benthic invertebrates were sampled by Dillon Consulting Limited (Dillon 2002b) at one location in Baker Creek upstream of the Giant Mine (BC-1) and two locations in the Baker Creek Outlet downstream of the Giant Mine site (BC-3 and BC-4), using Hester-Dendy Plates. In addition, sampling was undertaken in 2003 in 7 different locations (3 sites upstream of the mine and 4 downstream of the mine) (Dillon 2004). Samples analyzed for metals were immediately frozen after collection. A summary of the data from Dillon (2002b and 2004) is provided in Table A7-1. The arsenic level in the composite benthic sample from each location was divided by the measured water concentration from the same location to calculate a site-specific transfer factor (shown in Table A7-1). An assumed moisture content of 80% was used for the calculations.

TABLE A7-1
ARSENIC LEVELS IN BENTHIC INVERTEBRATES AND DERIVED WATER-TO-BENTHIC INVERTEBRATE TRANSFER FACTORS

Sample Location	Measured Arsenic Concentration (mg/kg dry weight)	Estimated Arsenic Concentration (mg/kg wet weight)	Arsenic Concentration in Water (mg/L)	Transfer Factor L/kg (based on wet weight)	Source of Measurement Data
BC-1	43	8.6	0.05	172.0	Dillon 2002b
BC-3	71	14.2	0.232	61.2	Dillon 2002b
BC-4	201	40.2	0.216	186.1	Dillon 2002b
Site 1	1.5	0.3	0.049	6.1	Dillon 2004
Site 2	3.8	0.76	0.052	14.6	Dillon 2004
Site 3	10.2	2.04	0.204	10.0	Dillon 2004
Site 4	7.7	1.54	0.248	6.2	Dillon 2004
Site 5	21.7	4.34	0.218	19.9	Dillon 2004
Site 6	50.2	10.04	0.233	43.1	Dillon 2004
Site 7	19.5	3.9	0.174	22.4	Dillon 2004

Note:
mg/kg = parts per million which is equivalent to µg/g or mg/kg.

The average value of the benthic transfer factor is 54 L/kg (ww) (or 5.4×10^{-5} m^3/g (ww)). The geometric mean value of the transfer factors shown in Table A7-1 is 27 L/kg (ww) (or 2.7×10^{-5} m^3/g (ww)). These values were combined with the water-to-snail transfer factors provided in Table A6-1 to give an average value of 63 L/kg (ww) (6.3×10^{-5} m^3/g (ww)) and a geometric mean of 32 L/kg (3.2×10^{-5} m^3/g (ww)). A lognormal distribution was used in the modeling based on the geometric mean of 3.2×10^{-5} m^3/g (ww) and a geometric standard deviation of 3.5.

A8.0 MEASURED CONCENTRATIONS OF ARSENIC IN LABRADOR TEA

A review of the information from the Medicinal Plants Study Report 2002-2003 indicates concentrations of arsenic in tea (Chan 2003). Table A8-1 provides samples selected from the Giant Mine site. Samples were selected for the purposes of this assessment as it is assumed that the medicinal plants (like berries) would be obtained from the site. Some of the other locations were a distance from the site. As seen in the Table A8-1, Labrador tea does not have the highest concentration, rather lichen tea (soup) has the highest concentrations. Table A8-2 provides samples selected from the Dettah Community. A review of the data in the report indicates that the next highest concentration of arsenic in tea is 170.4 µg/L from Tamarak bark and needles at the Con Mine near Meg Lake. Summary statistics for the derived probability distributions for Giant Mine and Dettah sites is provided in Table A8-3.

TABLE A8-1
ARSENIC CONCENTRATIONS IN MEDICINAL TEA AT GIANT MINE

Location	Plant Species	Arsenic Concentration in Tea µg/L
Giant Mine	Labrador Tea	29.1
Giant Mine Shore line	Labrador Tea	0.1
Yellowknife River	Labrador Tea	0.05
Giant Mine Shore line	Lichen	527.4
Giant Mine Shore line	Soapberry	1.5
Giant Mine Shore line	Spruce bark & needles	64.7
Yellowknife River	Spruce bark & cones	5.2
Giant Mine Shore line	Spruce gum	5.1
Giant Mine	Spruce bark & cones	63.5
Giant Mine Shore line	Willow branch & leaves	3.3

TABLE A8-2
ARSENIC CONCENTRATIONS IN MEDICINAL TEA AT DETTAH COMMUNITY

Location	Plant Species	Arsenic Concentration in Tea µg/L
Duck Lake	Labrador Tea	0.05
Con Mine near Meg Lake	Labrador Tea	0.1
Con Mine	Labrador Tea	16.2
Con Mine	Labrador Tea	1.6
Down creek Peg Lake	Labrador Tea	0.05
S of Con Mine shore of small lake	Labrador Tea	0.05
Mol K'e (near Wool Bay)	Labrador Tea	0.05
Wool Bay	Labrador Tea	0.05
Con Mine near Meg Lake	Tamarack	5.6
Con Mine near Meg Lake	Tamarak bark & needles	170.4
Down creek Peg Lake	Tamarak bark & needles	53.5
S of Con Mine shore of small lake	Tamarak bark & needles	9.7
Con Mine near Meg Lake	Lichen	101.8
Down creek Peg Lake	Lichen	9.5
S of Con Mine shore of small lake	Lichen	4.4
Mol K'e (near Wool Bay)	Birch tree	21.1
Wool Bay	Birch tree	158.5
Duck Lake	Spruce gum	0.05
Con Mine near Meg Lake	Spruce bark & needles	152.7
Con Mine near Meg Lake	Spruce gum	0.05
Con Mine near Meg Lake	Spruce gum	8.6
Con Mine	Spruce branch & cones	104.8
Down creek Peg Lake	Spruce gum	7.6
S of Con Mine shore of small lake	Spruce gum	1.2
Mol K'e (near Wool Bay)	Spruce gum	0.05
Wool Bay	Spruce gum	0.05
Con Mine near Meg Lake	Willow branch & leaves	27.2
Con Mine near Meg Lake	Willow branch & leaves	4.1
Duck Lake	Willow branch & leaves	0.05

TABLE A8-3
DERIVED PROBABILITY DISTRIBUTIONS FOR
LABRADOR TEA BY RECEPTOR LOCATION (µg/L)

Location	Summary of Measured Data						Lognormal Distribution Specifiers			
	# of Samples	Min	Max	Arith-metic Mean	Geo Mean	Geo Std. Dev.	GM	GSD	Min	Max
Giant Mine	9	0.1	527.4	77.8	32.3	3.8	32.3	3.8	0.58	1772
Dettah	29	0.05	170.4	29.6	14.5	3.3	14.5	3.3	0.40	521

Notes: GM – Geometric Mean
Geo Std Dev, GSD – Geometric Standard Deviation
Min and max for the lognormal distribution were set equal to 2 GSD from the GM

A9.0 MEASURED ARSENIC CONCENTRATIONS IN AIR

The Department of Resources, Wildlife and Economic Development monitors air quality in the Northwest Territories. Ambient air quality includes the contribution from all sources, including background. Several ambient air quality parameters are measured in Yellowknife including arsenic. Table A9-1 presents the highest 24-hr average, the lowest 24-hr average and the annual average for arsenic in Yellowknife air from NWT (2001).

TABLE A9-1
ARSENIC CONCENTRATIONS IN AIR FOR YELLOWKNIFE

Arsenic (µg/m^3)	1997	1998	1999	2000
Highest (24-hr)	0.063	0.037	0.026	0.026
Lowest (24-hr)	0.002	0.001	0.001	0.002
Annual average	0.005	0.004	0.005	0.004

A mean annual arsenic concentration of 0.004 µg/m^3 in Yellowknife was used to estimate the background arsenic concentration for the purposes of this assessment. It is recognized that not all locations will have the same background arsenic concentration. For example, it is expected that the background arsenic concentration in the Dettah community is likely less than 0.004 µg/m^3. This background concentration was added to the predicted concentrations at each receptor location to estimate total arsenic concentrations in air for comparison to the derived ambient criteria.

Monitoring of TSP and arsenic was carried out on the Giant Mine site at four locations during the summer of 2004 (i.e. from 9 July to 30 September 2004). The monitoring locations included the former Giant Mine Townsite, near the South Pond (i.e. tailings area), near the mill/roaster complex and near B3 pit (SENES 2005). A summary of the arsenic monitoring results is provided in Table A9-2.

TABLE A9-2
ARSENIC CONCENTRATION IN PARTICULATE MATTER AT GIANT MINE SITE

Arsenic ($\mu g/m^3$)	Giant Mine Townsite	South Pond	Mill/ Roaster	B3 Pit Area
Maximum	0.042	0.076	0.061	0.121
Median	0.0009	0.009	0.011	0.015
Mean	0.008	0.019	0.016	0.025

While the data in Tables A9-1 and A9-2 are not directly comparable as the former is based on annual measurements and the later is based on summer measurements only, it is noteworthy that the mean level reported for the Giant Mine Townsite on Table A9-2 is within a factor of two of the mean annual level reported on Table A9-1 for Yellowknife over the 1997 to 2000 period. As TSP and arsenic levels are much lower when the ground is frozen and covered with snow, it is expected that the annual average concentration at the Townsite is likely similar to that measured in the City of Yellowknife. Not surprising, the arsenic levels measured at the other locations noted on Table A9-2 are higher than reported for the Giant Mine Townsite. These monitors were purposely located in areas with elevated arsenic levels in either tailings or contaminated soils.

A10.0 REFERENCES

Boucher, M., M.S. Evans, P. Simon, G. Lowe and D. Muir 1997. *Fort Resolution Domestic Fishing Zone Contaminants Research.* From Synopsis of Research Conducted under the 1995-1997 Northern Contaminants Program. Indian and Northern Affairs Canada: 74.

Chan, L. 2003. Medicinal Plants Study Report 2002-2003. *Yellowknives Dene First Nation Land and Environment Committee's Report.* Northern Contaminants Program, DIAND. Presented by Dr. Lawrie Chan, School of Dietetics and Human Nutrition at Dettah Community Meeting. May 2003.

Con Mine 1987. ConMine_1987_data_rev.csv. Provided by Mark Richardson.

Davey, E., D. Maxwell and G. Stephens 1998. *Arsenic levels in berries and soils from the Yellowknives Dene First Nation Traditional Territory*. From Synopsis of Research Conducted under the 1998-1999 Northern Contaminants Program. Indian and Northern Affairs Canada: 81-85.

de Rosemond, S. and K. Liber 2004. *Arsenic Concentration and Speciation in Fish from Back Bay Near Yellowknife, NT.* Draft Report Submitted to Department of Indian Affairs and Northern Development, Yellowknife. June.

Dillon Consulting Limited 2004. *Biological Sampling at Baker Creek 2003.* Technical Report – Draft. Submitted to Indian and Northern Affairs Canada. April.

Dillon Consulting Limited 2002a. *Biological Sampling at Baker Creek – Summary Report.* Department of Indian and Northern Affairs Development. April 2002. 01-9670-3000.

Dillon Consulting Limited 2002b. *Biological Sampling at Baker Creek – Technical Report Final.* Department of Indian and Northern Affairs Development. November 2002. 02-0465.

Dushenko, W.T., D.A. Bright and K.J. Reimer 1995. *Arsenic Bioaccumulation and Toxicity in Aquatic Macrophytes Exposed to Gold-mine Effluent: Relationships with Environmental Partitioning, Metal Uptake and Nutrients*. Aquatic Botany 50: 141-158.

EBA Engineering Consultants 1998. *'Ebaarsenic98.xls'*. Provided by Mark Richardson.

Environmental Sciences Group (ESG) 2001a. *Characterization of Arsenic in Solid Phase Samples Collected on the Giant Mine Townsite, Yellowknife, NWT.* Environmental Sciences Group, Royal Military College of Canada: Ollson, Koch, Reimer. Prepared for Indian and Northern Affairs Canada.

Environmental Sciences Group (ESG) 2001b. *Characterization of the Potential Human Health Risk from Consumption of Garden Produce in Yellowknife, NWT.*

Environmental Sciences Group (ESG) 2000a. *Environmental Study of Arsenic Contamination from the Giant Mine, Yellowknife, Northwest Territories, Part 1*. Environmental Sciences Group, Royal Military College of Canada. Prepared for Indian and Northern Affairs Canada. RMC-CCE-ES-00-024. November.

Environmental Sciences Group (ESG) 2000b. *Distinguishing Between Natural and Anthropogenic Inputs of Arsenic in the Yellowknife Area*. Draft. Environmental Sciences Group, Royal Military College of Canada. Prepared for Indian and Northern Affairs Canada. RMC-CCE-ES-00-025. November.

Evans, M.S., D. Muir, M. Whittle, G.A. Stern, L. Lockhart, G. Low, M. Boucher, B. Parlee and T. Heron 2001. *Spatial and Long-term Trends in Organic Contaminants and Metals in Fish Species Important to the Commercial, Sports and Domestic Fisheries of Great Slave Lake and the Slave River Ecosystem.* Synopsis of Research conducted under the 2000-2001 Northern Contaminants Program. Indian and Northern Affairs Canada.

Falk, M.R., M.D. Miller and S.J.M. Kostiuk 1973. *Data Supplement to: Biological Effects of Mining Wastes in the Northwest Territories.* Environment Canada Fisheries and Marine Service.

Golder 2001. *Final Abandonment and Restoration Plan.* Prepared for Miramar Giant Mine Ltd. September.

Hutchinson, T.C., S. Aufreiter and R.G.V. Hancock 1982. *Arsenic Pollution in the Yellowknife Area from Gold Smelter Activities.* Journal of Radioanalytical Chemistry 71: 59-73.

Jackson, F.J., C.N. Lafontaine and J. Klaverkamp 1996. *Yellowknife – Back Bay Study on Metal and Trace Element Contamination of Water, Sediment and Fish*. Department of Indian and Northern Affairs Canada. November.

Koch, I., L. Wang, C.A. Ollson, W.R. Cullen and K.J.Reimer 2000. *The Predominance of Inorganic Arsenic Species in Plants from Yellowknife, Northwest Territories, Canada.* Environmental Science and Technology. 34(1): 22-26.

Koch, I. 1998. *Arsenic and Antimony Species in the Terrestrial Environment.* Thesis. University of Waterloo.

Northwest Territories (NWT) 2001. *Air Quality Monitoring in the Northwest Territories 2000/2001 data*. www.gov.nt.ca/RWED/eps/environ.htm.

Obst, J., W. Coedy and R.G. Bromley 2000. *Heavy Metals Analyses of Wild Edible Mushrooms in the North Great Slave Lake Region, Northwest Territories.* From Synopsis of Research Conducted under the 2000-2001 Northern Contaminants Program. Indian and Northern Affairs Canada: 263 – 266.

O'Toole, J.J., R.G. Clark, K.L. Malaby and D.L. Trauger 1972. *In Nuclear Methods in Environmental Research*. Univ. Missouri Press, Columbia, MO. 172-185. cited in Hutchinson et al. (1982).

Risklogic Scientific Services Inc. 2002. *Determining Natural (Background) Arsenic Soil Concentrations in Yellowknife, NWT, and Deriving Site-Specific Human Health-Based Remediation Objectives for Arsenic in the Yellowknife Area.* Final Report. Submitted to: Yellowknife Arsenic Soils Remediation Committee (YASRC). April.

Royal Military College of Canada (RMCC) 1999. *Arsenic Contamination of the Terrestrial and Freshwater Environment Impacted by Gold Mining Operations Yellowknife, NWT*. November. M.Sc. Thesis, Christopher Andrew Ollson.

SENES Consultants Limited (SENES) 2005. *Air Quality Monitoring at Giant Mine Site – Yellowknife A Baseline Study.* Prepared for Indian and Northern Affairs Canada. April.

Wagemann, R., N.B. Snow, D.M. Rosenberg and A. Lutz, 1978. *Arsenic in Sediments, Water and Aquatic Biota from Lakes in the Vicinity of Yellowknife, Northwest Territories, Canada.* Archives of Environmental Contamination and Toxicology 7: 129-191. cited by RMCC.

Yellowknife Arsenic Soils Remediation Committee (YASRC). *Locations Taiga General Export.xls.*

APPENDIX B

LAKEVIEW MODEL DESCRIPTION

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APPENDIX B LAKEVIEW MODEL DESCRIPTION

B1.0 OVERVIEW

To assess the fate of contaminants in the surface waters and sediments of lakes and rivers, a proprietary computer code called LAKEVIEW has been developed by SENES Consultants Limited. The code may be run in either a deterministic or a probabilistic manner. The code has been applied on several occasions on Hatchet Lake and Wollaston Lake in northern Saskatchewan to assess the impact of uranium mining operations within the watersheds on metal and radionuclide levels in several of the lake embayments (SENES 1989, 1993a, 1993b, 1995a, 1995b, 1995c, 1995d, 1996, 2000, 2001a). Recently, the LAKEVIEW model was applied to simulate the effects of arsenic input to Back Bay and Yellowknife Bay on Great Slave Lake (SENES, 2001b).

The important processes incorporated into the LAKEVIEW model include horizontal (lateral) and vertical transport of dissolved species, chemical and biochemical reactions, settling of particulate matter, and sediment exchange. Chemical reactions comprise both equilibrium and dynamic processes. Equilibrium processes are reactions for which the time scale is at least two orders of magnitude less than the time scale of the modelling. These may be metal-ligand interactions, adsorption to solids, colloidal chemistry, etc. (Stumm and Morgan 1981; Stumm 1992), which are necessary to estimate constituent ion balances and pH. Equilibrium modelling algorithms such as the MINTEQL (Schecker and McAvoy 1994) are well suited to establish these tasks. However, these programs are rather cumbersome (too slow and detailed) for probabilistic model applications and a simplified equilibrium modelling approach developed by Scharer *et al*. (1994) is employed in the LAKEVIEW model.

Another important equilibrium process is speciation resulting from the adsorption of dissolved species to solids. These solids may be organic or inorganic in nature. Dynamic processes comprise oxidative and reductive chemical reactions and consumption of nutrients by aquatic microbiota and macrobiota. An important, and often modeled, dynamic process is the uptake of dissolved oxygen in the water column and benthos underlying the water column. Often the time scale of these reactions is such that steady state approximation is possible on either an annual or monthly time scale. For example, the oxygen re-aeration and consumption rates in a water column may be calculated and be assumed to be constant for a period of one month, say in July, resulting in constant (steady state) concentration of oxygen for that particular month. These steady state approximations greatly reduce the complexity of the calculations.

The most important transport modes include net convective flow, horizontal dispersion, vertical dispersion, sediment exchange and settling (Schnoor 1996). The primary cause of net convective flow is hydraulic inputs and outputs due to river discharges and direct precipitation on the lake

surface. Vertical dispersion is important in stratified lakes or at time scales of less than one year. Due to spring and fall turnovers, vertical stratification at larger time scales is usually less important. In highly productive lake environments, the production, settling and decay of phytoplankton may play significant roles.

The movement of various chemical species in the sediment exchange zone is particularly important. Sediments may be a significant source of inorganic substances such as toxic metal ions, nutrient chemicals (phosphate, for example), and recalcitrant organic compounds. Since the solubility of most metal ions is considerably higher under aerobic than anaerobic conditions, most exchange occurs from the oxidized zone of sediments (Davé *et al*. 1997). A portion of the sedimented material may be re-introduced into the water column by diffusive processes; the remaining part becomes buried by the continual deposition of fresh settling mater.

B2.0 LAKE DISPERSION COMPONENT

B2.1 CONTROL VOLUME CONCEPT

A very common approach to water quality modelling is the control volume concept. The control volume is defined as a homogeneous volume segment of water with clearly defined boundaries and with water constituent concentrations either constant or varying continuously in a linear manner within the segment. The control volume segment may be hydraulically connected with one or more other segments forming a network. Control volume models with assumed constant concentrations are known as compartment or box models (Schnoor 1996). Key elements of the control volume models include volume, surface area, areas of exchange zones with adjacent control volumes, data on hydraulic inputs and outputs, information on transport, and reaction kinetics.

In the LAKEVIEW model, the lake is divided into one or more segments. Each segment consists of at least two zones: surface water and sediment. As an optional feature of the model, surface water is allowed to stratify into three layers during the summer months: the upper layer known as the epilimnion, a transition layer (metalimnion) and the bottom layer called the hypolimnion. Taking into account the horizontal aspect of mass transfer between the two or more segments and the vertical aspect between the epilimnion, the hypolimnion and the underlying sediment, the advective and dispersive processes in each segment are modelled by differential equations for the water column and sediment. An additional equation is used to simulate chemical constituent behaviour during thermal stratification. The volume change in stratified lake is subject to the following constraint:

$$\frac{dV_T}{dt} = \frac{dV_j}{dt} + \frac{dV_k}{dt} = 0 \qquad \text{(B2-1)}$$

where:

V_T = the total lake volume (m^3)
V_j = volume of the epilimnion (m^3)
V_k = volume of the hypolimnion (m^3)

According to equation (B2-1), the total lake volume is assumed to be constant. The rate of volume change is calculated explicitly for the hypolimnion (V_k) as follows:

$$\frac{dV_k}{dt} = Q_p - cV_k \tag{B2-2}$$

where:

Q_p = effluent discharge to the hypolimnion during lake stratification ($m^3 s^{-1}$)
V_k = volume of the hypolimnion (m^3)
c = characteristic time constant (yr^{-1})

Equation (B2-2) allows direct effluent loading to the hypolimnion. The characteristic time constant, c, is based on the inflection point of the themocline or chemocline.

The rate of change of the epilimnetic volume is found by employing equation (B2-1). The rate of contaminant mass change in the stratified layers is calculated using the chain rule. For example, the mass differential for the epilimnion is given by:

$$\frac{d(V_j C_j)}{dt} = V_j \frac{dC_j}{dt} + C_j \frac{dV_j}{dt} \tag{B2-3}$$

where:

V_j = volume of the epilimnion (m^3)
C_j = concentration of a contaminant in the epilimnion ($g\ m^{-3}$)

A similar equation can be written for the hypolimnion. The mass differential given by equation (B2-3) is equal to the mass input and output for a particular zone. The overall mass balance for the epilimnion is:

$$\frac{d(V_j C_{i,j})}{dt} = W_j(t) - Q_j(C_{i,j} - C_{i,j-1}) + E'_{j-1}A_{j-1}(C_{i,j-1} - C_{i,j}) - E'_j A_j(C_{i,j} - C_{i,j+1}) + E_k A_k(C_{i,k} - C_{i,j}) - S_j\ \sigma_j\ k_D\ A_k\ C_{i,j} + V_j\left(\frac{dC_{i,j}}{dt}\right)_{RX} \tag{B2-4}$$

where:

$C_{i,j-1}, C_{i,j}, C_{i,j+1}$	=	concentration of the 'ith' contaminant in the epilimnion of adjacent segments "j-1", "j" and "j+1 (g m^{-3})
W_j	=	effluent load (time dependent) to the epilimnion of segment "j" (g yr^{-1})
V_j	=	volume of epilimnion in segment "j" (m^3)
Q_j	=	outflow from segment "j" (m^3 yr^{-1})
E'_j, E'_{j-1}	=	lateral mass transfer (i.e. dispersion) coefficient between adjacent segments (m yr^{-1})
A_{j-1}, A_j	=	contact area between segments "j-1 and j" and "j and j+1" (m^2)
E_k	=	mass transfer coefficient across the thermocline (m yr^{-1})
A_k	=	contact area between the epilimnion and the hypolimnion (m^2)
$C_{i,k}$	=	concentration of "ith" contaminant in the hypolimnion below segment "j" (g m^{-3})
S_j	=	solids settling velocity in the epilimnion (m yr^{-1})
k_D	=	sorption coefficient for "ith" contaminant on sedimenting organic matter (m^3 g^{-1})
σ_j	=	suspended solids concentration in epilimnion (g m^{-3})
$(dC_{i,j}/dt)_{RX}$	=	reaction rate of the "ith" contaminant in the epilimnion (g m^{-3} yr^{-1})

A nominal step size of one month (Δt = 1 month) is often used for computational purposes. The model allows time step sizes of up to one year. The internal step size for computing is considerably smaller. Dispersion is a two way process, while sedimentation is a one way process. Sedimentation transport is based on adsorption equilibrium with solid surfaces and settling to the bottom of the water column. The numerator in the sedimentation transport term is known as the sedimentation flux for a particular contaminant. The solids settling velocity is the gravitational settling velocity in water calculated from a modified Stokes law equation (Thibodeaux 1996) using median settling particle size and density. The chemical settling flux resulting from sedimenting phytoplankton is also considered. Parameter estimates for dispersive transport are derived in the following section.

The differential equation for the hypolimnion was developed in an analogous manner:

$$\frac{d(V_k C_{i,k})}{dt} = W_k(t) - S_k\ \sigma_j\ k_D\ A_k\ C_{i,k} + K_l A_s (C_{i,s} - C_{i,k}) + E_k A_k (C_{i,j} - C_{i,k}) + V_k \left(\frac{dC_{i,k}}{dt} \right)_{RX} \qquad \text{(B2-5)}$$

where:

$C_{i,k}$	=	concentration of the "ith" contaminant in the hypolimnion (g m^{-3})
$C_{i,s}$	=	concentration of the "ith" contaminant in the sediment pore water (g m^{-3})

K_l	=	mass transfer coefficient between the sediment and the water column (m yr^{-1})
A_s	=	interfacial area between the sediment and the water column (m^2)
σ_j	=	suspended solids concentration (g m^{-3})
W_k	=	contaminant load for "ith" contaminant to the hypolimnion (g yr^{-1})
S_k	=	solids settling velocity in the hypolimnion (m yr^{-1})
k_D	=	overall sediment sorption coefficient for "ith" contaminant (m^3 g^{-1})
$(dC_{i,k}/dt)_{RX}$	=	reaction rate of the "ith" contaminant in the hypolimnion (g m^{-3} yr^{-1})

For sediments, the "retardation factor", Ψ is first calculated:

$$\Psi = [(1-\varepsilon)\rho\, k'_D + \varepsilon)]\, A_s\, z_j \qquad \text{(B2-6)}$$

Using this factor, the differential equation is given as follows:

$$\Psi \frac{dC_{i,s}}{dt} = K_\ell\, A_s (C_{i,k} - C_{i,s}) + \sigma_k\, k_D\, S_k\, C_i\, A_s - \Psi \frac{v_b}{z_j} C_{i,s} \qquad \text{(B2-7)}$$

where:

$C_{i,s}$	=	concentration of the "ith" contaminant in the sediment porewater (g m^{-3})
A_s	=	interfacial area between the sediment and water column (m^2)
K_ℓ	=	mass transfer coefficient between the sediment and the water column (m yr^{-1})
ρ	=	density of sediment (g m^{-3})
ν_b	=	sediment burial velocity (m yr^{-1})
σ_k	=	suspended solids concentration in hypolimnion (g m^{-3})
ψ	=	retardation factor (m^3)
k'_D	=	overall sediment sorption coefficient for "ith" contaminant (m^3 g^{-1})
z_j	=	the thickness of the sediment exchange zone of segment "j" (m)
ε	=	porosity of sediment
k_D	=	sorption coefficient for "ith" contaminant on sedimenting organic matter (m^3 g^{-1})
S_k	=	solids settling velocity in the hypolimnion (m yr^{-1})

Equation (B2-7) is the mass balance for the exchange zone of the sediment. The thickness of the zone (z_j) is assumed to be equal to the oxidized (i.e. oxygenated) microzone. This zone is estimated from the sedimental oxygen demand flux. The estimated thickness of the sediment exchange zone may range from less than 1 cm to 15 cm, depending on the oxygen demand. Both the epilimnion

and the hypolimnion were assumed to be in contact with the sediment. The oxidation rate, of course, is constrained to the sediment surface area in touch with the water column.

The approach used in developing the above equations and estimating the parameter is known as Thomann's (1972) "finite section" method. The mass balances and species-specific parameter estimates results in fourteen differential equations that are solved simultaneously.

B2.2 DISPERSIVE / DIFFUSIVE PARAMETERS

Prior to solving the differential equations outlined above, the nominal parameter values for the various exchange processes and their stochastic distributions need to be established (Duever and Reilly 1990). Particularly, the dispersive parameter estimates require close analysis. It has been shown in numerous studies, that the turbulent dispersion coefficient in lakes is scale dependent. This is due to the increasingly large eddies coming into play as two points are further and further apart (Csanady 1973). The horizontal dispersion is generally governed by the 4/3 power law (Thibodeaux 1996), thus the dispersion coefficient is given by the following expression:

$$E = 10^{-4}\ \ell^{4/3} \qquad \text{(B2-8)}$$

where:

E = dispersion coefficient between two points estimated "ℓ" distance apart ($m^2\ s^{-1}$)

ℓ = distance between two points (m)

Equation (B2-8) refers to dispersion between two points. The expression seems to applies from point-to-point distances of one meter to in excess of one hundred kilometres. It is well known, however, that the coefficient of dispersion does not remain constant for a region, but varies from a distance 0 to a distance "ℓ" between two points within connected regions. Thus, to calculate the expected dispersion between two hydraulically connected segments, an averaging of the dispersion is necessary. In the LAKEVIEW model, the line average dispersion coefficient between the midpoints of connected segments is calculated by integral averaging:

$$\overline{E} = \frac{10^{-4}\int_0^1 \ell^{4/3}\, d\ell}{\ell} = \frac{3}{7} \times 10^{-4} \times \ell^{4/3} \qquad \text{(B2-9)}$$

where:

$\overline{E}$ = scale (line) averaged turbulent dispersion coefficient ($m^2\ s^{-1}$)

An important modelling criterion is the assessment of the relative significance of the dispersion with respect to other transport processes. To appraise this relative significance, Schnoor (1996) suggests that the Nusselt number (Nu) be evaluated for a particular water body by the following modified equation:

$$Nu = \frac{Q_o \ell}{A \overline{E}} \qquad \text{(B2-10)}$$

where:

Q_o	=	outflow from the control volume ($m^3 s^{-1}$)
A	=	an average cross-sectional area (depth x width) in the direction of outflow (m^2)
ℓ	=	distance between the midpoints of hydraulically connected segments (m)
$\overline{E}$	=	characteristic turbulent dispersion coefficient between hydraulically connected segments ($m^2 s^{-1}$).

Dispersion becomes important for Nusselt numbers less then unity. It can be easily shown that dispersion dominance is a characteristic feature of pollutant transport from embayments to midlake in most mid to large sized lakes. Once the average dispersion coefficient is obtained, the volumetric (bulk average) dispersion coefficient for a control volume can be evaluated as follows (Chapra and Rechow 1983):

$$E' = \frac{\overline{E} A_c}{\ell} \qquad \text{(B2-11)}$$

where:

E'	=	volumetric dispersion coefficient ($m^3 s^{-1}$)
$\overline{E}$	=	scale average dispersion coefficient between two connected segments ($m^2 s^{-1}$)
A_c	=	cross-sectional area (depth x width) between two connected segments (m^2)
ℓ	=	distance between the midpoints of two connected segments (m).

It is the volumetric dispersion coefficient (E') that is employed in the differential equation given by equation (B2-4). The value calculated as outlined above is the nominal value. For probabilistic calculations, the probability density function for the volumetric dispersion coefficient is assumed to be distributed symmetrically (i.e. mean and the median being the same) about the nominal value.

Although the two processes are structurally similar (see equation (B2-4)), a different methodology is applied for calculating the diffusive exchange between the sediment and the overlying water column. The scale of the transport parameters differs by several orders of magnitude. Lateral transport is orders of magnitude greater than molecular diffusion, while the

vertical transport parameter is the same order of magnitude as molecular diffusion for a given dissolved species in water. First, the effective molecular diffusion coefficient (D_e) is evaluated by the following equation (Dulien 1979):

$$D_e = \frac{\varepsilon D_m}{\tau} \qquad \text{(B2-12)}$$

where:

D_m	=	the molecular diffusion coefficient for a given dissolved species in water ($m^2 s^{-1}$)
ε	=	porosity of the sediment
τ	=	tortuosity

The minimum value of the tortuosity determined from Einstein's relationship is 1.417 in unconsolidated porous media (Cussler 1984) while the maximum realistic value in sediments is about 4.5. In the LAKEVIEW model, a tortuosity factor of 3.14 (i.e. π) is usually employed as a mid-range estimate.

The estimation of the effective diffusion coefficient for dissolved oxygen transport in the sediment is particularly important. Since most metal ions and their stable counterions are more soluble in oxidative environments, the thickness of the sediment exchange zone needs to be established. In concept, this zone is equivalent to the boundary layer concept of Thibodeaux (1996). For zero order or near zero order (i.e. hyperbolic reaction rate expressions with a relatively low half saturation constant) reactions with respect to dissolved oxygen, the thickness of the oxygenated zone can be established from the benthic oxygen demand, the effective diffusion coefficient, and the oxygen concentration at the outer limit of the viscous sublayer above the sediment surface (Scharer *et al*. 1991) as described by the following equation:

$$z_j = 2\frac{D_e C_{O_2}}{J_{O_2}} \qquad \text{(B2-13)}$$

where:

z_j	=	thickness of the sediment exchange zone (m)
D_e	=	effective diffusion coefficient of oxygen ($m^2 s^{-1}$)
C_{O_2}	=	concentration of oxygen at the water sediment interface ($g m^{-3}$)
J_{O_2}	=	benthic oxygen demand ($g m^{-2} s^{-1}$)

The numerical factor of 2 in equation (B2-13) is due to a parabolic oxygen profile resulting from constant effective diffusivity and zero order oxygen consumption in the exchange (reactive)

zone. The oxygen concentration reaches zero at the lower limit of the reactive zone. The benthic oxygen demand ranges from 7.4 $g(O_2)$ m^{-2} yr^{-1} for inorganic reactive tailings under a water cover (Davé *et al*. 1997) to 24 $g(O_2)$ m^{-2} yr^{-1} in cold oligotrophic lakes (Wetzel 1975). Combining these values with an effective oxygen diffusion coefficient of 0.0385 m^2 yr^{-1} and a dissolved oxygen concentration of 12 $g(O_2)$ m^{-3} at the interface, yields a sediment exchange zone thickness (z_j in equation (B2-9) and in equation (B2-13)) of 3.85 cm for oligotrophic lakes to 12.5 cm in reactive tailings deposits. The experimentally observed oxidative zone in reactive tailings was approximately 10 cm (Davé *et al*. 1997). In mesotrophic to eutrophic lakes where the benthic oxygen demand may range up to greater than 100 $g(O_2)$ m^{-2} yr^{-1} in eutrophic systems, the boundary layer thickness is in the order of 1 cm or less resulting in much higher mass transfer rates from the boundary layer (Schnoor 1996).

In the LAKEVIEW model, linear adsorption models are used as the dissolved concentration of the individual species are expected to be 10^{-3} mol L^{-1} or less. The mass transfer coefficient for use in equation (B2-7) takes into account the effective diffusion coefficient (D_e) for a given species, the thickness of the exchange zone (z_j), the porosity of the exchange zone (ε), the solid-liquid distribution coefficient (k_d), and the density of the sediment in the exchange zone (ρ) as follows:

(B2-14)

$$K_\ell = \frac{D_e \, \varepsilon}{z_j \left[\varepsilon + (1 - \varepsilon)\, k'_d \, \rho\right]}$$

where:

K_ℓ	=	mass transfer coefficient in the sediment (m s^{-1})
ε	=	porosity of sediment
D_e	=	effective diffusion coefficient (m^2 s^{-1})
ρ	=	solids density (kg m^{-3})
k'_d	=	overall sediment sorption coefficient for the "ith" contaminant (m^3 kg^{-1})
z_j	=	thickness of the sediment exchange zone (m)

Molecular diffusion coefficients, hence, effective diffusion coefficients are evaluated as a function of temperature (4°C in deep water bodies).

The nominal values of the lake specific parameter values are summarized in Table B2.2-1.

TABLE B2.2-1
YELLOWKNIFE BAY MODEL INPUT PARAMETERS

Parameter Symbol	Description	Units	Nominal Value
V_T	Volume of Lake Segment: - Segment 1: Back Bay - Segment 2: Yellowknife Bay - Segment 3: Yellowknife Bay	 m^3 m^3 m^3	 $1.73x10^7$ $4.40x10^7$ $1.43x10^8$
Z	Mean Depth of Segment: - Segment 1: Back Bay - Segment 2: Yellowknife Bay - Segment 3: Yellowknife Bay	 m m m	 6.9 8.3 11.2
A	Surface Area of Segment: - Segment 1: Back Bay - Segment 2: Yellowknife Bay - Segment 3: Yellowknife Bay	 km^2 km^2 km^2	 2.51 5.30 12.73
ℓ	Distance Between Segment Midpoints: - Segment 1 and 2 - Segment 2 and 3	 km km	 1.3 7.6
A_j	Cross-sectional Area Between Segments: - Segment 1 and 2 - Segment 2 and 3	 m^2 m^2	 1.81×10^4 1.14×10^4
Q_j	Freshwater Inflow to Segment: - Segment 1 Back Bay - Segment 2 Yellowknife Bay - Segment 3 Yellowknife Bay	 $m^3\ yr^{-1}$ $m^3\ yr^{-1}$ $m^3\ yr^{-1}$	 5.66×10^6 1.126×10^9 -
k_D	Liquid-to-Solid Partition Coefficient for Arsenic Removal on Suspended Organic Matter in Segment: - Segment 1: Back Bay - Segment 2: Yellowknife Bay - Segment 3: Yellowknife Bay	 $m^3\ kg^{-1}$ $m^3\ kg^{-1}$ $m^3\ kg^{-1}$	 5 5 5
k_D[1]	Overall Sediment Sorption Coefficient: - Segment 1 Back Bay - Segment 2 Yellowknife Bay - Segment 3 Yellowknife Bay	 $m^3\ kg^{-1}$ $m^3\ kg^{-1}$ $m^3\ kg^{-1}$	 0.5 0.5 0.5
ε	Porosity of Lake Sediment	-	0.85
S_j	Solids Settling Velocity in Lake Water Column	$m.yr^{-1}$	36.5
σ_j	Sediment Solids Concentration	$g\ m^{-3}$	2
ρ	Sediment Dry Density	$kg\ m^{-3}$	1,500
z_j	Thickness of Sediment Exchange Zone	cm	3

B2.3 COMPUTATIONAL PROCEDURE

The LAKEVIEW model comprises a defined number of connected segments. A segment comprises a water body of a given area and means depth and the underlying sediment exchange zone. The change of water quality in each segment is formulated by a pair ordinary differential equations (one for the water body and one for the sediment) such as equation (B2-4) and equation (B2-7). These differential equations are assumed to have the following structure:

$$\frac{dC_{i,j}}{dt} = \{\omega_i\} + \{a_{i,j}\} \text{x } \{C_{i,j}\} \qquad \text{(B2-15)}$$

where:

$C_{i,j}$	=	dependent variable (concentrations)
t	=	independent variable (time)
ω_i	=	mass loading (time dependent)
$a_{i,j}$	=	constants
{ }	=	matrix (vector) designation

The left-hand side of equation (B2-15) is a "n x 1" vector of the differentials. The term $\{\omega_i\}$ is a time dependent vector of the normalized mass loadings to each segment. The term $\{a_{i,j}\}$ designates an "n x n" sparse matrix of inverse time constants, while $\{C_i\}$ is the local concentration vector. It can be shown, that the series of equations may be solved analytically in the following format (Wiley 1960):

$$\{C_{i,j,}\ (t + \Delta t\} = \overline{C_{i,j}} + \left[\left(C_{i,j}(t) - \overline{C_{i,j}} \right) \sum_{k=1}^{n} \alpha_k \ \exp\left(-\lambda_k \ \Delta t\right) \right] \qquad \text{(B2-16)}$$

where:

$C_{i,j}(t)$	=	concentration in the "ith" contaminant in the "jth" segment at time t
$C_{i,j}(t + \Delta t)$	=	concentration in the "ith" contaminant in the "jth" segment at t+Δt
$\overline{C_{i,j}}$	=	theoretical "steady state" "ith" concentration in the 'jth' segment
α_k	=	partitioning coefficient
λ_k	=	eigenvalues
{ }	=	matrix elements

In equation (B2-16), $\overline{C_{i,j}}$ represents a theoretical steady state concentration provided the external mass load, ω_i, remains invariant between t=t and t=∞. This concentration, however, is never reached in practice, since the load varies continually. Thus, the numerical values of $\overline{C_{i,j}}$ were derived using Cramer's method as follows:

$$\overline{C_{i,j}} = \frac{\left|\omega_i, a_{k,j}\right|}{\left|a_{i,j}\right|} \qquad \text{(B2-17)}$$

where:

$|\omega_i, a_{k,j}|$ = determinant of matrix with column vector $\{\omega_i\}$ substituting for matrix elements $a_{k,i}$

$|a_{k,j}|$ = determinant of the original matrix

Next, the eigenvalues of the matrix, $\{a_{i,j}\}$, are obtained. Since the matrix is sparse, further reduction of the matrix is achieved in the following manner. Horizontal dispersion between connecting segments is the only major two-way exchange mechanism between adjacent water bodies, hence the dispersive flow into the segment's water body is treated as an additional temporal (i.e. time dependent) load:

$$\overline{C}^*_{i,j} = \overline{C}_{i,j} + \sum_k \frac{a_{i,k}\left(\overline{C}_{k,j} - \overline{C}_{k,j}\right)\Delta t}{V_j} \qquad \text{(B2-18)}$$

where:

$\overline{C^*_{i,j}}$ = adjusted pseudo-steady state concentration

$a_{i,k}\left(\overline{C}_{k,j} - \overline{C}_{k,j}\right)\Delta t$ = "load" resulting from dispersive flow from segment "k"

Unlike the effluent load to the segments, this dispersive load is adjusted in every time step (Δt). Thus, the original $\{a_{i,j}\}$ matrix is approximated by a set 2 x 2 sub-matrices (altogether (½) x n sub-matrices) – each sub-matrix representing the water column and the underlying sediment. In the case of summer stratification, the matrix is resolved into a pair of 2 x 2 sub-matrices per segment; one sub-matrix for the epilimnion/hypolimnion and another sub-matrix for the hypolimnion/sediment interaction. The eigenvalues of each sub-matrix are found by conventional second order root finding methods:

$$(a_{i,i} + \lambda)(a_{i+1,i+1} + \lambda) - (a_{i,i+1}\, a_{i+1,i}) = 0\,; \quad i = 1, 3, 5, 7 \text{ etc} \qquad \text{(B2-19)}$$

$$a_{i,i} \geq a_{i,i+1} \quad ; \quad a_{i+1,i+1} \geq a_{i+1,i}$$

The inequalities shown above for the diagonal elements ensured that the eigenvalues (λ_k) were negative thereby guaranteeing convergence. The eigenvalues, are independent of the load and depend only on the outflow, the local exchange, speciation and the reactivity of the particular contaminant species.

To obtain numerical values for the partitioning coefficients, α_k, the normalized form of equation (B2-16) was used:

$$\frac{C_{i,j}\,(t+\Delta t)-\overline{C^*_{i,j}}}{C_{i,j}(t)-\overline{C^*_{i,j}}}=\sum_k\ \alpha_k\ \exp(-\lambda_k\ \Delta t) \qquad \text{(B2-20)}$$

and

$$\sum \alpha_k\ =1 \qquad \text{(B2-21)}$$

Basically, the right side of equation (B2-20) is substituted for $C_{i,j}$ in equation (B2-15) and the values of "α_k" are derived from algebraic equations after collecting and equating terms for each exponential ($\exp(-\lambda_k\,\Delta t)$). To ensure accuracy, the program is run at variable time steps, Δt. The time step is progressively reduced until a desired convergence is reached.

The computer-based code does not allow the convenient use of standard mathematical libraries. Therefore, all matrix and other operations are programmed as subroutines of the source code. To test the validity of the code and verify the correctness of the operations, the predictions of the revised model are compared with numerical integral solutions generated by orthogonal collocation. For all practical purposes, the predictions are identical for surface waters and very slight differences were noted for sediments.

The water quality predictions are performed at yearly intervals. The dynamics of the system are such that near steady state concentrations (i.e. $C_{i,j}\,(t) \approx C_{i,j}\,(t+\Delta t) \approx \overline{C^*_{i,j}}$) are generated in the epilimnion after a few years while the dynamic response of the sediments contain considerable inertia (i.e. significant time lag between $C_{i,j}\,(t)$ and $\overline{C^*_{i,j}}$). These are expected from an "*a priori*" casual examination of arsenic reservoir in the sediments and the inverse time matrix, $\{a_{i,j}\}$.

B3.0 SEDIMENT CHEMISTRY COMPONENT

B3.1 ARSENIC SPECIATION IN SEDIMENT

Soluble arsenic species have two common oxidation states: +3 and +5. Other arsenic compounds such as reduced arsine compounds (oxidation state = -3) and elemental arsenic (oxidation state = 0) are too unstable in natural environments and are readily converted to the more stable forms. Several stable solid phases of arsenic (Faust and Aly, 1981) include arsenious oxide (As_2O_3), arsenic (V) oxide (As_2O_5), arsenopyrite (FeAsS), arsenious sulphide (As_2S_3), arsenic sulphide (As_2S_5), calcium arsenate ($Ca_3(AsO_4)_2$), magnesium arsenate ($Mg_3(AsO_4)_2$) and co-precipitates (Paige *et al*., 1996a, 1996b, 1997) with iron(III) oxyhydroxides (FeOOH $xFeAsO_3$, $FeOOH \cdot x\ FeAsO_4$). In oxygenated waters and at near neutral pH, dissolved arsenic usually occurs as arsenate species ($HAsO_4^{2-}$, $H_2AsO_4^-$). Under anoxic conditions, arsenic (V) is reduced to arsenic (III) and is expected to be present as a hydroxo (H_3AsO_3, $H_2AsO_3^-$) or sulphide (AsS_2^-) ligand (Faust and Aly, 1981).

The solubility of arsenic (III) compounds is substantially less than the solubility of arsenic (V) compounds. The reduction from arsenic (V) to arsenic (III) is kinetically controlled and it is facilitated by the presence of high organic content and reducing bacteria (Harrington *et al*., 1998). In an anoxic environment, the kinetics of reduction are sufficiently fast so that dissolved arsenic is effectively removed from the sediment porewater as it precipitates as a solid phase, most likely as arsenic sulphide (As_2S_3). This process may be modelled either by a first order rate constant representing "disappearance" from the liquid porewater phase or by introducing a pseudo-equilibrium K_d value incorporating a near equilibrium removal process. The later approach is particularly useful if the reduction of arsenic is incomplete. Consequently, high removal efficiency of arsenic may be realised in wetlands and bogs. The disadvantage of this approach is that the removal needs to be calibrated for each system.

The partitioning of metal species between aqueous and secondary solid phases is based on two processes, chemical equilibrium and adsorption process. The chemical equilibrium involves solid - liquid equilibrium. The solids are modelled implicitly, i.e. all secondary solids for each are stored as a single variable.

B3.2 ARSENIC SOLID SOLUTIONS AND COPRECIPITATION

Coprecipitation involves the formation of a host solid (precipitate) and the entrapment of a "trace" metal or ligand within the parent solid (Appelo and Postma, 1993). Often, coprecipitate forms when a solution is supersaturated with respect to the host mineral, but undersaturated with respect to the compounds of the trace element. Coprecipitatation must have favourable thermodynamics and kinetics. Well known solid solutions are the coprecipitation of radium sulphate with either calcium ($Ca_{1-x}Ra_xSO_4$) or barium sulphate ($Ba_{1-x}Ra_xSO_4$), and the formation of lead and lead-210 compounds ($Pb_{1-x}Pb_x^{210}SO_4$, for example). Magnesium co-precipitates with gypsum forming a calcium-magnesium sulphate solid solution $Ca_{1-x}Mg_xSO_4$ (Gamsjäger 1989). High levels of both arsenic and iron oxides in the sediment of Back Bay and Yellowknife Bay are indicators of contamination from mining operations (Murdoch at al., 1989). Ferric arsenate interacts with ferric hydroxide forming a precipitate, $xFeAsO_4 \bullet (1\text{-}x)Fe(OH)_3$ resembling solid solutions (Harris and Krause, 1994; Paige *et al*., 1996a,b, 1997).

For binary solids that form regular but non-ideal precipitates, the solubility product obeying the Gibbs-Duhem criterion is the following (Gresens, 1981):

$$K_S^0 = K_{S,1}^0 \cdot x \cdot e^{a(1-x^2)} + K_{S,2}^0 \cdot (1-x) \cdot e^{ax^2} \qquad \text{(B3-1)}$$

where:

K_S^0	=	solubility product of solid solution
$K_{S,1}^0$	=	solubility product of the pure minor (trace) component
$K_{S,2}^0$	=	solubility product of the pure major component
x	=	mole fraction of the minor component in the solid
a	=	coefficient of non-ideality

The mole fraction, x, is estimated considering the total amount (sum of liquid and solid phase) of components in the system. The coefficient of non-ideality is based on published data. All solubility products are defined as activity (rather than concentration) products and extrapolated to zero ionic strength. For simple binary solid solutions ($M_{1,x}M_{2,1-x}L_1$) the activity product becomes:

$$K_S^0 = \left[\left\{M_1^{n+}\right\} + \left\{M_2^{n+}\right\}\right] \cdot \left\{L_1^{n-}\right\} \qquad \text{(B3-2)}$$

where:

$\{M_1^{n+}\}$	=	activity of the minor ion component
$\{M_2^{n+}\}$	=	activity of the major ion component
$\{L_1^{n-}\}$	=	activity of the ligand (anion)
n	=	ion valence number

When the molar concentration of the minor component is exceedingly low ($x \approx 0$) such as radionuclides dissolving in the host mineral ($RaSO_4$ in $CaSO_4$,, for example), equation (B3-1) and equation (B3-2) can be simplified to yield:

$$\frac{\{M_1^{n+}\}}{\{M_2^{n+}\}} = |a| \frac{K_{S,1}^0}{K_{S,2}^0} x \tag{B3-3}$$

The term before the mole fraction on the right side of equation (B3-3) is often called the distribution coefficient (D). Knowing the activity of the major ion in the liquid phase and the mole fraction, the activity of the minor component in the liquid phase can be readily calculated.

For the $FeAsO_4$ / $Fe(OH)_3$ system, the overall mole fraction (x) of $FeAsO_4$, is given by:

$$x = \frac{[As]_T}{[Fe(III)]_T} \tag{B3-4}$$

where:

x	=	mole fraction
$[As]_T$	=	total concentration (solid and liquid phase) of arsenic (mol m^{-3} sediment)
[Fe(III)]	=	total concentration (solid and liquid phase) of ferric iron (mol m^{-3} sediment)

Combining equations (B3-3) and (B3-4), the liquid phase arsenic concentration becomes:

$$[As]_{T,L} = \frac{K_{S,1}^0 \cdot x^2 \cdot e^{a(1-x^2)} + K_{S,2}^0 \cdot x \cdot (1-x) \cdot e^{ax^2}}{f_{Fe^{3+}} f_{AsO_4^{3-}} F_{Fe} F_{As}} \tag{B3-5}$$

where:

$[As]_{T,L}$	=	concentration of arsenic in the liquid phase (mol m^{-3})

$f_{Fe^{3+}}$, $f_{AsO_4^{3-1}}$ = activity coefficients

$$F_{Fe} = \frac{10^{(12-3pH)}}{1+10^{(6.33-pH)}+10^{(9.82-2pH)}+10^{(12-3pH)}} \quad \text{(B3-6)}$$

$$F_{As} = \frac{10^{(3pH-20.59)}}{1+10^{(pH-2.24)}+10^{(2pH-9.1)}+10^{(3pH-20.59)}} \quad \text{(B3-7)}$$

$$K_{S,1}^{0} = 10^{-21.6}\,(\text{mol m}^{-3})$$

$$K_{S,2}^{0} = 10^{-33}\,(\text{mol m}^{-3})$$

$$a = -1.48$$

B3.3 ARSENIC SURFACE ADSORPTION IN SEDIMENT

Under certain conditions, the concentration of some ligands such as arsenates may be controlled by sorption onto solid ferric hydroxide, aluminum hydroxide, quartz, and organic surfaces. Both arsenates and arsenites are readily adsorbed to ferric hydroxide and aluminum hydroxide surfaces. The sorption coefficient (k_D) is defined as follows (Sheppard and Thibault, 1990):

$$k_D = \frac{[As]_S}{[As]_L} \quad \text{(B3-8)}$$

where:

k_D = sorption coefficient ($m^3\,kg^{-1}$)
$[As]_L$ = concentration of dissolved arsenic in the liquid phase ($g\,m^{-3}$)
$[As]_S$ = concentration of surface adsorbed arsenic ($g\,kg^{-1}$)

The model employs three separate adsorption coefficients to calculate the amount of arsenic adsorbed onto the three adsorbents. Thus, in addition to co-precipitation, adsorption onto ferric hydroxide, aluminum hydroxide, quartz and organic surfaces is evaluated. However, besides ferric and aluminum hydroxides, the sorption (K_d) of arsenic to either organic or inorganic solids is not particularly high (Sheppard and Thibault, 1990). Consequently, different k_D values are used to represent arsenic sorption to settling solids, the sediment overall, and ferric hydroxide in the sediment. In the sediment porewater, the dissolved concentration of arsenate is given by:

$$[As]_L = \frac{[As]_T}{\varepsilon + k_D(1-\varepsilon)\rho} \qquad \text{(B3-9)}$$

where:

$[As]_L$	=	concentration of dissolved arsenic in the porewater (g m^{-3})
$[As]_T$	=	concentration of total arsenic (liquid and solid phase) (g m^{-3})
k_D	=	sorption coefficient (m^3 kg^{-1})
ε	=	liquid filled porosity in the sediment
ρ	=	sediment solids density (kg m^{-3})

B4.0 PARAMETER ESTIMATION

Parameter estimation from data is often an important task to be undertaken prior to predictive simulations. Often, parameters are not known precisely at the onset of the modeling but must be estimated from available data. Several regression techniques, both linear and non-linear, are available to estimate parameter values for algebraic expression. However, these techniques break down if the parameters are defined on the differential level, but observations need to be compared with integral values (see Equation B4-1). Under these circumstances the Metropolis-Hastings method may be employed. This method is applicable to observations that conform to a Markov chain.

Let us assume that $y_1, y_2, y_3 \ldots y_{n-1}, y_n$ are independent observations that have been collected in a temporal or physical space. These observations are said to form a Markov chain if the following conditional probability is obeyed:

$$P(y_n | y_1, y_2, \ldots y_{n-2}, y_{n-1}) = P(y_n | y_{n-1}) \qquad \text{(B4-1)}$$

Thus the probability of obtaining the current observation (i.e. observation "n") is conditional on only the previous (i.e. penultimate) observation ("n-1"). A number of data sets resulting from natural processes such as growth, death, reaction products etc., are in fact Markov chain observations. If one wishes to develop a model to represent these observations, these models will contain a set of parameters θ (i.e. θ_1, θ_2, θ_3 etc.) that needs to be estimated given the set of observations, y. The Bayesian inference of model parameters θ given the observed data involves the joint posterior probability distribution conditional on the data: $P(\theta \mid y)$. According to Bayes' theorem, the joint probability of θ given y is:

$$P(\theta \mid y) \propto P(\theta)\, P(y \mid \theta) \qquad \text{(B4-2)}$$

Where:

$P(\theta)$ = prior distribution of the parameters
$P(y \mid \theta)$ = likelihood function conditional on θ.

The Metropolis-Hastings algorithm can be summarized as follows (Gilks *et al*., 1996):

a) Assign prior probability distribution $J(\theta_n \mid \theta_{n-1})$ for each parameter to be estimated.
b) Obtain $\theta_{(0)}$ as the initial realization of the parameter vector.
c) Sample the proposed distribution $J(\theta_n \mid \theta_{n-1})$ to obtain parameter value θ*.
d) Calculate

$$r = \frac{P(\theta^* \mid y) / J(\theta^* \mid \theta_{n-1})}{P(\theta_{n-1} \mid y) / J(\theta_{n-1} \mid \theta^*)} \qquad \text{(B4-3)}$$

set $\theta_n = \theta^*$ with probability min (r, 1)

$\theta_n = \theta_{n-1}$ otherwise

e) Continue obtaining realizations of θ_n for n = 1, 2, 3, N until $P(\theta_n \mid y)$ converges to $P(\theta \mid y)$.

In the LAKEVIEW model, each parameter to be estimated is assigned a triangular *apriori* distribution, $J(\theta_n|\theta_{n-1})$, containing the maximum, mean (most likely) and the minimum possible values. The mean value is established either by graphical means or "eyeballing" the experimental data. The obvious advantage of choosing a symmetric prior distribution such as the triangular distribution is that $J(\theta^*|\theta_{n-1}) = J(\theta_{n-1}|\theta^*)$. Consequently "*r*" reduces to:

$$r = \frac{P(\theta^* \mid y)}{P(\theta_{n-1} \mid y)} \qquad \text{(B4-4)}$$

The parameter distribution space is sampled and a given parameter value is assigned by a Monte Carlo draw. Using the sampled value, the predicted concentrations are computed by integrating the differential equations. The predictions are then compared with measurements by calculating the normalized sum of squares between the observations and predictions. In LAKEVIEW, the normalized squared differences are calculated between predictions and observations for surface and/or porewater. The parameter sampling procedure is repeated a number of times. Both the new parameter and its value are chosen by random draw using the Monte Carlo technique. The most likely set of parameters is found after numerous (at least 50) trials. The calculation procedure is detailed below.

Parameter estimation for modeling the porewater sulphate concentration (equations B2-6 and B2-7) is used as an example of the calculation procedure:

$$\begin{aligned} \Psi &= ((1-\varepsilon)\rho\, k'_D + \varepsilon) A_S z_j \\ \Psi \frac{dC_{i,S}}{dt} &= K_l A_S (C_{i,k} - C_{i,s}) + \sigma_k\, k_D\, S_k\, C_{i,k}\, A_s - \Psi \frac{\nu_b}{z_i} C_{i,S} \end{aligned} \qquad \text{(B4-5)}$$

The parameters to be estimated directly in this equation include the mass transfer coefficient (K_l) and the sediment burial rate (v). The temporal concentration profile also depends on retardation factor (Ψ). For calculating Ψ, in turn, the knowledge of the sorption coefficient (k'_D) is also needed. Thus, the three parameters to be established for estimating the total biomass are: K_l, v_b, and k'_D). Let θ_1, θ_2, and θ_3 be the parameter estimates for K_l, v_b, and k'_D, respectively. Since the

parameter values (θ_1, θ_2, and θ_3) are not known exactly, each parameter "i" ($\theta_i \geq 0$) is assigned an equilateral triangular distribution. For example, the minimum ($\theta_{i,min}$) and maximum ($\theta_{i,max}$) delimit the range for parameter θ_i. The range is chosen such that the mean parameter value ($\theta_{i,mean}=0.5[\theta_{i,min} + \theta_{i,max}]$) is the expected value, $E(\theta_i)$ that has been established by previous (or preliminary) considerations. The triangular distribution density function is normalized by calculating the apex (height of the triangle, h_i) as follows:

$$h_i = \frac{2}{\theta_{i,max} - \theta_{i,min}} \qquad \text{(B4-6)}$$

The h_i value assures that the area delimited by the triangle (i.e. 0.5 x ($\theta_{i,max}$ - $\theta_{i,min}$) x h_I) equals to unity. The parameter sampling procedure begins by establishing the parameter type (i = 1,2 or 3…) by random draw. The sampling is based on generating an unobservable variable δ_i (Min, 1998) also by random draw, such that $\left|\pm \delta_i\right| \leq \frac{(\theta_{i,max} - \theta_{i,min})}{2}$ and a new estimate of the parameter $\theta^* = E(\theta_i) + \delta_i$. The sign of δ, of course, depends on the random number representing the integral (i.e. area) under the triangular distribution density function ($\delta_i<0$ for area <0.5).

The objective of the parameter estimation is to minimize the sum of squared differences between predicted and observed dependent variables. A "benchmark" conditional model probability is calculated after substituting the expected parameter values $\theta_{mean} = E(\theta_{Ii})$ into the differential equations and integrating to give the predicted dependent variable $y(\theta_{mean},t)$:

$$p\left(y|\theta_{mean}\right) \propto exp - \frac{\Sigma\left[y|\left(\theta_{mean}, t_k\right) - y(t_k)\right]^2}{2\, n_t\, \sigma^2} \qquad \text{(B4-7)}$$

where:

$p(y\|\theta_{,mean})$	=	conditional "baseline" model probability
$y(\theta_{mean},t_k)$	=	model prediction using expected parameter values at time t_k
$y(t_k)$	=	observation taken at time t_k
n_t	=	total number of observations
σ^2	=	variance

Using Bayesian inference, the posterior probability is given by:

$$P(\theta_{mean} \mid y) \propto p(\theta_{mean}) \times p(y|\theta_{mean}) \qquad \text{(B4-8)}$$

where:

$P(\theta_{mean} \mid y)$ = "benchmark" predictive model probability for a given set of observations, y_k

$p(\theta_{,mean})$ = the probability of selecting parameter set $\theta_{,mean}$

It is evident that minimizing the sum of squared differences between predictions and observations implies maximizing the probability $P(\theta_{mean} \mid y)$. The posterior probability is re-calculated after each parameter selection.

The sampling space consists of a *n* trials and a maximum of *m* iterations per trial. This leads to a maximum of *n* x *m* parameter selection. The number of trials should be at least 50 resulting in a set (θ_1, θ_2, θ_3 etc.) of 50 parameter estimates. The number of iterations is based on the number of parameters to be estimated. As a rule of thumb, the number of draws should be about 10 for each parameter. Thus, the total number of iterations, *m*, should be about tenfold the number of parameters to be estimated. In the present case, the value of *m* for the three-parameter system is 30.

Trial 1 commences with the calculation of the benchmark (or baseline probability) with the expected parameter values, $E(\theta_i)$-s. Then the selection of the parameter, say parameter "i" and its value (θ_i) is performed by random draw as described above. Model probability with the selected value, θ_i is given as:

$$P(\theta_i \mid y) = p(\theta i \mid \delta_{i,}\theta) \text{ x } p(y \mid \theta_i, \theta) \tag{B4-9}$$

where:

$$p\left(y \mid \theta_i, \theta\right) \propto exp - \frac{\Sigma\left[y \mid (\theta_i, \theta, t_k) - y(t_k)\right]^2}{2\, n_t\, \sigma^2} \tag{B4-10}$$

where:

θ_I = most recently drawn parameter value

It should be noted that the Metropolis- Hastings algorithm is quite flexible and the posterior probability function (equation B4-7) can also be defined as:

$$p\left(y \mid \theta_i, \theta\right) \propto exp - \frac{\Sigma\left[\ln \frac{y \mid (\theta_i, \theta, t_k)}{y(t_k)}\right]^2}{2\, n_t\, \sigma^2} \tag{B4-11}$$

If the probability min{r, 1} for equation (B4-3) is satisfied with respect to the benchmark value in trial 1, then $P(\theta_i| y)$ becomes a "first" estimate of a "global probability maximum", $P(\theta^g |y)$.

In practice, the variance, σ^2, is not known. An estimate of σ^2 is obtained from the "previous best" $p(y|\theta_i, \theta)$ value that resulted in the acceptance of θ_i in a following manner:

$$\sigma^2 \approx s^2 = \frac{\Sigma[y|(\theta_i, \theta, t_k) - y(t_k)]^2}{n_t} \qquad \text{(B4-12)}$$

It can be shown that the estimated variance σ^2 is X^2 distributed.

After storing the accepted parameter values, the parameters are reset to their expectations ($E(\theta)$) and a new trial begins. Alternatively, the selected parameter value is replaced and the draw may continue until the number of iterations, *n*, is exceeded. At this point the parameters are also reset to $E(\theta)$-s and a new trial begins. In subsequent trials, if the probability $P(\theta_i| y)$ is greater than the bench mark probability, but less than the current global probability, then the drawn parameter θ_i is retained and is used in subsequent iterations within the trial as a starter value. If, however, the probability $P(\theta_I | y)$ is less than the benchmark probability then θ_i is replaced by previous estimate. The iteration continues until either $P(\theta_i| y)$ becomes greater than then the "current global maximum", $P(\theta^g|y)$, or the number of iterations exceeds "*m*". In the former case, the iteration stops, the parameter values are stored, and $P(\theta^*|y)$ is assigned as the new $P(\theta^g | y)$. In either case, the parameters are reset to $E(\theta)$ and a new trial search commences until the maximum number of trials exceeds "*n*". Using this method, each successful trial results in a unique set of parameter values in which at least one of the parameters differs in value from the same parameter type in previous trials. An accelerated search technique allows the parameters corresponding to interim $P(\theta |y)$ to become the benchmark parameters (new $E(\theta_i)$-s by centering the triangular distribution on the best current estimate of θ^g) after a certain number of unsuccessful trials (i.e. trials in which the number of iterations exceeded "*n*"). If needed, the triangular parameter range (i.e. θ_{max} and θ_{min}) is adjusted to maintain symmetry. This adjustment becomes necessary whenever the current best parameter value approaches but cannot be less than zero.

A difficult problem remains the need to estimate the marginal posterior density for the parameter set θ:

$$P(\theta| y(t_k)\} = \int p\{\theta| \sigma,\delta, y(t_k)) \, d\delta \qquad \text{(B4-13)}$$

It is well known that the posterior density for θ possesses a multivariate normal distribution, but the performance of the integration analytically is far too difficult (Min, 1998). An acceptable empirical solution is to save the set of θ-s after the last ten or so trials. The argument is that, if the maximum trial number "*n*" is chosen large enough, then the parameters have converged (or nearly so) to their best estimate by trial *n-l* so the variance may be calculated empirically as follows:

$$\sigma_i^2 = \frac{\sum_{l} \{\theta_{i,j} - \theta_{i,mean})^2}{\ell - 1} \qquad \text{(B4-14)}$$

where:

σ_i^2	=	variance of parameter "i"
$\theta_{i,j}$	=	the parameter value of type "i" in the j th data set (*n-k* <= *j* <= *n*)
$\theta_{i,mean}$	=	the average value of parameter "i"
ℓ	=	total number of parameter sets in $n - \ell$ *to n* number of trials

B5.0 HISTORICAL WATER AND SEDIMENT QUALITY

B5.1 ARSENIC CONCENTRATIONS IN WATER

In preparing this report, surface water quality data for grab samples collected at the outlet of Baker Creek, various locations in Back Bay, and in Yellowknife Bay both north and south of Latham Island, were reviewed and assessed. It was determined that the water quality data included in the reports do not provide a complete database as there are years for which there are no data. The assessment of the historical water quality arsenic concentration data for the Baker Creek outlet, for Back Bay and for the north and south parts of Yellowknife Bay was useful, as it indicated general conditions and trends. More specifically;

- Arsenic concentrations in the outflow from Baker Creek have fallen from levels identified in the 1970's. The most recent data (1999) suggest that the arsenic concentrations in the Baker Creek outlet remain above arsenic concentration guidelines for drinking water supply and the protection of freshwater aquatic life.

- The arsenic concentration data for Back Bay includes data from surface grab samples, and sampling of various strata to investigate spatial and temporal effects. As a general observation, data for 1987-88 suggest that there was a reduction in the arsenic concentration of surface water in Back Bay in comparison to 1972 sampling results. Arsenic concentration data for water samples collected between 1972 and 1994 in Yellowknife Bay, north of Latham Island, also suggest that arsenic concentrations have fallen from the 1972 levels.

HydroQual (1989) identified the effluent from the Giant Mine gold operation discharged through Baker Creek as the principal source of metal contamination to Back Bay and Yellowknife Bay, in addition to surface run-off and atmospheric deposition. Average Giant Mine tailings decant arsenic concentrations and annual arsenic loadings to Baker Creek are presented in Table B5.1-1.

TABLE B5.1-1
AVERAGE GIANT MINE TAILINGS DECANT CONCENTRATIONS AND ESTIMATED ANNUAL LOADINGS TO BAKER CREEK

Year	1978 [(A)]	1987 [(A)]	1991 [(B)]	1992 [(B)]	1993 [(B)]
Average Arsenic Concentration of Tailings Decant	21.25 mg/L	0.38 mg/L	0.39 mg/L	0.58 mg/L	0.35 mg/L
Arsenic Loading	15,450 kg/y	498 kg/y	956 kg/y	1,237 kg/y	1,098 kg/y

[(A)] Source: Table 3.4.2, HydroQual (1989).
[(B)] Source: Table 2, Jackson *et al.* (1996).

Baker Creek Outlet Water Quality:

Moore *et al*. (1978) reported that when gold production at the Giant Mine commenced in 1948, tailings were subaerially deposited east of the mill. From 1951 to 1968, tailings were deposited into a lake located north of the mill. During this period of time, tailings water drained to Baker Creek, and hence to Back Bay, as well as to the head of Yellowknife Bay – the latter flow stopped in 1968.

Up to 1981, arsenic removal was accomplished by settling and precipitation of metal hydroxides in the tailings pond (Moore *et al*. 1978). In 1981, a chemical treatment plant was commissioned to treat tailings pond overflow prior to its seasonal discharge to Baker Creek.

For the last several years discharge to Baker Creek has been limited to the summer months. Therefore there is a wide variation in the measured water quality data depending on the time of year the sample was collected.

Water quality data with respect to arsenic concentrations in samples collected at the outlet of Baker Creek for selected dates from 1972 to 1999 are presented in Table B5.1-2. As a general observation, measured total arsenic concentrations in water samples collected from the outlet of Baker Creek have typically exceeded the Health Canada (2004) proposed drinking water supply guideline of 5 µg As/L and the CCME (2002) freshwater aquatic life guideline of 5 µg As/L.

TABLE B5.1-2
ARSENIC CONCENTRATIONS IN SELECTED SURFACE WATER SAMPLES COLLECTED AT THE OUTLET OF BAKER CREEK (1972-2004)

Date Sampled	Total Arsenic Concentration µg/L	Data Source
April 30, 1972	12,600	Falk *et al*. 1973 (Station GM-1)
May 30, 1972	3,750	
June 14, 1972	2,950	
September 14, 1972	5,800	
December 10, 1974	500	HydroQual 1989 (Station 901)
March 26, 1975	300	HydroQual 1989 (Station 901)
May 28, 1975	850	
September 18, 1975	5,000	
December 18, 1975	70	

TABLE B5.1-2 (Cont'd)
ARSENIC CONCENTRATIONS IN SELECTED SURFACE WATER SAMPLES COLLECTED AT THE OUTLET OF BAKER CREEK (1972-2004)

Date Sampled	Total Arsenic Concentration µg/L	Data Source
February 17, 1976	2,060	HydroQual 1989 (Station 901) and Moore *et al.* 1978 (Station 1)
May 3, 1976	400	
September 13, 1976	9,100	HydroQual 1989 (Station 901) and Moore *et al.* 1978 (Station 1)
May 12, 1977	1,000	HydroQual 1989 (Station 901)
June 6, 1977	1,740	
June 5, 1987	1,400	HydroQual 1989 (Station 901)
October 8, 1987	21	
May 9, 1988	312	HydroQual 1989 (Station 901)
June 18, 1988	39	
September 1992	5.7	Jackson *et al*. 1998 (Station: Baker Creek)
March 1993	0.4	
June 1993	247	
August 1993	170	
February 1994	0.9	
March 1994	0.3	
8 June 1994	130	
14 June 1994	93.4	
21 June 1994	191	
29 June 1994	153	
5 July 1994	300	
13 July 1994	26.3	
20 July 1994	8.6	
27 July 1994	315	
8 August 1994	208	
17 August 1994	222	
6 September 1994	265	
23 September 1994	32.7	
Summer 1997	102	Mace 1998 (Stations 20, 18 and 22)
Summer 1997	276	
Summer 1997	114	
June 28, 1999*	175*	Lorax 1999 (Station 43-5)
July 18, 1999*	182*	
October 2001	66	Dillon 2002a (Stations BC-3 and BC-4)
October 2001	68	
June 2002	20.4	Dillon 2002b (Stations BC-3 and BC-4)

TABLE B5.1-2 (Cont'd)
ARSENIC CONCENTRATIONS IN SELECTED SURFACE WATER SAMPLES COLLECTED AT THE OUTLET OF BAKER CREEK (1972-2004)

Date Sampled	Total Arsenic Concentration µg/L	Data Source
June 2002	60.3	
July 2002	232	
July 2002	216	
November 2002	20.4	Dillon 2002c (Site 3 and 4 downstream)
November 2002	232	
November 2002	60.3	
November 2002	216	
March 2004	204	Dillon 2004 (Site 5, 6 and 7)
March 2004	218	
March 2004	203	
March 2004	233	
March 2004	107	
March 2004	174	

Note:
* Labile Arsenic.

Back Bay Water Quality:

Back Bay is located in the northwest section of Yellowknife Bay, west of Latham Island. Baker Creek discharges into Back Bay, in addition, the water quality will be influenced by the historical placement of tailings in Yellowknife Bay. There are several technical reports that provide water quality data with regards to arsenic concentrations in Back Bay. Selected data for years 1972 to 1994 are presented in Table B5.1-3.

TABLE B5.1-3
ARSENIC CONCENTRATIONS OF SELECTED SURFACE WATER SAMPLES COLLECTED IN BACK BAY (1972-1994)

Date Sampled	As (Total) (µg/L)	Data Source
June 14, 1972	20	Falk *et al*. 1973 – Stn GM-8
June 14, 1972	1	Falk *et al*. 1973 – Stn GM-7
June 14, 1972	12	Falk *et al*. 1973 – Stn GM-14
June 14, 1972	86	Falk *et al*. 1973 – Stn GM-15
June 14, 1972	120	Falk *et al*. 1973 – Stn GM-17
June 14, 1972	140	Falk *et al.* 1973 – Stn GM-18
June 14, 1972	120	Falk *et al*. 1973 – Stn GM-16

TABLE B5.1-3 (Cont'd)
ARSENIC CONCENTRATIONS OF SELECTED SURFACE WATER SAMPLES COLLECTED IN BACK BAY (1972-1994)

Date Sampled	As (Total) (µg/L)	Data Source
June 1, 1976	740	Moore *et al.* 1978-Stn 7
June 1, 1976	50	Moore *et al.* 1978-Stn 9
June 1, 1976	<20	Moore *et al*. 1978-Stn 10
June 1, 1976	<20	Moore *et al*. 1978-Stn 11
June 1, 1976	<20	Moore *et al*. 1978-Stn 17
June 9, 1976	60	Moore *et al*. 1978-Stn e
June 9, 1976	<20	Moore *et al*. 1978-Stn f

Date Sampled	As (Total) (µg/L)	Data Source
June 3, 1987	7 @ 1 m depth	HydroQual 1989-Stn 2027
June 3, 1987	6 @ 13 m depth	

Date Sampled	As (Total) (µg/L)	Data Source
June 25, 1988	7 @ 1 m depth	HydroQual 1989-Stn 2027
June 25 1988	8 @ 7 m depth	
June 25, 1988	7 @ 13 m depth	

Date Sampled	As (Total) (µg/L)	Data Source
September 1992	8.6	Jackson *et al*. 1996 (Station 3)
September 1992	4.5	Jackson *et al*. 1996 (Station 4)
September 1992	4.3	Jackson *et al*. 1996 (Station 5)
September 1992	4.4	Jackson *et al*. 1996 (Station 6)
September 1992	0.50	Jackson *et al*. 1996 (Station 13)
February/March 1993	1.4	Jackson *et al*. 1996 (Station 3)
February/March 1993	1	Jackson *et al*. 1996 (Station 4)
February/March 1993	0.6	Jackson *et al*. 1996 (Station 5)
February/March 1993	1.9	Jackson *et al*. 1996 (Station 6)
February/March 1993	0.4	Jackson *et al*. 1996 (Station 13)
June 1993	4.8	Jackson *et al*. 1996 (Station 3)
June 1993	6.6	Jackson *et al*. 1996 (Station 4)
June 1993	3.8	Jackson *et al*. 1996 (Station 5)
June 1993	3	Jackson *et al*. 1996 (Station 6)
June 1993	3.3	Jackson *et al*. 1996 (Station 13)
August 1993	5.9	Jackson *et al*. 1996 (Station 3)

TABLE B5.1-3 (Cont'd)
ARSENIC CONCENTRATIONS OF SELECTED SURFACE WATER SAMPLES COLLECTED IN BACK BAY (1972-1994)

Date Sampled	As (Total) (µg/L)	Data Source
August 1993	6.2	Jackson *et al*. 1996 (Station 4)
August 1993	5.9	Jackson *et al*. 1996 (Station 5)
August 1993	5.5	Jackson *et al*. 1996 (Station 6)
August 1993	6.9	Jackson *et al*. 1996 (Station 13)
February 1994	6.9	Jackson *et al*. 1996 (Station 3)
February 1994	0.7	Jackson *et al*. 1996 (Station 4)
February 1994	0.4	Jackson *et al*. 1996 (Station 5)
February 1994	1.4	Jackson *et al*. 1996 (Station 6)
February 1994	0.3	Jackson *et al*. 1996 (Station 13)
March 1994	0.4	Jackson *et al*. 1996 (Station 3)
March 1994	0.3	Jackson *et al*. 1996 (Station 4)
March 1994	0.4	Jackson *et al*. 1996 (Station 5)
March 1994	2.1	Jackson *et al*. 1996 (Station 6)
March 1994	0.4	Jackson *et al*. 1996 (Station 13)

Yellowknife Bay (North of Latham Island) Water Quality:

The section of Yellowknife Bay that extends northwards from Latham Island receives flows from the Yellowknife River and Back Bay. Selected water quality data for the Yellowknife River, which flows into the north end of Yellowknife Bay, are presented in Table B5.1-4. Selected water quality data for Yellowknife Bay (north of Latham Island) are presented in Table B5.1-5.

TABLE B5.1-4
ARSENIC CONCENTRATIONS OF SELECTED SURFACE WATER SAMPLES COLLECTED FROM THE YELLOWKNIFE RIVER AT/NEAR ITS CONFLUENCE WITH YELLOWKNIFE BAY (1993-1994)

Date Sampled	Arsenic Concentration (Total) (µg/L)	Data Source
March 1993	0.3	Jackson *et al*. 1996 (Station 1, Yellowknife River)
June 1993	0.3	
August 1993	0.3	
February 1994	0.3	Jackson *et al*. 1996 (Station 1, Yellowknife River), and Jackson 1998 (June-August)
March 1994	0.3	
June 21, 1994	0.3	
July 13, 1994	<0.3	
August 17, 1994	<0.3	

TABLE B5.1-5
ARSENIC CONCENTRATIONS OF SELECTED SURFACE WATER SAMPLES COLLECTED IN YELLOWKNIFE BAY, NORTH OF LATHAM ISLAND (1972-1994)

Date Sampled	Arsenic Concentration (Total) (µg/L)	Data Source
June 14, 1972	19	Falk *et al*. 1973-Stn. GM-11
June 14, 1972	4	Falk *et al.* 1973-Stn. GM-10
June 14, 1972	69	Falk *et al*. 1973-Stn. GM-12
June 14, 1972	83	Falk *et al*. 1973-Stn. GM-13
June 6, 1976	19	HydroQual 1989-Stn. 3011
June 5, 1987	5 @ 1 m depth	HydroQual 1989-Stn. 3011
June 18, 1988	0.6 @ 1 m depth	HydroQual 1989-Stn. 3011
June 18, 1988	7.4 @ 8 m depth	HydroQual 1989-Stn. 3011
September 1992	0.5	Jackson *et al*. 1998 (Station: Latham Island)
March 1993	0.4	
June 1993	3.3	
August 1993	6.9	
February 1994	0.3	
March 1994	0.4	
8 June 1994	1.0	
14 June 1994	5.2	
21 June 1994	4.1	
29 June 1994	3.8	
5 July 1994	4.1	
13 July 1994	4.6	Jackson *et al*. 1998 (Station: Latham Island)
20 July 1994	5.0	
27 July 1994	4.3	
8 August 1994	5.7	
17 August 1994	5.2	
6 September 1994	5.4	
23 September 1994	5.9	

Yellowknife Bay (South of Latham Island) Water Quality:

This section of Yellowknife Bay receives flows from the portion of the Bay located north of Latham Island, as well as from the Con Mine treated effluent, and drainage from the City of Yellowknife. The south end of Yellowknife Bay is open to the North Arm of Great Slave Lake. Selected water quality data for Yellowknife Bay between Latham Island and the North Arm are presented in Table B5.1-6.

TABLE B5.1-6
ARSENIC CONCENTRATIONS OF SELECTED SURFACE WATER SAMPLES COLLECTED FROM YELLOWKNIFE BAY – SOUTH OF LATHAM ISLAND (1972-1998)

Date Sampled	Total Arsenic Concentrations (µg/L)	Data Source
June 28, 1972	530	Falk *et a*l. 1973 Con 9 at Mosher Island
October 10, 1987	1 @ 1 m depth	HydroQual 1989-Stn. 6005
October 10, 1987	1 @ 10 m depth	
October 10, 1987	1 @ 20 m depth	
June 19, 1988	1 @ 1 m depth	HydroQual 1989-Stn. 6005
June 19, 1988	1 @ 10 m depth	
June 19, 1988	1 @ 20 m depth	
September 1992	4.4	Jackson *et al*. 1996 (Station 7)
September 1992	4.6	Jackson *et al*. 1996 (Station 8)
September 1992	3.4	Jackson *et al*. 1996 (Station 9)
September 1992	1.3	Jackson *et al.* 1996 (Station 11)
February/March 1993	0.8	Jackson *et al*. 1996 (Station 7)
February/March 1993	1.3	Jackson *et al.* 1996 (Station 8)
February/March 1993	0.8	Jackson *et al*. 1996 (Station 9)
February/March 1993	1.0	Jackson *et al.* 1996 (Station 11)
June 1993	2.2	Jackson *et al*. 1996 (Station 7)
June 1993	2.9	Jackson *et al*. 1996 (Station 8)
June 1993	1.4	Jackson *et al*. 1996 (Station 9)
June 1993	1.1	Jackson *et al*. 1996 (Station 11)
August 1993	3.7	Jackson *et al*. 1996 (Station 7)
August 1993	4.2	Jackson *et al*. 1996 (Station 8)
August 1993	0.6	Jackson *et al.* 1996 (Station 9)
August 1993	0.7	Jackson *et al.* 1996 (Station 11)

TABLE B5.1-6 (Cont'd)
ARSENIC CONCENTRATIONS OF SELECTED SURFACE WATER SAMPLES COLLECTED FROM YELLOWKNIFE BAY – SOUTH OF LATHAM ISLAND (1972-1998)

Date Sampled	Total Arsenic Concentrations (µg/L)	Data Source
February 1994	0.4	Jackson *et al*. 1998 (Dettah Dock Station)
March 1994	0.4	
8 June 1994	0.6	
14 June 1994	0.4	
29 June 1994	0.4	
13 July 1994	0.3	
20 July 1994	0.5	
27 July 1994	0.9	
8 August 1994	0.6	
17 August 1994	0.8	
6 September 1994	0.8	
Summer 1998	14.1	Ollson 1999 (Stn. 101)

The data reported by Ollson (1999) was not included in the calibration as it is not consistent with the other data points. The measurements seem to have been influenced by some local source and does not reflect the quality of the south Yellowknife Bay, which is being represented in the modelling effort.

B5.2 ARSENIC CONCENTRATIONS IN SEDIMENT

The concentrations of arsenic in the sediments at the outlet of Baker Creek and in Back Bay and Yellowknife Bay were investigated on several occasions as part of technical studies. While these studies had different objectives and incorporated a range of sampling and QA/QC procedures, they provide useful data for present purposes and for a technical understanding of arsenic concentrations spatial in sediments. The results of relevant studies are reviewed below.

Arsenic concentrations in sediment samples collected at the Baker Creek outlet to Back Bay are summarized in Table B5.2-1.

TABLE B5.2-1
MEAN ARSENIC CONCENTRATIONS OF SEDIMENT SAMPLES COLLECTED AT BAKER CREEK OUTLET

Date Collected	Mean Arsenic Concentration (µg/g)	Data Source
September 1992	2,550	Jackson *et al.* 1996 (Station 2)
June 1993	1337	Jackson *et al.* 1996 (Station 2)
August 1993	1595	Jackson *et al.* 1996 (Station 2)
Summer 1997	1839	Mace 1998 (Station 16)
Summer 1997	1946	Mace 1998 (Station 17)
Summer 1997	2838	Mace 1998 (Station 18)
Summer 1997	3757	Mace 1998 (Station 19)
Summer 1997	1736	Mace 1998 (Station 20)
Summer 1997	1825	Mace 1998 (Station 22)
Summer 1997	272	Mace 1998 (Station BC-30)
October 2001	1790	Dillon 2002a (Station BC-3)
October 2001	220	Dillon 2002a (Station BC-3)
October 2001	1270	Dillon 2002a (Station BC-3)
October 2001	1810	Dillon 2002a (Station BC-4)
October 2001	2270	Dillon 2002a (Station BC-4)
October 2001	1980	Dillon 2002a (Station BC-4)
June 2002	13.5*	Dillon 2002b (Station BC-3)
June 2002	40.0*	Dillon 2002b (Station BC-3)
June 2002	46.2*	Dillon 2002b (Station BC-3)
June 2002	27.5*	Dillon 2002b (Station BC-4)
June 2002	33.7*	Dillon 2002b (Station BC-4)
June 2002	46.4*	Dillon 2002b (Station BC-4)
November 2002	1710	Dillon 2002b (Site 3 and 4 downstream)
November 2002	1630	
November 2002	1660	
November 2002	961	
November 2002	881	
November 2002	124	
November 2002	69.7	
November 2002	782	
November 2002	1666.7	
November 2002	655.3	
March 2004	615	Dillon 2004 (Site 5, 6 and 7)
March 2004	686	
March 2004	2350	

* Geometric mean value of three replicate sample analyses.

Jackson *et al*. (1996) also report that the benthic fauna of Yellowknife Bay was investigated by Moore *et al*. (1979) who found that as a general trend, benthic diversity progressively increased with distance from the outlet of Baker Creek. Signs of recovery of the benthic populations occurred at a distance of 1000 to 1200 m from the outlet of Baker Creek.

Dillon (2002a) had trouble collecting the minimum 5 gram benthos sample size from Baker Creek, downstream of the Giant Mine site. A possible explanation for the low abundance of aquatic macro-organisms was the timing of the sampling event. Under early winter conditions (mid-October), aquatic species in Baker Creek would have been in an over-wintering state (Dillon 2002a). Additional sampling was completed in the summer of 2002 and adequate beathos were collected for concentration analysis, however there was no comment on the abundance of benthic species relative to upstream locations. No connection was made between benthic diversity or abundance and water and sediment quality in Baker Creek.

Back Bay and Yellowknife Bay Sediment Quality:

Mudroch *et al*. (1989) investigated the geochemistry of sediments in Back Bay and Yellowknife Bay as part of a comprehensive investigation of the effect of gold mine wastes on aquatic ecosystems. The specific objectives of their study were to identify the extent of contamination from gold mine waste in Yellowknife Bay and to assess the effectiveness of mine effluent treatment commenced in the early 1980's.

Sediment cores were collected in Back Bay and in Yellowknife Bay downstream of the City of Yellowknife. Radiometric data for Pb-210 and Cs-137 istope concentrations were used to date the sediment cores. It was concluded that the cores reflected the depositional history of arsenic in the area of Back Bay and Yellowknife Bay (Mudroch *et al*. 1989). The concentrations of arsenic in three sediment cores collected in Back Bay and Yellowknife Bay by Murdoch *et al*. (1989) are presented in Table B5.2-2.

TABLE B5.2-2
ARSENIC CONCENTRATIONS IN SEDIMENT CORES COLLECTED FROM BACK BAY AND YELLOWKNIFE BAY (MUDROCH *et al*. 1989)

Sediment Sample Location	Depth (cm)	Arsenic Concentrations (µg/g – dry weight)		
		Core 1	Core 2	Core 3
Back Bay	0 – 1	1,011	630	2,800
	1 – 2	720	790	1,160
	2 – 3	890	1,130	990
	3 – 4	1,230	960	670
	4 – 5	790	550	640
	5 – 6	595	145	176
	6 – 7	180	68	169
	7 – 8	172	54	182
	8 – 9	160	59	160
	9 – 10	140	65	145
	25 – 26	20	25	23
Yellowknife Bay	0 – 1	550	490	280
	1 – 2	890	590	543
	2 – 3	95	700	433
	3 – 4	53	108	72
	4 – 5	12	54	23
	5 – 6	15	26	26
	6 – 7	19	23	17
	7 – 8	21	19	11
	8 – 9	25	21	19
	9 – 10	22	18	36
	20 – 21	12	15	11
	27 – 28	16	18	15

Based on the data, Mudroch *et al*. (1989) consider that the background concentrations of arsenic in the sediments are in range of 15 to 25 µg As/g. The concentrations of arsenic (and zinc) in the sediments of Back Bay and Yellowknife Bay where investigated with respect to time as well as gold mining/primary/effluent treatment events. Significant decreases in arsenic concentrations in the sediment of both bays was observed to coincide with the implementation of an alkaline chlorination and arsenic oxidation effluent treatment process at the Giant Mine. It was additionally concluded that it would take about 20 years for clean sediments to cover the arsenic contaminated sediment with a 5 cm uncontaminated layer assuming the elimination of arsenic inputs into Yellowknife Bay.

In another study, Jackson *et al*. (1996) sampled sediments at thirteen locations including the outlet of Baker Creek and in Back Bay and Yellowknife Bay. The sampling was carried out in September 1992 and repeated in June and August 1993. Sampling locations are shown in Figure B5.2-1 and the range of sediment sample arsenic concentrations for the sampling stations are shown in Figure B5.2-2.

The results of the sediment sampling campaigns show that arsenic contamination of the sediment has occurred throughout Back Bay and upper Yellowknife Bay, with concentrations declining with distance from Baker Creek and the tailings placed in Yellowknife Bay.

As discussed there are studies such as Jackson *et al*. (1998) that investigated sediment quality in Back Bay and Yellowknife Bay. This information is useful in determining areas of elevated arsenic due to operations of the Giant mine. However, it must be noted that the modelling effort is focussed on the average sediment qulaity over the entire section of the bay, of which there are large areas that are not generally captured during the monitoring program. This is an important consideration in the data selected for calibration purposes.

An investigation was undertaken in 1997 (Mace 1998) to determine arsenic levels in sediment in Back Bay and Yellowknife Bay. Table B5.2-3 provides a summary of the data collected for Back Bay. To provide a sediment concentration appropriate for calibration purposes a very rough estimate of the fraction of the bay area that would be represented by the sediment level was conducted. An area-weighted average was then derived, as shown in Table B5.2-3.

TABLE B5.2-3
ARSENIC CONCENTRATIONS IN BACK BAY SEDIMENT BASED ON MACE (1998)

Station Number	Average Concentration (µg/g)	Estimated Fraction of Bay Area
4	3140	0.17
6	200	0.33
8	259	0.33
21	1193	0.17
Weighted Average	875	

HydroQual (1989) undertook a comprehensive program which included the sampling of sediments in Back Bay and Yellowknife Bay in order to characterize the spatial distribution of trace metals in surface (e.g. top 5 cm) sediments. Sampling sites were selected to allow comparisons with historic data. The spatial distribution of arsenic concentrations in the surface sediments as determined by HydroQual (1989) is shown in Figure B5.2-3. This data, summarized in Table B5.2-4 is used in the calibration of the model for application to this project.

TABLE B5.2-4
ARSENIC CONCENTRATIONS IN BACK BAY AND YELLOWKNIFE BAY SEDIMENT BASED ON HYDROQUAL (1989)

	Number of Samples	Arsenic Concentration (μg/g)		
		Average	Minimum	Maximum
Back Bay	32	707	28	3000
North Yellowknife Bay (surficial samples)	24	197	13	1000
South Yellowknife Bay (surficial samples)	11	32	10	90

It should be noted that data from HydroQual (1989) is consistent with the data collected by other investigations. For example, Mudroch *et al*. (1989) collected samples in Yellowknife Bay (Table B5.2-2) that had an average concentration of 308 μg/g in the upper 5 cm. This was supported by Jackson *et al.* (1996) which collected data in a similar location and reported concentrations in the range of 171 to 421 μg/g and Mace (1998) which measured arsenic levels of 173 and 302 μg/g. In this same area, HydroQual (1989) reports sediment levels of 180 to 230 μg/g. Therefore, it was judged that HydroQual captured the areas of concern as well as providing a comprehensive survey of Yellowknife Bay which could be used for the purposes of model calibration.

In 2004, Golder carried out a sampling program in Back Bay and North Yellowknife Bay. The results of this sampling is presented in Table B5.2-5.

TABLE B5.2-5
MEAN ARSENIC CONCENTRATIONS OF SEDIMENT SAMPLES COLLECTED FROM BACK BAY AND YELLOWKNIFE BAY

Sediment Sample Location	Date Collected	Mean Arsenic Concentration (µg/g)	Data Source
Back Bay	2004	171	Golder 2005
		1080	
		718	
		870	
		874	
		305	
		74.7	
		59.2	
		67	
		73.1	
		894	
		2250	
		1780	
		954	
		363	
		498	
North Yellowknife Bay	2004	189	Golder 2005
		378	
		39.9	
		59.5	
		28	
		37	
		401	
		317	
		106	
		33.8	
		36	
		45.9	
		47.7	
		353	
		14.6	
		136	
		13.1	
		12.5	
		37.5	
		611	
		470	
		765	

TABLE B5.2-5 (Cont'd)
MEAN ARSENIC CONCENTRATIONS OF SEDIMENT SAMPLES COLLECTED FROM BACK BAY AND YELLOWKNIFE BAY

Sediment Sample Location	Date Collected	Mean Arsenic Concentration (µg/g)	Data Source
North Yellowknife Bay	2004	744	Golder 2005
		154	
		56.5	
		20.2	
		219	
		6.9	
		915	
		822	
		76.9	
		1160	
		1870	
		487	
		64.4	
		58.4	
		55.7	

B5.3 MEASURED ARSENIC LEVELS IN POREWATER

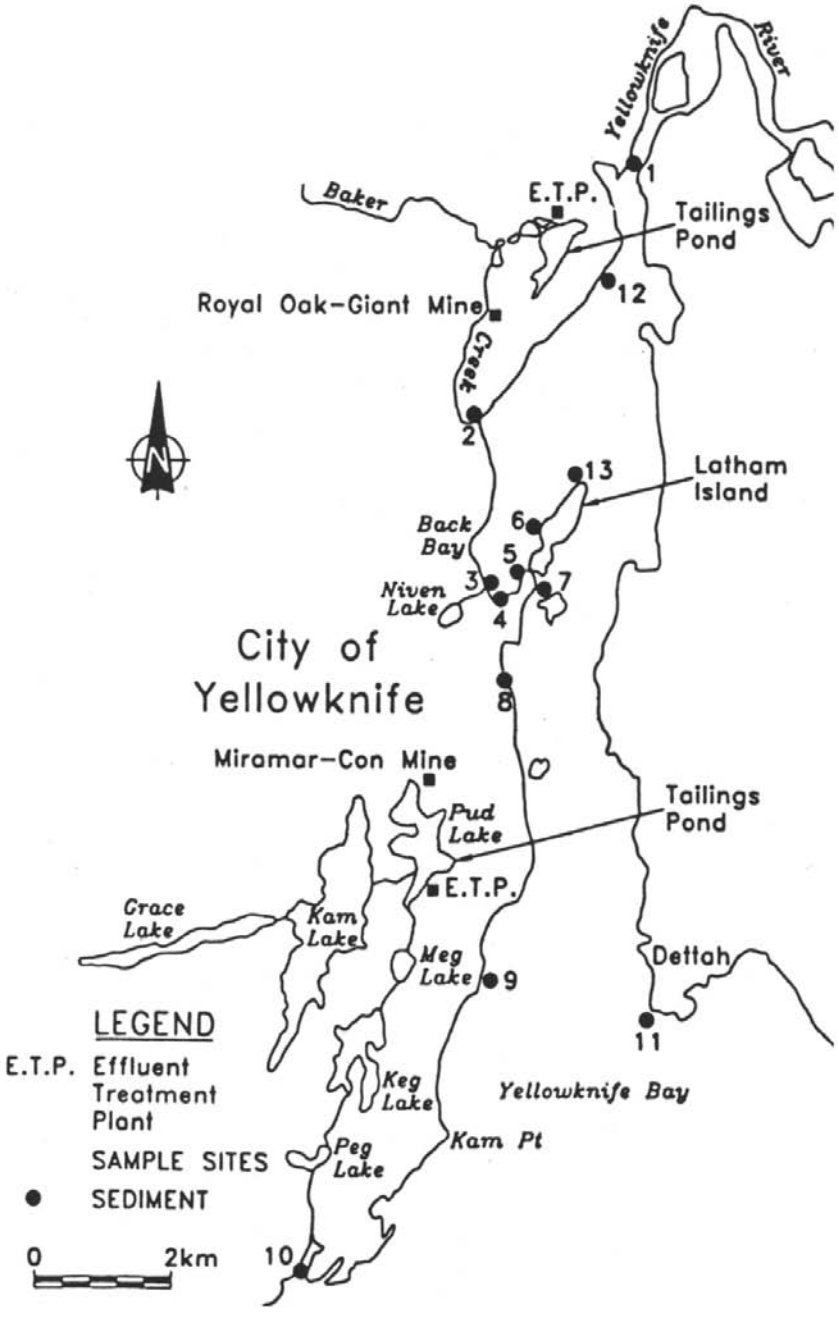
Porewater concentrations of arsenic are available for both tailings and sediments and the values from literature are summarized in Table B5.3-1. Samples were taken from 1998 to 1999 for the sediment porewater and 2000 for tailings porewater.

TABLE B5.3-1
MEASURED ARSENIC LEVELS IN POREWATER

Sediment (ppb)							
Location	**# of Samples**	**Min**	**Max**	**Geo Mean**	**Average**	**Avg Std Dev**	**Reference**
Baker Creek Marsh	3	1384	1760	1537	1545	143	RMCC (1998)
Baker Creek Pond	3	504	1336	798	865	314	RMCC (1998)
Yellowknife Bay, near Baker Creek	15	22	3371	266	721	777	RMCC (1998)
Yellowknife Bay (17.5 cm)	3	111	194	143	147	31	RMCC (1999)
Con Mine Lakes	12	228	2700	839	1096	652	RMCC (1999)
Tailings (ppb)							
Location	**# of Samples**	**Min**	**Max**	**Geo Mean**	**Average**	**Avg Std Dev**	**Reference**
Back Bay	30	53	1430	287	370	204	EBA (2001)

FIGURE B5.2-1

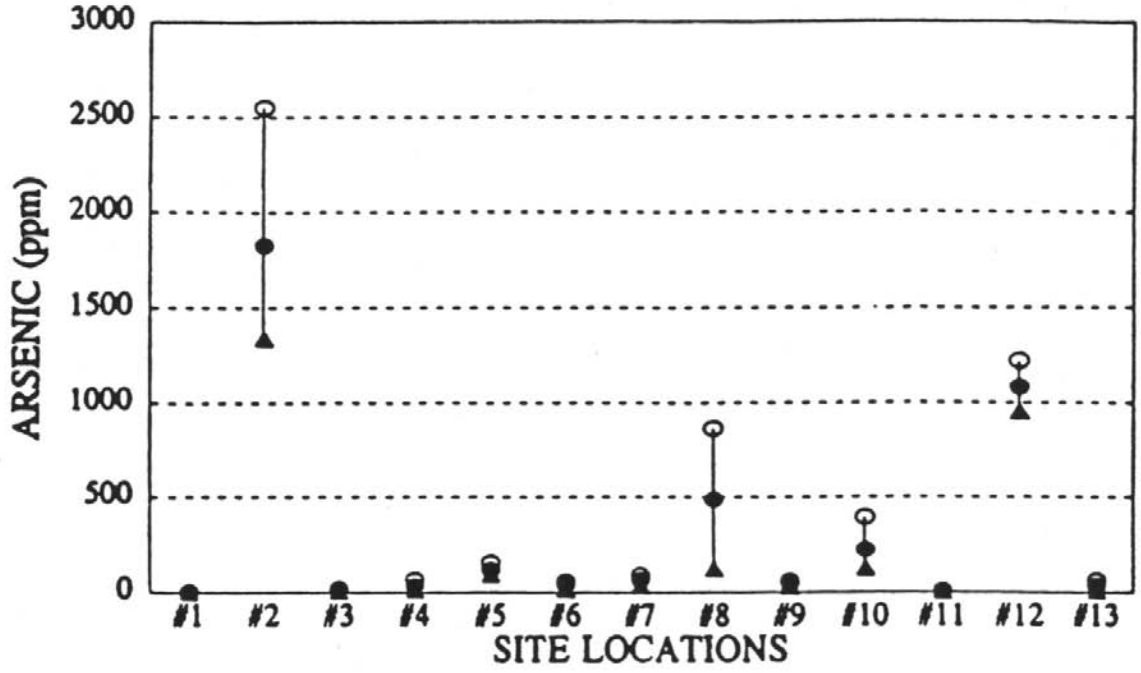
SEDIMENT SAMPLING SITES IN THE YELLOWKNIFE - BACK BAY AREA



SOURCE: FROM JACKSON et al (1996)

FIGURE B5.2-2

YELLOWKNIFE - BACK BAY (Sediment) ARSENIC VALUES, 1992-93



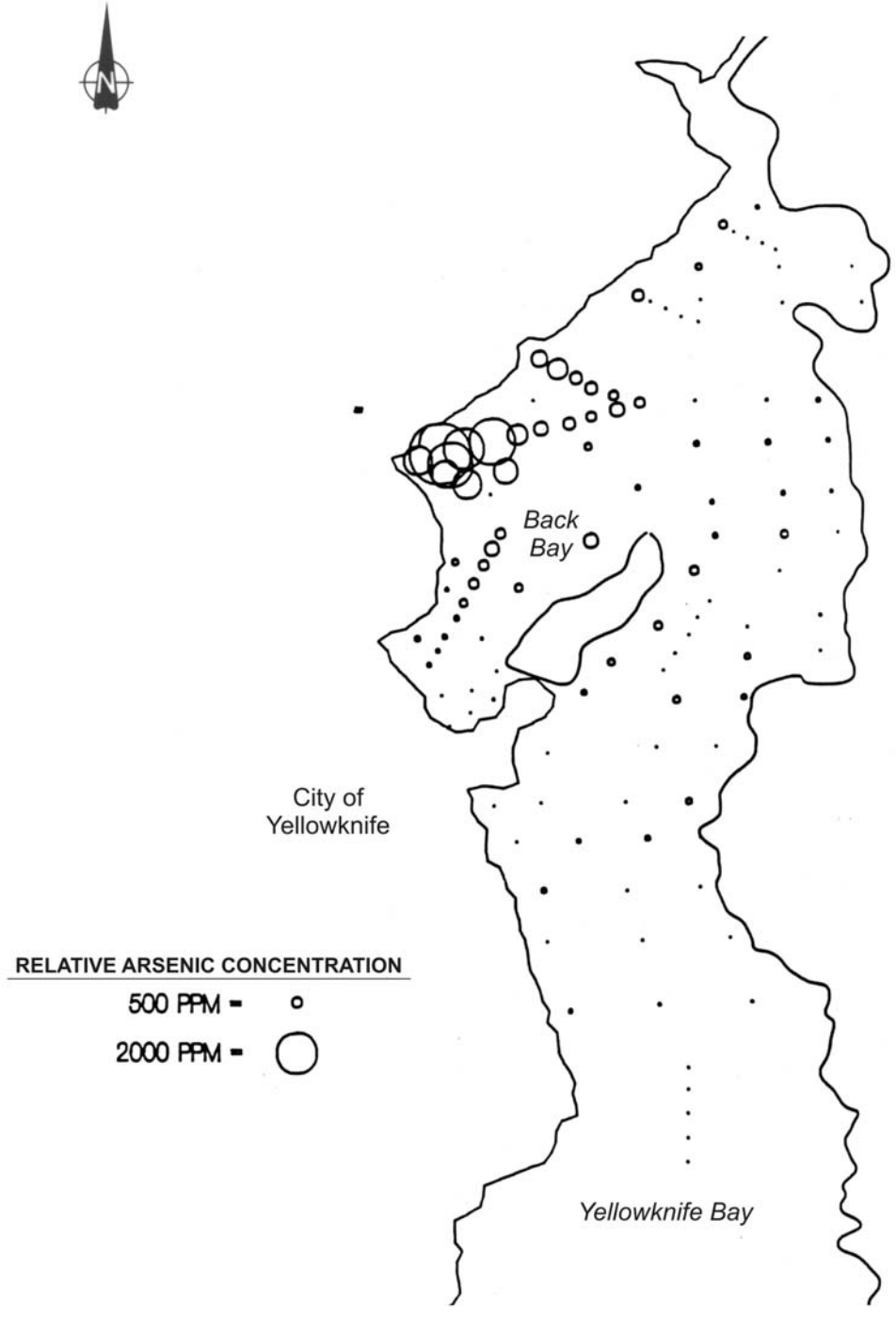
LEGEND:

- ○ MAXIMUM VALUES
- ▲ MINIMUM VALUES
- ● MEAN VALUES

SOURCE: FROM JACKSON et al (1996)

FIGURE B5.2-3

SEDIMENT ARSENIC CONCENTRATIONS



SOURCE: ADAPTED FROM HYDROQUAL (1989)

B6.0 LAKE MODEL CALIBRATION

Calibration of the lake dispersion model (LAKEVIEW) was carried out using historical arsenic loading data for Baker Creek, monitoring data from the study by HydroQual and Gore & Storrie (1989), publications by Murdoch *et al*. (1989), Mace (1998), Jackson (1998) and other sources. Year 1950 was taken as the initial datum point. The calibration comprised: (i) quantifying sediment porewater, surface water, and sediment solids interactions, (ii) arsenic loading, and (ii) convective / dispersive transport of dissolved arsenic in surface water.

B6.1 SEDIMENT POREWATER AND SOLIDS INTERACTIONS

Mace (1998) studied the speciation of arsenic in sediments in the Back Bay area. Some 16 sediment solids were analysed for solid phase arsenic distribution. Using the original data, the speciation of arsenic is summarised in Table B6.1-1.

TABLE B6.1-1
ARSENIC SPECIATION IN BACK BAY SEDIMENTS

Arsenic adsorbed to non-specific surfaces (clay and loam)	1.3 %
Arsenic in carbonaceous sediment	3.1 %
Arsenic in organic sediment	2.3 %
Arsenic in iron and manganese oxides	35.2 %
Residual arsenic	58.4 %

Presumably, the residual arsenic fraction includes arsenopyrite and other reduced arsenic minerals essentially insoluble, hence, not available for exchange unless the arsenic moiety is oxidized. The largest potentially exchangeable fraction is arsenic associated with iron and manganese oxides. Depending on the sample, this fraction ranged from 14.7 % to 76.9 % of the total arsenic in Back Bay sediments. It follows that the concentration of the dissolved arsenic in the porewater is dependent on this fraction. The mobilization of iron oxihydroxide - bound arsenic is discussed in Section B3.2. Mace (1998) further analysed 6 samples to assess porewater / sediment speciation. The results are summarized in Table B6.1-2.

TABLE B6.1-2
ARSENIC SPECIATION IN BACK BAY POREWATER AND SEDIMENT

Porewater			Sediment Solids				
As (µg/L)	As (mol/L)	Fe (mg/L)	As (mg/kg)	As (mol/kg)	Fe (mg/kg)	Fe (mol/kg)	As/Fe ratio
1336	$2.25x10^{-5}$	0.84	2332	0.311	16490	0.297	0.1047
1384	$1.85 x10^{-5}$	25.63	2197	0.293	35448	0.640	0.0458
1642	$2.19 x10^{-5}$	0.7	3726	0.0497	20717	0.374	0.1316
794	$1.06 x10^{-5}$	2.8	1428	0.0191	41982	0.757	0.0343
57	$0.08 x10^{-5}$	1.0	525	0.0070	33018	0.595	0.0117
188	$0.20 x10^{-5}$	7.2	509	0.0068	40490	0.730	0.0093

Table B6.1-2 also summarizes the necessary data for modeling arsenic in the sediment as outlined in Section B3.2. The prediction of porewater concentrations with the solid phase (sediment) data shown in Table B6.1-2 was employed as a test for model efficacy. The results of the simulations are compared with observations in Table B6.1-3.

TABLE B6.1-3
COMPARISON OF OBSERVED AND SIMULATED DISSOLVED POREWATER CONCENTRATIONS

Test #	Observed Porewater Arsenic (µg /L)	Predicted Porewater Arsenic (µg /L)
1	1336	1,686
2	1384	742
3	1642	2110
4	794	551
5	57	187
6	188	149

Model predictions were based on 35 % of arsenic bound to ferric hydroxide (see Table B6.1-1). The database covers an approximately 10-fold variation in arsenic concentrations. For this reason the difference between the natural logarithms of the predicted and observed values were used as the test for the validity of the solid-solution based model. Applying the Chi-Square test for model validity, the model was found to be appropriate for the present application.

B6.2 ARSENIC LOADING

The primary sources of arsenic input to Back Bay from the Giant Mine site have been and continue to be:

- The effluent discharge from the effluent treatment plant (ETP) at the mine. Effluent from this plant is released to Baker Creek over the summer months only. Since the plant was last upgraded in 1993, the arsenic level in the treated effluent has averaged approximately 0.3 mg/L (300 µg/L) arsenic. The discharge flow over the 1994 to 1999 period has been estimated to average about 1.7 x 10^6 m^3/yr, which results in an average arsenic loading of approximately 500 kg/yr. Much higher loadings in Baker Creek have been estimated for the earlier period (i.e. 1,500 kg/y between 1981 and 1993; 8,000 kg/y between 1968 and 1980; and 12,500 kg/y prior to 1968). These estimates do not include the arsenic loading resulting from atmospheric emissions from the roaster operation nor the background arsenic load carried with runoff in the Baker Creek watershed.

- The loading carried with runoff from contaminated soils in the Baker Creek watershed. The mean annual flow in Baker Creek over the 1972 to 1998 period measured 0.18 m^3/s (5.7 x 10^6 m^3/y) excluding the ETP discharge. Over the 1988 to 1997 period, the mean arsenic level measured in Baker Creek when there was no effluent discharge equalled 0.08 mg/L (80 µg/L). The equivalent annual arsenic loading to Back Bay therefore, is estimated to equal 450 kg/y.

The combined arsenic load from these sources was estimated to equal about 12,500 kg/y prior to 1968, 8,000 kg/y from 1968 to 1980, 1,300 kg/y between 1981 and 1993 inclusive and 950 kg/y from 1994 to 2000. During the model calibration procedure, however, it became evident that these initial load estimates were insufficient for predicting water quality in either Back Bay or Yellowknife Bay. Evidently, unaccounted arsenic in mine water seepage and airborne arsenic sources contributed significantly to the total arsenic load in the early years of operation. Stack emissions of arsenic prior to 1950 have been estimated to be in excess of 7,000 kg/day (CPHA 1977) and from 1951 to 1970 to average in excess of 2,500 kg/day (Environment Canada, 1996). Thus, load estimates from 1950 to 2000 were treated as unknown distributed parameters. The load for each time period was assigned a triangular distribution initially and revised load estimates were obtained by the Metropolis-Hastings procedure described in Section B4.0. Measured dissolved arsenic concentrations at several locations in Back Bay and Yellowknife Bay were employed as target values. The computer predicted, revised load estimates were 25,000 kg/y prior to 1968, 14,000 kg/y between 1968 and 1980, and 1,700 kg/y from 1981 to 1993. The previous estimate of 950 kg/y from 1994 to the present remained unchanged. In addition to dissolved arsenic loading, direct loading to the sediment (tailings deposition) was required to model the current sediment arsenic levels properly. During the early days of mining activity, the top 3 cm of sediments in Back Bay and the two segments of Yellowknife Bay were estimated to receive 0.630 kg/m^2 and 0.233 kg/m^2 particulate arsenic, respectively. These revised loadings were used in all subsequent calibration procedure.

B6.3 TRANSPORT PARAMETERS

This calibration entailed the estimation of the horizontal dispersion and sediment / surface water exchange. The calibrated parameters included: i) correction factors for nominal values of horizontal dispersion coefficients between Back Bay and Northern Yellowknife Bay segments, Northern Yellowknife Bay and Southern Yellowknife Bay segments, and Southern Yellowknife Bay and Great Slave Lake, ii) the distribution of the outflow from Northern Yellowknife Bay (essentially the Yellowknife River flow) between Back Bay and. Southern Yellowknife Bay and iii) arsenic retardation in Back Bay and Yellowknife Bay sediments. Since the 1989 study (HydroQual and Gore & Storrie, 1989) provided the most comprehensive database on both water and sediment quality in Back Bay and Yellowknife Bay, year 38 (1988) was used as the primary model calibration point. Each parameter was assigned a symmetric triangular distribution defined by the minimum value, the mean (nominal) value and the maximum value. The Metropolis-Hastings algorithm as described in Section B4.0 was employed for parameter estimation. Altogether 5 parameters were estimated simultaneously through approximately 10,000 iterations covering 1950 to 1999. Convergence was achieved for each parameter. The posterior distribution of the parameter estimates indicated insignificant co-variance between the parameters. The initial distributions and the calibrated values of the 5 transport parameters are summarized in Table B6.3-1.

TABLE B6.3-1
CALIBRATED MASS TRANSFER COEFFICIENTS

Parameter	Initial Assigned Distribution			Calibrated value
	Minimum	Mean	Maximum	
$\overline{E}_{3,5}$ (m^2s^{-1})	0.0	0.60	1.2	1.08
$\overline{E}_{5,7}$ (m^2s^{-1})	0.0	6.4	12.8	0.10
$\overline{E}_{7,9}$ (m^2s^{-1})	0.0	40.0	80.0	78.8
Flow Fraction	0.0	0.2	0.4	0.301
Retardation	0.0	600	1200	118.2

Note:

$\overline{E}_{3,5}$ = horizontal dispersion coefficient between Back Bay and Northern Yellowknife Bay.

$\overline{E}_{5,7}$ = horizontal dispersion coefficient between Northern and Southern Yellowknife Bay.

$\overline{E}_{7,9}$ = dispersion coefficient between Southern Yellowknife Bay and Great Slave Lake.

Flow Fraction = fraction of flow into Back Bay from Northern Yellowknife Bay.

Retardation = Retardation coefficient for sediments.

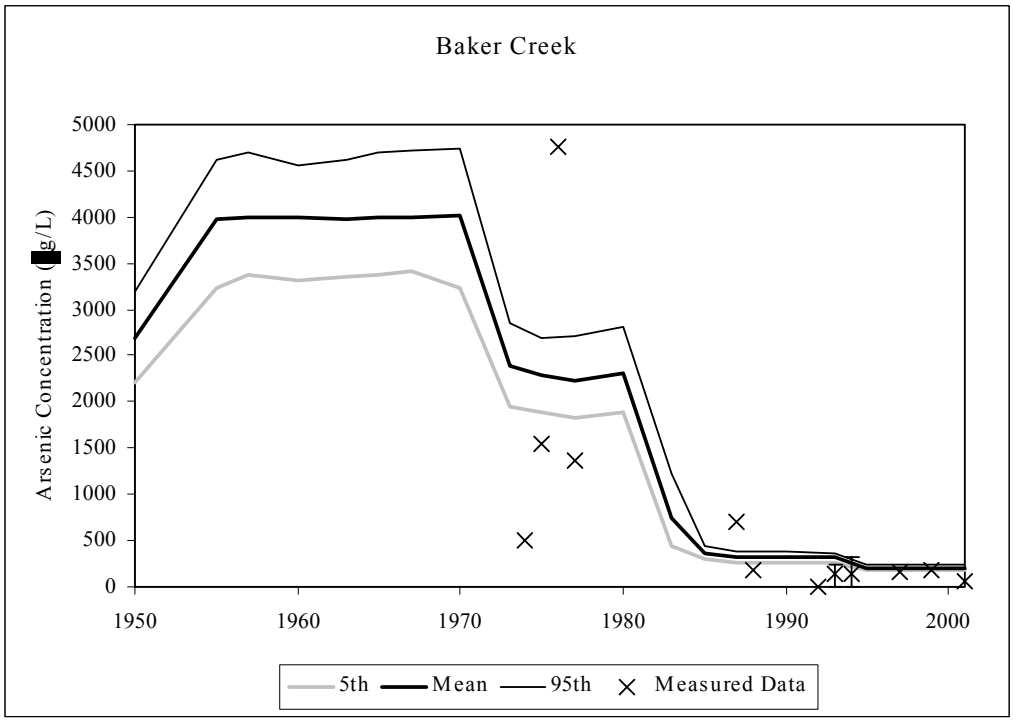
Mean assigned values given in Table B6.3-1 were derived using the nominal parameter values given in Table B2.3-1 for the distance between the midpoints of adjacent segments. According to the calibration results, the dominant transport mode into and out of Back Bay and between Southern Yellowknife Bay and Great Slave Lake is horizontal dispersion. Between the two Yellowknife Bay segments, however, dispersion was not significant and transport was completely flow (Yellowknife River) dominated. Apparently, Back Bay was affected by the periodic inflow of water from the Yellowknife River into this region. On an annual average basis, this inflow amounted to approximately 30% of the Yellowknife River flow. The calibrated retardation coefficient is related to the sediment sorption coefficient ($k_D^{'}$) as shown below:

$$Retardation = \varepsilon + (1-\varepsilon)k_D^{'}\rho \qquad \text{(B6-1)}$$

For nomenclature and further explanation concerning the application of the retardation coefficient, equation (B2.6) should be consulted. A retardation factor of 118 corresponds to a sediment sorption capacity of 0.5 m^3 kg^{-1}. This value is well within the range expected for arsenic (Sheppard and Thibault 1990). The variability of $k_D^{'}$ between sediments in various segments was small. Consequently, a constant sediment $k_D^{'}$ value was used throughout the region. It was concluded that all calibrated values were acceptable and should be used in further assessment.

The results of the simulations using all calibrated values are shown in on Figures B6.3-1 and B6.3-2 for water and sediment respectively. The predicted water and sediment concentrations are seen to agree reasonably well with the measured levels in all three segments of Yellowknife Bay as well as in Baker Creek. Generally, the arsenic concentrations in the surface water responded rapidly (within 2 years) to changes to arsenic loading. The response of sediment solid phase concentrations to loading was much slower. Current sediment arsenic levels (year 2000) were predicted to still be influenced by high arsenic loadings prior to 1980.

FIGURE B6.3-1
CALIBRATION RESULTS FOR WATER QUALITY



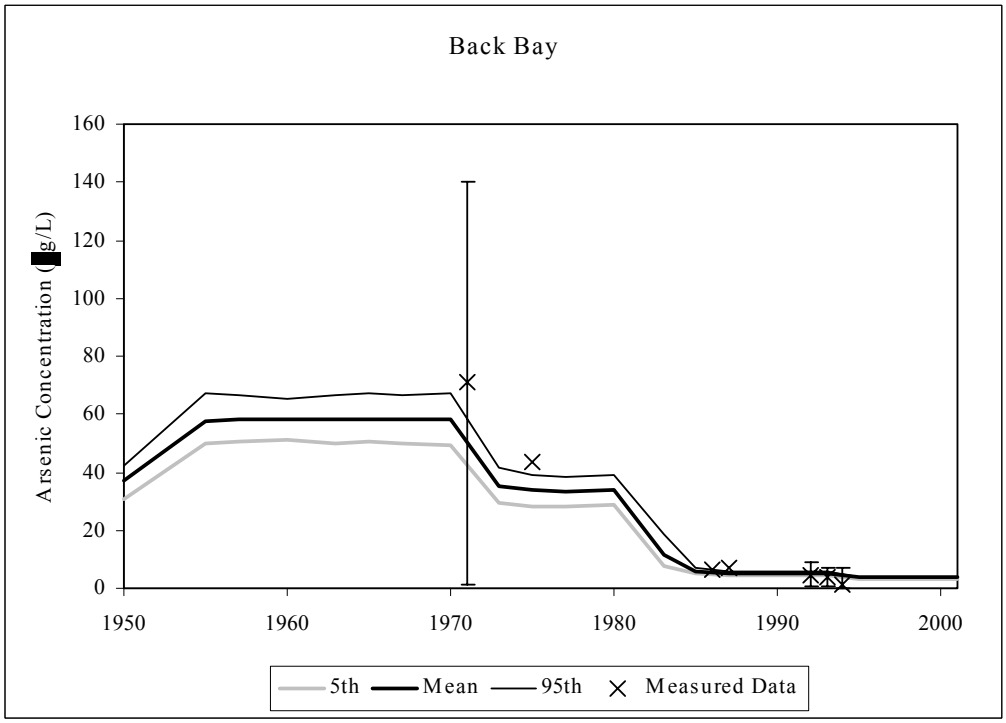
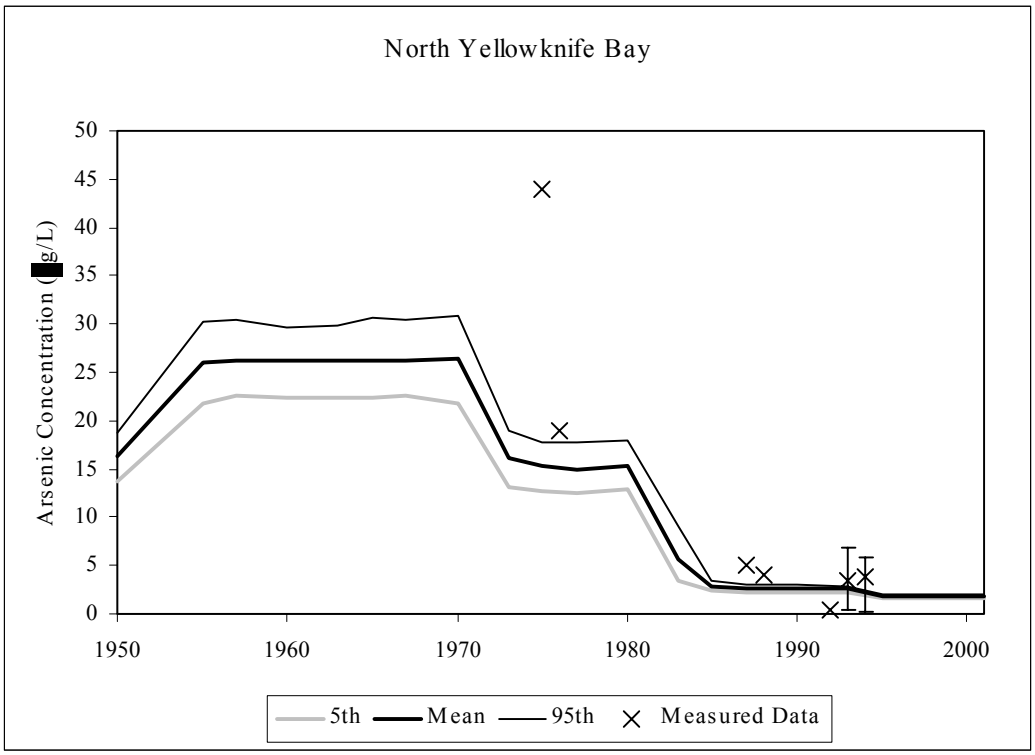


FIGURE B6.3-1 (Cont'd)
CALIBRATION RESULTS FOR WATER QUALITY



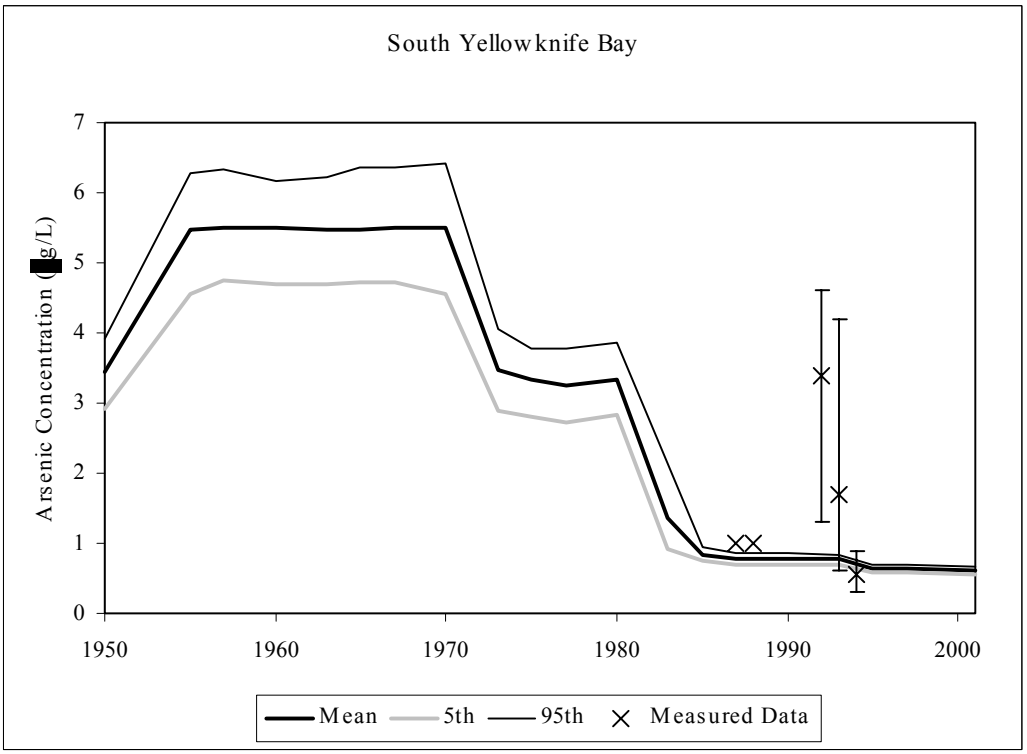
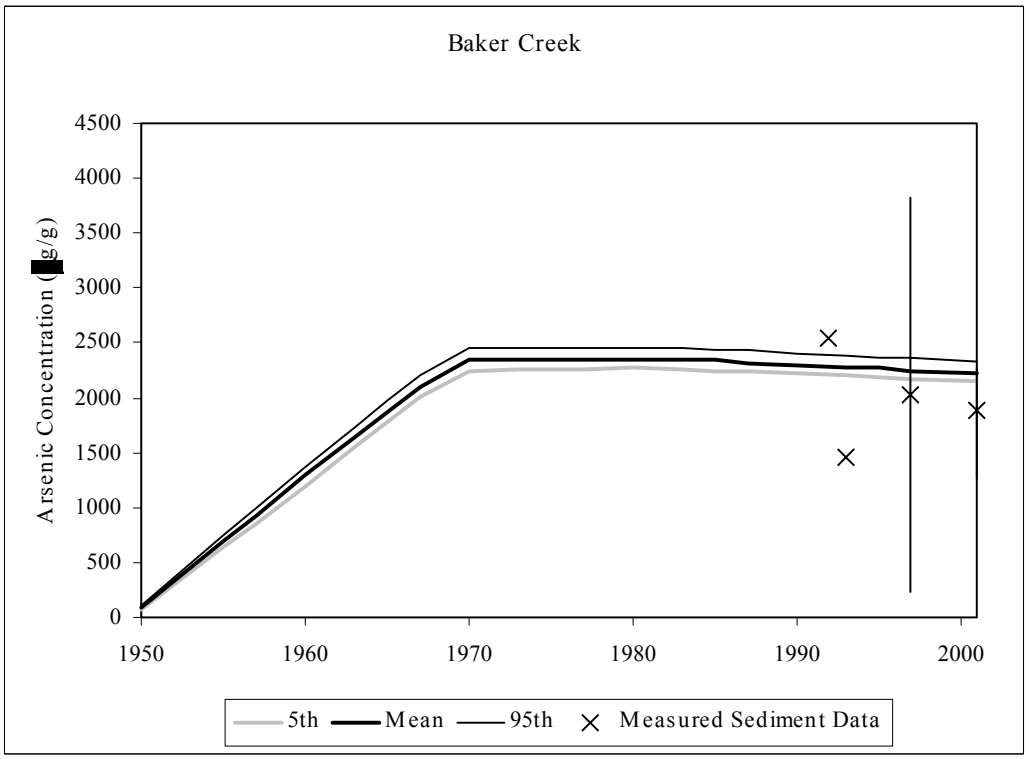


FIGURE B6.3-2
CALIBRATION RESULTS FOR SEDIMENT QUALITY



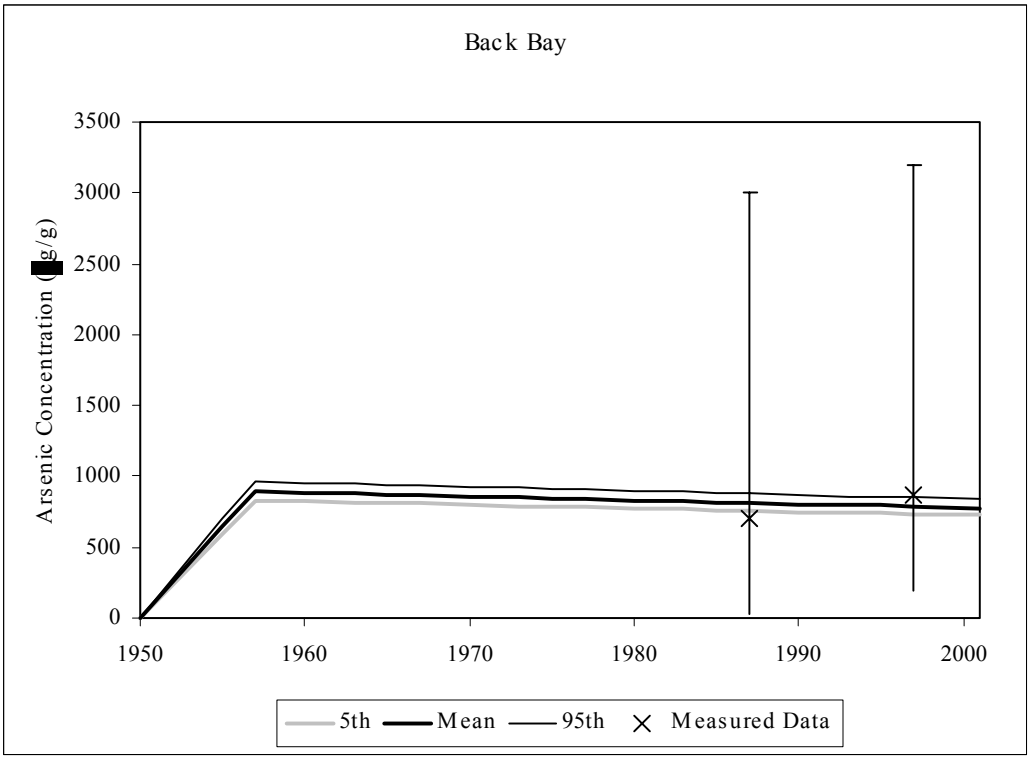
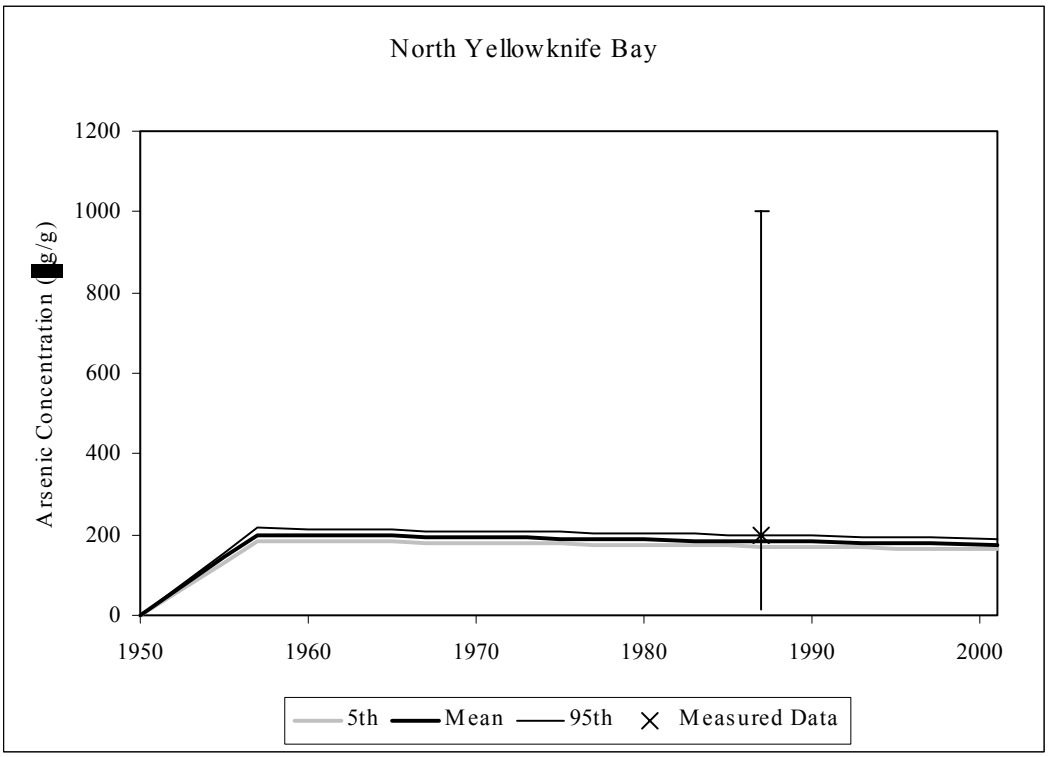
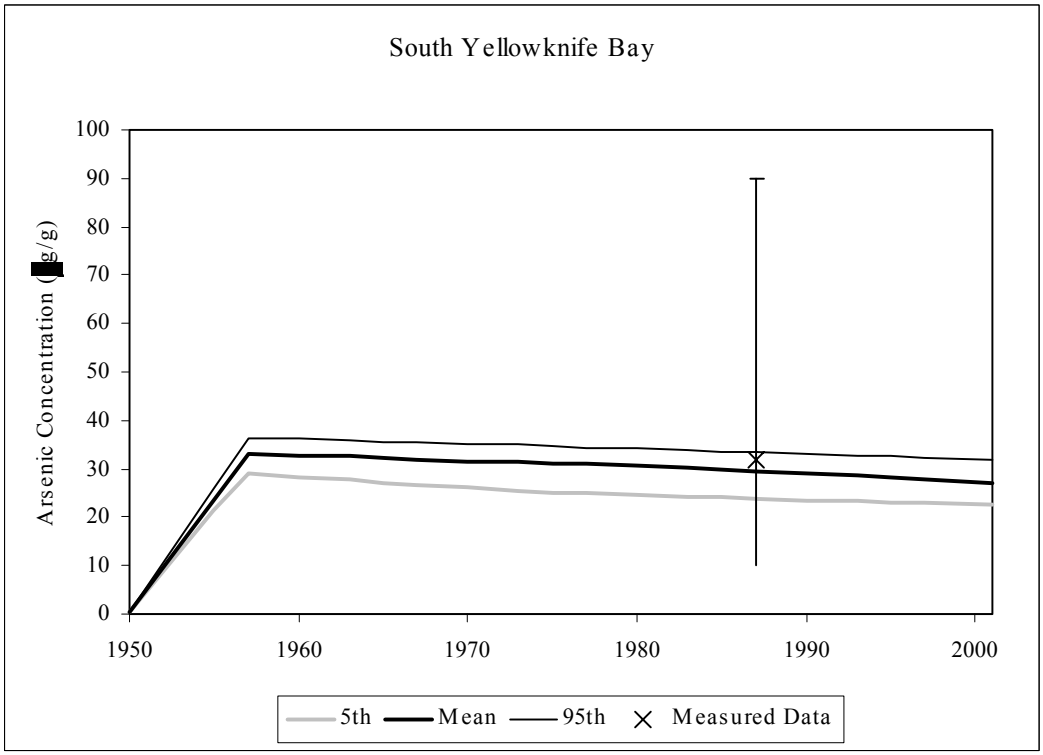


FIGURE B6.3-2 (Cont'd)
CALIBRATION RESULTS FOR SEDIMENT QUALITY





B7.0 REFERENCES

Appelo, C.A.J. and D. Postma 1993. Geochemistry, Groundwater, and Pollution. A.A. Balkema Publishing Co., Rotterdam.

Canadian Council of Ministers of the Environment (CCME) 2002. *Canadian Environmental Quality Guidelines*.

Canadian Public Health Association (CPHA) 1977. *Final Report of Task Force on Arsenic: Yellowknife, Northwest Territories.*

Chapra S.C and K.H. Rechow 1983. *Engineering Approaches for Lake Management.* Mechanistic Modeling. Vol 2, Butterworth Publishers, Boston, MA, U.S.A.

Csanady, G.T. 1973. *Turbulent Diffusion in the Environment.* D. Reidel Publishers. Boston, MA.

Cussler, E.L. 1984. *Diffusion - Mass Transfer in Fluid Systems.* Cambridge University Press, New York, NY.

Davé, N.K., T.P. Lim, D. Horne, Y. Boucher, and R. Stuparyck 1997. *Water Cover on Reactive Tailings and Wasterock: Laboratory Studies of Oxidation and Metal Release Characteristics.* Proceedings of the VI International Conference on Acid Rock Drainage, Vancouver, BC, Vol. 2:779-795.

Dillon Consulting Limited 2004. *Biological Sampling at Baker Creek 2003 – Technical Report Final.* Department of Indian and Northern Affairs Development. November 2002. 02-0465.

Dillon Consulting Limited 2002a. *Biological Sampling at Baker Creek – Summary Report.* Department of Indian and Northern Affairs Development. April 2002. 01-9670-3000.

Dillon Consulting Limited 2002b. *Biological Sampling at Baker Creek – Technical Report Final.* Department of Indian and Northern Affairs Development. November 2002. 02-0465.

Dillon Consulting Limited 2002c. *Biological Sampling at Baker Creek 2002 – Technical Report Final.* Department of Indian and Northern Affairs Development. November 2002. 02-0465.

Duever, T.A. and P.M. Reilly 1990. *Estimation of Parameters in Monte Carlo Modelling.* Canadian Journal of Chemical Engineering, Vol. 68:1040-1051.

Dulien, F.A.L. 1979. *Porous Media: Fluid Transport and Pore Structure.* Academic Press, New York, NY.

EBA Engineering Consultants Ltd. 2001. *Assessment of Back Bay Tailings Deposit, Giant Mine Yellowknife, NT.* For Deton'Cho Environmental Alliance, Yellowknife, NT. Project No. 0701-99-14263.0008.

Environment Canada (1996). *National Pollutant Release Inventory.* Northwest Territories.

Falk, M.R., M.D. Miller, and S.J.M. Kostiuk 1973. *Data Supplement to: Biological Effects of Mining Wastes in the Northwest Territories.* Data Report Series No. CEN/D-73-1. Department of the Environment. Fisheries and Marine Service. Fisheries Operations Directorate. Central Region.

Faust, S.D and O.M. Aly 1981. *Chemistry of Natural Waters*. Ann Arbor Science. 320-335.

Gamsjäger, H. 1989. *Thermodynamic Aspects of Dissolution Reactions in the System* Mg^{2+} - Ca^{2+} - CO_2 - H_2O. Monograph series on Mineral Deposits 28: 269-285: Gebrüder Borntraeger, Berlin-Stuttgart.

Gilks, W.R., S. Richardson, and Spiegelhalter 1996. *Markov Chain Monte Carlo in Practice.* Chapman and Hall (CRC) New York.

Golder Associates 2005. *Investigation of the Aerial Extent of Historic Tailings in Back Bay / Yellowknife Bay. Prepared for Giant Mine Remediation Project.* March.

Gresens, R.L. 1981, *The Aqueous Solubility Product of Solid Solutions,* Chem.Geol., 32, 73-86.

Harrington, J.M., S.E. Fendorf, and R.F. Rozenzweig 1998. *Biotic Regeneration of Arsenic (III) in Metalloid Contaminated Lake Sediments*. Environ. Sc. Technol. 32, 2525-2530.

Harris, G.B. and E. Krause 1994. *Disposal of Arsenic from Metallurgical Processes.* 122 TMS Annual Meeting and Exhibition, Denver, Co. September 21-25.

Health Canada 2004. *Arsenic in Drinking Water. Document for Public Comment.* Prepared by the Federal-Provincial-Territorial Committee on Drinking Water. Comment period ends May 17, 2005. November.

HydroQual Consultants Inc. and Gore & Storrie Ltd. 1989. *Water Quality Study of Yellowknife Bay: 1987-1988*. Prepared for Water Resources Division, Department of Indian and Northern Affairs.

Jackson, F.J. 1998. *Yellowknife – Back Bay Summer Water Quality Monitoring Program (September 1992 to June 1995).* Water Resources Division. Indian and Northern Affairs Canada. April.

Jackson, F.J., C.N. Lafontaine and J. Klaverkamp 1996. *Yellowknife – Back Bay Study on Metal and Trace Element Contamination of Water, Sediment and Fish*. Department of Indian and Northern Affairs Canada. November.

Lorax Environmental 1999. *Assessment of Metal Speciation in Baker Creek, Yellowknife.* Report to Indian and Northern Affairs Canada. Water Resources Division.

Mace, I.S. 1998. *A Study of Arsenic Contamination from the Royal Oak Giant Mine, Yellowknife, Northwest Territories*, M.Eng. Thesis. May.

Min, C-K. 1998. *A Gibbs Sampling Approach to Estimation and Prediction of Time-Varying Parameter Models.* Computational Statistics and Data Analysis, 27: 171-194.

Moore, J.W., S.J. Wheeler and D.J. Sutherland 1978. *The Effects of Metal Mines on Aquatic Ecosystems in the Northwest Territories II.* Giant Yellowknife Mines Limited. Report EPS 5-NW-78-9. June.

Murdoch, A., S.R. Joshi, D. Sutherland, P. Murdoch and K.M. Dickson 1989. *Geochemistry of Sediments in the Back Bay and Yellowknife Bay of the Great Slave Lake.* Enviro. Geol. Sci. Vol. 14, No. 1.

Ollson, C.A. 1999. *Arsenic Contamination of the Terrestrial and Freshwater Environment Impacted by Gold Mining Operations*. Yellowknife, NWT. M.Sc. Thesis. Royal Military College of Canada. November.

Paige, C.R., W.J. Snodgrass, R.V. Nicholson, J.M. Scharer, and Q.H. He 1997. *The Effect of Phosphate on the Transformation of Ferrihydrate into Crystalline Products in Alkaline Media*. Water Air and Soil Pollution, 98: 397-412.

Paige, C.R., W.J. Snodgrass, R.V. Nicholson, and J.M. Scharer 1996a. *The Release of Arsenic from Model Waste-water Treatment Solids: A Mechanism Based on Surface Ligand Exchanges*. Water Poll. Res. J. Can, 29: 507-530.

Paige CR., W.J. Snodgrass, R.V. Nicholson, and J.M. Scharer 1996b. *The Crystallization of Arsenate-contaminated Iron Hydroxide Solids at High pH.* Water Environ. Res,. 68: 981-988.

Royal Military College of Canada 1999. *Arsenic Contamination of the Terrestrial and Freshwater Environment Impacted by Gold Mining Operations Yellowknife, NWT*. November. M.Sc. Thesis, Christopher Andrew Ollson.

Royal Military College of Canada 1998. *A Study of Arsenic Contamination from the Royal Oak Giant Mine, Yellowknife, Northwest Territories*. May. M. Eng. Thesis, Ian Stuart Mace.

SENES Consultants Limited 2001a. *Pathways Analysis of Reclamation Alternatives for Collins Bay A-Zone Pond*. Prepared for Cameco Corporation. November.

SENES Consultants Limited 2001b. *Risk Assessment Giant Mine Base Case Scenario: Supporting Document A4 and Attachments.* Prepared for Department of Indian Affairs and Northern Development. May.

SENES Consultants Limited 2000. *Consequence Assessment of a Uranium Solution Accidental Spill to Collins Creek.* Draft Report Prepared for Cameco Corporation. December.

SENES Consultants Limited 1996. *Addendum to the Revised Pathways Analysis of Decommissioning Alternatives for Collins Bay A-Zone, B-Zone and D-Zone Pits and the B-Zone and Eagle Point Waste Rock Piles.* Prepared for Cameco Corporation. January.

SENES Consultants Limited 1995a. *Revised Pathways Analysis of Decommissioning Alternatives for Collins Bay A-Zone, B-Zone and D-Zone Pits and the B-Zone and Eagle Point Waste Rock Piles.* Prepared for Cameco Corporation. January.

SENES Consultants Limited 1995b. *Supporting Document 6: Part A – Midwest Local Environmental Impact Assessment*. Prepared for Cameco Corporation. June.

SENES Consultants Limited 1995c. *Cigar Lake Project Environmental Impact Statement – Regional Cumulative Impact Study*. Prepared for Cameco Corporation. July.

SENES Consultants Limited 1995d. *Supporting Document 6: Part A – Midwest Local Environmental Impact Assessment*. Prepared for Cameco Corporation. June.

SENES Consultants Limited 1993a. *Revised Pathways Analysis of Decommissioning Alternatives for Collins Bay A-Zone, B-Zone and D-Zone.* Prepared for Cameco Corporation. May.

SENES Consultants Limited 1993b. *Revised Pathways Analysis of Decommissioning Alternatives for Collins Bay B-Zone Pit and Waste Rock Pile.* Prepared for Cameco Corporation. July.

SENES Consultants Limited 1989. *Decommissioning the Collins Bay and Eagle Point Mines. Open Pit Decommissioning Collins Bay A-Zone, B-Zone and D-Zone.* Pathways Analysis of Decommissioning Alternatives. Prepared for Cameco Corporation.

Scharer, J.M., R.V. Nicholson, B.E. Halbert and W.J. Snodgrass 1994. *A Computer Program to Assess Acid Generation in Pyritic Tailings.* Environmental Geochemistry of Sulfide Oxidation (C.N. Alpers and D.W. Blowes, eds.), ACS Symposium Series, 550: 132-152.

Scharer, J.M., V. Garga, R. Smith and B.E. Halbert 1991. *Use of Steady State Models for Assessing Acid Generation in Pyritic Mine Tailings.* Proceedings of the 2nd International Conference on the Abatement of Acid Mine Drainage. 2:211-229, Montreal PQ.

Schecker, W.D. and D.C. McAvoy 1994. *MINTEQL; A Chemical Equilibrium Model for Personal Computers.* Environmental Research Software, Hallowell, ME.

Schnoor, J.L. 1996. *Environmental Modeling*. John Wiley and Sons Inc. New York.

Sheppard, M.I. and D.H. Thibault (1990). *Default Soil Solid / Liquid Partition Coefficients K_ds for Four Major Soil Types.* Health Physics, 59: 471-482.

Stumm, W. 1992. *Chemistry of Solid-Water Interface.* John Wiley and Sons, Inc. New York, NY.

Stumm, W. and J.J. Morgan 1981. *Aquatic Chemistry. 2nd Edition.* John Wiley and Sons, Inc. New York, NY.

Thibodeaux, L.J. 1996. *Environmental Chemodynamics. 2nd Edition.* Wiley Interscience Publication. John Wiley and Sons, Inc. New York, NY.

Thomann, R.V. 1972. *Systems Analysis in Water Quality Management*. McGraw-Hill, New York.

Wetzel, R.G. 1975. *Limnology.* Saunders College Publishing, Philadelphia, PA, U.S.A.

Wiley, R. 1960. *Advanced Engineering Mathematics.* McGraw-Hill, New York, N.Y.

APPENDIX C

METAL BIOAVAILABILITY

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APPENDIX C METAL BIOAVAILABILITY

C1.0 LITERATURE REVIEW

C1.1 BACKGROUND

The extent that metals "adsorb onto or absorb into and across biological membranes of organisms expressed as a fraction of the total amount of metal the organism is proximately exposed" is called bioavailability (Dresler *et al.*, 2003). Scientists involved in risk assessment or risk decision making tasks differentiate between absolute and relative bioavailability. Absolute bioavailability (ABA) is conventionally defined as the fraction of an externally administered dose of a metal that reaches the systemic circulation or a central compartment of the receptor, whether through the gastrointestinal tract, skin or lungs (NEPI 2000, Dresler *et al.*, 2003). Evaluation of the effects of different routes of exposure on the bioavailability of a compound, usually determined under controlled environmental conditions, is known as relative bioavailability (RBA). Relative bioavailability represents the comparative bioavailability of different forms of a compound (e.g., metal salts), different doses, different physiological state of the receptor, or different exposure media (e.g., soil vs. water) (NEPI 2000). Consideration of relative bioavailability allows intakes to be adjusted to reflect differences in the bioavailability of a compound in various exposure media (MOE 2001).

In addition to bioavailability terminology discussed above, the term bioaccessibility is also used. The bioaccessible fraction (BF) is often interpreted, albeit incorrectly, as a measure of bioavailability. Bioaccessibility is based on *in vitro* studies and it refers to the fraction of potentially available particulate matter (generally 250 μm or less in diameter) that may be adsorbed or absorbed by an organism upon contact (Dresler *et al.*, 2003). Dissolved arsenic species in contaminated water, drinking water for example, are considered 100% bioaccessible. In order to become bioaccessible, particulate matter must dissolve during the test procedure. Some dissolved metals, however, may re-precipitate. Thus, the bioaccessible fraction represents the maximal amount of metal that remains in solution after the test procedure is completed. (Hamel *et al.*, 1999). For most metals, the BF value for the same soil sample is considerably larger than RBA, thus, the former is a conservative estimate of the latter

Oral exposures are described in terms of an external dose or intake, as opposed to an absorbed dose or uptake (MOE 2001). Intake occurs as an agent enters the body of a human or animal without passing an absorption barrier (e.g., through ingestion or inhalation), while uptake occurs as an agent passes across the absorption barrier (IPCS 2000). Not all materials (e.g., metals, nutrients) that enter the body as intake are absorbed into the body as uptake. Many are passed through the body and expelled without effect.

A number of different methods have been employed to evaluate the bioavailability of metals for humans. *In vivo* methods utilizing animal models (mammals, invertebrates) and plants have been developed, but they are expensive, time consuming, and require specialized facilities. It has become evident that extrapolation from one soil sample to another or from one species to another is a tenuous task (Kapuska *et al.*, 2003). Therefore, *in vitro* methods for assessing the oral bioavailability of metals in impacted soils and subsequently in exposure assessments of the human digestive system have been under active development for the past decade. The test is normally a two-step procedure, the first step emulating the stomach and the second the intestines. These *in-vitro* assays are referred to as relative bioaccessibility or simply as bioaccessibility.

Oomen *et al.* (2002) compared five European *in vitro* digestion methods to assess the bioacessibility of three soil contaminants (arsenic, cadmium and lead). While each laboratory reported very similar total metal concentrations, the bioaccessibility factors varied significantly. They found that the most significant contributor to the difference is the test pH value. Low pH exposure as a first step (pH of 1.1 to 2.0) greatly enhanced bioaccessibility. There was some evidence that organic complexation of the metals during the second experimental step (intestines under neutral pH conditions) also enhanced bioaccessibility. The reproducibility of the results was reasonably good.

The Ontario Ministry of the Environment (MOE) carried out similar tests to determine the bioaccessibility of metals in soils from the Port Colborne area in southern Ontario (MOE 2001). The results of this test work are summarized below.

C1.2 MOE METAL LEACH TESTS

The MOE used two methods to assess the bioaccessibility of soils. The first test involved the fasting stomach simulation test based on a 24-hour extraction. The second test involved the simulation of the stomach as well as the small intestine.

C1.2.1 Simulated Stomach Acid Leach Test

The acid leach test is used to determine the amount of metal that would be leached into the stomach from soil samples. In the test procedure followed by the MOE (2001), the pH was adjusted to a value of 1 using 0.17 N HCl to simulate the action in the stomach and the test was carried out for a 24 hour period. The 24 hour digestion period is longer than the typical residency time for food in the stomach and thus overestimates the amount of metal that would be released and available.

Ten soil samples were selected for the stomach acid leach test. The results of this test are shown in Table C1.2-1. As seen from the table, lead is almost 100% bioavailable, while the maximum leachable fractions of the other metals are in the range of 20 to 45%. Antimony is an exception with less than 4% being available.

TABLE C1.2-1
RESULTS OF SIMULATED STOMACH ACID LEACHATE TEST

Metal	Minimum % leached	Maximum % leached
Antimony	2.1	3.9
Arsenic	12.6	28.4
Cobalt	12.5	24.7
Copper	31.8	44.4
Lead	61.1	90.3
Nickel	11.8	23.3

Source: MOE 2001.

C1.2.2 Simulated Stomach and Intestine Leach Tests

The second set of tests was carried out to simulate the environment in the stomach and the small intestine. This bioaccessibility test procedure simulates the acid environment in the stomach followed by the neutral environment of the small intestine; thus an acid extraction was carried out first and followed by neutral extraction. In the test, acid extraction was carried out under acidic conditions at pH 1.5 for 1 hr; a sample was removed for analysis; then the liquid was adjusted to pH 7 and enzymes were added (bile salts and pancreatin). The neutralized sample was extracted for 4 hours and followed by subsequent chemical analysis.

Both sieved and unsieved soil samples were used in these tests. The results presented in Table C1.2-2 are the overall summary of these different soil fractions. Ten samples were also selected for these tests. The samples were sieved because it is believed that particles < 20μm in size are more likely to adhere to human hands and become ingested during hand-to-mouth activity.

TABLE C1.2-2
RESULTS OF *in vitro* BIOACCESSIBILITY TESTS

Metal	Leached fraction in acid extraction of Port Colborne soils (%)	Leached fraction in neutral extraction of Port Colborne soils (%)	Choice by MOE (%)	Leached fraction from reference Montana contaminated soil (%)
Antimony	24 (5.2 – 84)	31 (6.1 – 64)	32	38.3
Arsenic	33 (19 –51)	27 (8.5 – 56)	-	59.5
Beryllium	55 (39 – 81)	10 (1.9 – 15)	59	41.6
Cadmium	73 (58 – 86)	56 (4 – 79)	76	93.4
Cobalt	26 (15 – 35)	18 (7.5 – 26)	29	44.9
Copper	32 (0.4 – 46)	41 (8.8 – 60)	43	46.3
Lead	71 (34 – 86)	5 (1.3 – 14)	-	80
Nickel	16 (7.6 – 28)	16 (9.2 – 24)	19	25.7

Source: MOE 2001.

As seen in Table C1.2-2, much higher fractions of beryllium, cadmium and lead were extracted under acidic conditions then when the extraction solution was neutralized. The overall bioaccessibility of arsenic, cobalt and nickel did not change significantly between the acid and neutral extractions. For antimony and copper, there was a slight increase in the extraction at neutral pH. The table also shows the final bioaccessibility values selected by the MOE for the various metals in Port Colborne.

The MOE provided analysis of bioaccessibility of metals from a Montana soil to validate their results. These values are also presented in Table C1.2-2. As seen in the above table, the bioavailability of the metals ranged from approximately 19% to 76% in Port Colborne soils compared to a range of 26% to 93% in Montana soils.

C1.3 ARSENIC BIOAVAILABILITY

As arsenic is the principal contaminant of concern at the Giant Mine site, this section focuses on investigations into the bioavailability of arsenic in soils.

C1.3.1 General Considerations

Arsenic is a metalloid with oxidation state ranging from –2 to +5. Most risk from arsenic is associated with the forms of arsenic that are biologically available for absorption into the body or "bioavailable" to humans. These mobile forms of arsenic are mainly oxy-anions that are subject to ferrihydrite adsorption under neutral to acid pH conditions (Langmuir *et al.*, 2003). The complexation of mobile forms of arsenic with organics or precipitation as a solid phase in the intestines is not likely. Presently, precise methods are not available to quantify the percentage of bioavailable arsenic in soils for humans. Most baseline risk assessments developed for contaminated sites have used the conservative assumption that 100% of the arsenic present in soils is bioavailable. However, arsenic may exist in many geochemical forms (e.g., oxides, arsenides, arsenosulphides) and physical forms (e.g., dust, slag, tailings, calcine, waste ore) at hazardous waste sites contaminated by mining and smelter wastes. These waste forms vary in their solubilities and geochemical stabilities and may not be very bioavailable and therefore may pose less risk to humans and other biological receptors.

Recent research indicates that arsenic must be dissolved in order to be absorbed in the body (IPCS 2000) Arsenic compounds in soil will be less available than dissolved arsenic in drinking water. As a result, a bioavailability adjustment factor is deemed to be appropriate to improve the accuracy of the assessment of potential risks associated with ingestion of soil.

Arsenic occurs in soil as a complex mixture of solid phase compounds of varying particle size and morphology. These compounds include various mineral phases: co-precipitated and sorbed species associated with soil minerals or organic matter, and dissolved species that may be complexed by a variety of organic and inorganic ligands. The occurrence and relative distribution of an element among these various phases, and the physical relation between the phases and the soil will control arsenic dissolution properties and, hence, its bioavailability.

Thus, in the last several years, considerable effort has been applied to developing methods for measurement of bioavailability of arsenic with the hope that the results shed light on the uncertainty issues in risk assessment and identify chemical amendment options that would reduce the bioavailability of arsenic. Although these studies have not provided a solid correlation between the arsenic species in soil and bioavailability, a fundamental understanding has been developed.

Bioavailability of soil arsenic has been studied in two experimental categories: *in vivo* and *in vitro* experiments. *In vivo* studies have involved test animals, such as swine, rabbit, monkey, and rat, to model the bioavailability of oral arsenic. *In vitro* experiments have measured the extent of dissolution of arsenic under simulated gastrointestinal conditions. *In vitro* experiments include the *in vitro* gastrointestinal (IVG) method and the physiologically based extraction test (PBET).

It is important to note that the *in vitro* method is not designed to supplant bioavailability studies using animal models, but rather to estimate arsenic bioavailability when animal study results are not available. It should also be noted that the *in vitro* method is a screening-level test and does not mimic the entire physiological process controlling uptake of arsenic.

Measurements from *in vitro* methods are sometimes referred to as bioaccessibility rather than bioavailability. This reflects the fact that the dissolved arsenic may not be 100% bioavailable. In calculating the relative bioavailability in animal studies, absolute bioavailability of soil arsenic is compared with absolute bioavailability of dissolved arsenic. For this purpose, absolute bioavailability of dissolved arsenic has been measured in most of the studies. The measured absolute bioavailability of dissolved arsenic ranges between 65% and 98% in most cases.

C1.3.2 Human Bioavailability Studies

Research work carried out since 1992 on arsenic was reviewed to identify pertinent information on human bioavailability investigations. The results from these studies were summarized in Table C1.3-1. It should be mentioned that there is large uncertainty in the bioavailability data from both *in vivo* and *in vitro* studies. One such uncertainty is whether bioavailability study results serves as a sufficiently valid predictor of human response. One of the research studies

reviewed, addressed this by comparing pharmacokinetic and excretion behaviour of sodium arsenate in the *Cebus* monkey with previous studies in human volunteers. Although the results were quite similar, it is always possible that there could be species differences when arsenic is present in a soil matrix. Unfortunately, there are no reliable measurements of arsenic bioavailability from soil samples in human subjects to serve as a basis for comparison, making true validation of animal models difficult.

Another uncertainty relates to the possible effect of the arsenic concentration in soil on bioavailability. Existing measurements of relative bioavailability of arsenic use soil samples with arsenic concentrations in the hundreds or thousands of parts per millon. These high levels are needed to ensure that there is sufficient arsenic dose to the experimental animal for detection of a response, particularly for blood measurement methods. This raises the question of whether bioavailability measurements at these concentrations are predictive of the bioavailability in lesser-contaminated soils. It is probable that arsenic bioavailability from soils is concentration dependent, but this issue has not been well studied.

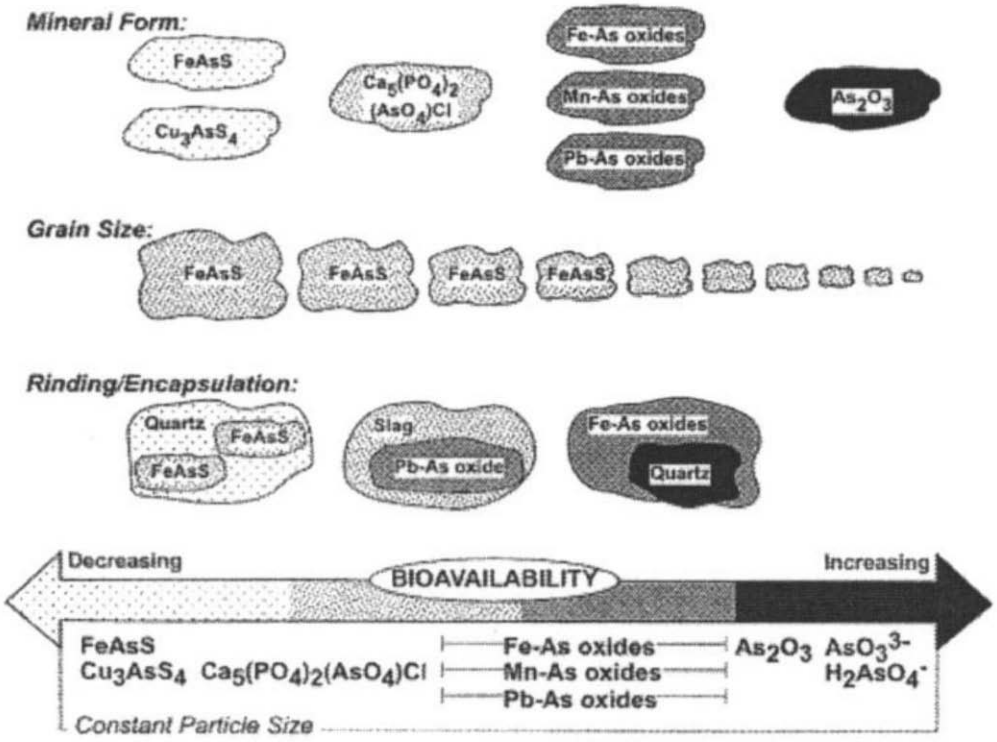
The effect of the physical and chemical states of arsenic on its bioavailability is illustrated in Figure C1.3-1. The feeding status of the animals during bioavailability measurements is also important. In most of the animal studies, soil is administrated on an empty stomach. Administering the dose on an empty stomach aids in reducing the variability associated with the bioavailability measurements, but probably does not accurately mimic the circumstances under which humans ingest soil. If the presence of food diminishes the bioavailability of arsenic from soil, as might be expected, then the measurements as conducted in these studies are upper-bound estimates and therefore useful for regulatory purposes. However, this is speculative, and basic information on the effects of food on bioavailability from soils is lacking. Previous studies of interactions between metals and partially digested stomach matter indicate that such material may adsorb metals following their dissolution from soil particles, thus reducing net bioavailability. In addition, an empty stomach is associated with lower pH, which increases the solubility of soil arsenic. The simulative *in vitro* method may, therefore, be considered to represent a worst-case scenario, yielding predictive arsenic absorption values which significantly exceed normal *in vivo* levels. However, the results from these studies do not confirm this speculation.

Given the variability in soil-associated arsenic compounds, soil matrices, animal models, and experimental design, further research of bioavailability of arsenic through soil ingestion is warranted. Also, limited knowledge is available regarding differential absorption and metabolism of arsenic in juvenile versus adult animals at the present time. FIFRA SAP[1] (2001) recommended a preliminary value of 25% as representative relative bioavailability of arsenic

[1] The Federal Insecticide, Fungicide, and Rodenticide Act (FIFRA), Scientific Advisory Panel (SAP).

FIGURE C1.3-1

SCHEMATIC OF HOW DIFFERENT ARSENIC SPECIES, PARTICLE SIZES AND MORPHALOGIES AFFECT ARSENIC BIOAVAILABILITY



*SOURCE: Ruby et al (1999)

TABLE C1.3-1 – SUMMARY OF ARSENIC BIOAVAILABILITY STUDIES SINCE 1992

Study	Method	Soil Type	Relative Bioavailability	Arsenic Concentrations (mg/kg)	Arsenic Species	Note
Freeman *et al*., 1993	*In vivo*, rabbit	Smelter area soils	28% (with soil),	3.9		Based upon Urinary Excretion
			48% (with soluble form)	1.95	Sodium arsenate	
Lorenzana *et al*., 1996	*In vivo,* swine	Smelter/mining area residential soil	78%	1600		Based upon blood AUC
		Smelter/Mining area slag	42%	10100		Based upon blood AUC
Battelle, 1996	*In vivo*, swine	Mining area slag	Not Detected			
Casteel *et al*., 1997a-b	*In vivo*, swine	Soils and mining area wastes	<50%			
	In vivo, swine	Tailing	37%	150		
Casteel *et al*., 1996a-d	*In vivo*, swine	Berm soil	62% ± 55%	67		
	In vivo, swine	Residential soil	34%-52%	67		
	In vivo, swine	Slag	18%-51%	590-700		
Casteel *et al*., 1998a-e	*In vivo*, swine	Residential soil	10%	240		
	In vivo, swine	Slag	15% ± 1%	1050		
Casteel *et al*., 2000	*In vivo*, swine	Residential soils	18%-45%			
Freeman *et al*., 1995	*In vivo*, monkey	Mining area soils	20%			Based upon Urinary Excretion
		Mining area dusts	28%			Based upon Urinary Excretion
Roberts *et al*., 2001	*In vivo*, monkey	Electrical substation soils	14.6% ± 5.1%			Based upon Urinary Excretion
		wood preservative treatment site soils	16.3 %± 6.5%			Based upon Urinary Excretion

TABLE C1.3-1 – SUMMARY OF ARSENIC BIOAVAILABILITY STUDIES SINCE 1992 (CONT'D)

Study	Method	Soil Type	Relative Bioavailability	Arsenic Concentrations (mg/kg)	Arsenic Species	Note
Roberts *et al*., 2001	*In vivo*, monkey	Cattle dip vat site soils	24.7% ± 3.2%			Based upon Urinary Excretion
		Pesticide sites I soils	10.7 %± 4.9%			Based upon Urinary Excretion
		Pesticide sites II soils	17.0 %± 10.0%			Based upon Urinary Excretion
Rodriguez *et al*. 1999	*In vitro* gastrointestinal, IVG	Mining/smelter calcine	2.7%-30.1%	6250-17500	40% As jarosite 60% associated with iron and manganese	
	In vitro gastrointestinal, IVG	Mining/smelter slag	16.4%-30.1%	405-4650	17% As jarosite 53% associated with iron and manganese, 30% associated with lead	
Ruby *et al*. 1996	Physiologically based extraction test (PBET)	Copper smelter residential soil	44%-50%			pH=1.3
	Physiologically based extraction test (PBET)	Copper smelter residential soil	31%-32%			pH=2.5
	Physiologically based extraction test (PBET)	Copper smelter house dust	34%			pH=2.5
	In vivo, monkey	Smelter house dust	28%			Based upon Urinary Excretion

TABLE C1.3-1 – SUMMARY OF ARSENIC BIOAVAILABILITY STUDIES SINCE 1992 (CONT'D)

STUDY	METHOD	SOIL TYPE	RELATIVE BIOAVAILABILITY	ARSENIC CONCENTRATIONS	ARSENIC SPECIES	NOTE
Groen *et al*. 1994	*In vivo*, dogs	Bog ore soil,	8.3% (absolute)		Associated with clay surface and iron and aluminum oxides	Based upon Urinary Excretion
Williams *et al*., 1998	Physiologically based extraction test (PBET)	Mine site fluvial soil	16%-25%	1400-2120	Arsenopyrite, As2O3	pH = 2.5 and 7
	Physiologically based extraction test (PBET)	Slug	35.6%	20000		pH = 2.5 and 7
Ellickson *et al*. 2001	*In vivo* rat	NIST SRM 2710 reference soil	37.8% (Absolute)			Urinary Excretion
	Physiologically based extraction test (PBET)	NIST SRM 2710 reference soil	65% (Absolute)			
Borch *et al*. 1994	*In vitro* extraction	Tailings	1.3%			
Ng *et al*. 1996	*In vivo*, rat	Timber treatment site soil	13.5% ± 4.5	52-138		Blood As concentration AUC
	In vivo, rat	Cattle dip site soil	8.1% ± 4	700-2100		Blood As concentration AUC
Ng *et al*. 1998	*In vivo*, rat	Pesticide site soil	1.2-9.6%	32-1600	32-56% 3+	Blood As concentration AUC
	In vivo, rat	Pesticide site rock	1%	435	44% 3+	Blood As concentration AUC

through soil ingestion. This value is based upon the data of Roberts *et al*. (2001) and Freeman *et al*. (1995) using non-human primates as the test subjects. The SAP felt that this value best represent relative bioavailability of inorganic arsenic in soil (based on the use of non-human primates and the physiological similarity in the pattern of metabolism with humans and the use of arsenic-contaminated soil in the study for estimation of bioavailability); however, some members suggested an interim value of 50%, while only one member suggested consideration of the full range of bioavailability for arsenic in soil reported in the literature (near zero to 98%). There are other relative bioavailability studies indicating results ranging from near zero to 50%, with the exception of two soils from Aspen, Colorado, that have higher bioavailabilities with extremely wide confidence intervals (62% ± 55 %, 98% ± 86 %; Casteel *et al*., 1997a; Ruby *et al*., 1999). Note the confidence intervals in the papers refer to individual observations not the confidence intervals of the mean values.

In a baseline risk assessment of a superfund site in Denver, Colorado, a value of 42% of relative bioavailability was used in risk calculations (U.S. EPA, 2001). This value was based on the study of Casteel *et al*. (2000), which reported a mean relative bioavailability of 31% (using a swine model) with a 95% upper confidence limit of 42%.

C1.3.3 Biota Bioavailability Studies

The bioavailability to aquatic and terrestrial organisms of arsenic depends on several physical and chemical factors. The texture and chemical composition of the soil are important factors that govern the availability of arsenic to plants. Iron and aluminum adsorb anionic arsenic species extensively in acidic soils, whereas calcium oxides in alkaline soils adsorb anionic arsenic species to a lesser extent. Anionic arsenic species are therefore in general more available to crops grown in alkaline than in acidic soil. Thus, pH plays a very important factor in determining the bioavailability of arsenic in soil. Soil with a sandy texture normally has a low content of minerals which are capable of binding anionic arsenic species, and therefore a relatively high mobility of arsenic into the soil pore water is expected.

In the aquatic environment, speciation and thus mobility of arsenic, is highly correlated with pH, redox potential, and presence of oxihydroxides of Fe and Mn. Thus, it is appropriate to address the bioavailability through speciation of arsenic. Recent studies on bioavailability of arsenic to aquatic organisms have focused on four arsenic species: As(III), As(V), monomethyl arsenic acid (MMAA), and dimethyl arsenic acid (MAA).

Carbonell-Barrachina *et al*. (1999) reported bioaccumulation factors (BAF) for turnip roots and shoots in a series of experiments under conditions without soil with concentrations of 1, 2, and 5 mg/L of various arsenic species. BAF is evaluated as the concentration ratio of mg (As)/kg biomass to mg/L aqueous solution. The root BAFs for As(III), As(V), MMAA, and DMAA were 9, 10, 7, and 5 L/kg respectively. The shoot BAFs were 2.5, 4, 6, and 7.5 L/kg respectively.

Carbonell *et al*. (1998) reported bioavailability of arsenic to wetland vegetation. BAFs reported for *Spartina Patens*, grown at 0, 0.2, 0.8, and 2.0 mg/L of various arsenic species. Results indicated root BAFs of 200, 150, 140, and 2 L/kg for As(III), As(V), MMAA, and DMAA, respectively. The shoot BAFs were 12, 10, 10, and 5 L/kg respectively. These results indicate that As(III) and As(V) are the most bioavailable form of arsenic species. DMAA is the least available species to the roots. The results also indicate that while DMAA easily translocates from roots to shoots, other species do not translocate to shoots effectively.

Canivet *et al*. (2000) reported that BAFs of As(III) in several macroinvertebrates ranges from 1 to 1000. In a series of experiments, after exposure for 10 days to 100 µg/L of As(III), the concentrations of arsenic in *G. Fossarum, H. Sulphurea, P. fontinalis, A. aquaticus, N. rhenorhodanensis,* and *H. pellucidula* were 20.3, 0.12, 1.9, 26.5, 22.6, and 107.2 mg/kg. The corresponding BAF were 200, 1, 20, 260, 220, and 1000 L/kg respectively.

While the BAFs, reported in literature, span orders of magnitude among organisms, the variation is partly attributed to the lack of arsenic speciation information. The above results along with other available literature data can be used as a rough estimate of BAFs of arsenic species to aquatic and terrestrial species.

C2.0 SITE SPECIFIC BIOACCESSIBILITY STUDIES

The availability of metals in the sediments of Baker Creek, which flows through the Giant Mine site, was evaluated by two extraction methods. The first method, the sequential extraction method, was developed for geochemical partitioning of soil samples. The test is designed to differentiate between various physico-chemical forms of arsenic in the test sample. The second method, the gastric fluid extraction procedure is a two-step method designed to mimic the mammalian gastro-intestinal tract. The procedures and the results of the tests are presented below.

C2.1 EXTRACTION TEST PROCEDURES

C2.1.1 Sequential Extraction Procedure

The sequential extraction test procedure developed by Tessier *et al.* (1979) was employed to partition metal binding in sediments into the following five fractions:

Ion exchangeable fraction (Extract 1)

- Extraction with $MgCl_2$ (magnesium chloride) or NaAc (sodium acetate).
- Metal release by ion exchange of adsorbed ions on minerals surfaces or with carboxyl or hydroxyl organic ligands.

Acid soluble fraction (Extract 2)

- Extraction with HAc (acetic acid) and NaAc (sodium acetate) at pH of 4.5 – 5.0.
- Metal release by dissolution of carbonates.

Metal oxide bound fraction. (Extract 3)

- Extraction with NH_2•HCl (hydroxylamine hydrochloride) and sodium citrate in acid solution
- Metal release by dissolution of reducible metal oxides (amorphous iron hydroxide, hematite, manganese oxides) and metal oxy-anions (manganates, chromates).

Oxidizable fraction (Extract 4)

- Extraction with H_2O_2 (hydrogen peroxide) and HNO_3 (nitric acid).
- Metal release by oxidative dissolution of sulphides, selenides, oxidizable oxides, arsenides, organics containing covalently bound metals, metal chelates and coordination compounds, metals entrained in living or dead organisms.

Residue (Extract 5)

- Extraction with aqua regia (HNO_3/HCl) or perchloric acid.
- Metal release from the crystal lattice of silicates and other non-reactive, stable minerals. These minerals are not expected to dissolve readily under conditions normally expected in nature.

The test procedure is designed to measure the relative leachability of the metals from readily leachable (Extract 1) to least leachable (Extract 5). Additional extraction tests were performed using one step (aqua regia) extraction only. This test is designed to give the total extractable arsenic content. The analytical results of the sequential extraction tests are summarized in Attachment 1.

C2.1.2 Gastric Fluid Extraction Procedure

This extraction test is an attempt to mimic the human gastro-intestinal tract and was employed previously for analyzing Giant Mine soil samples by Ollson *et. al.* (2003). A slightly modified version of the test was used for analyzing sediment samples in this study. The test procedure is detailed below.

Dry sediment samples were ground and homogenized using mortar and pestle. Exactly 1.0 gram of sample was weighed into a 50 ml centrifuge tube, to which 20 ml of solution at pH 1.8 (adjusted with HCl) containing 1.25 g/L pepsin, 0.5 g/L sodium citrate, 0.5 g/L malic acid, 1 mL/L glacial acetic acid, and 0.15 M NaCl was added.

Samples were extracted for 1 hr at 37°C on a reciprocating shaker. After 1 hour, each solution pH was adjusted to pH 7 with saturated Na_2CO_3 solution, using a pH strip, and double checked with a pH meter. After pH adjustment, 1 mL solution containing 0.18 g/L bile extract and

0.05 g/L pancreatin was added to each test solution. The samples were then further extracted for 4 hours on a reciprocating shaker at 37°C. Following the second extraction, the tubes were centrifuged at 3800 rpm for 30 minutes and the supernatant was stored frozen at – 20°C.

For the analysis of metal and metalloid content, 9 mL of the thawed extract was mixed with 1 mL of concentrated ultra pure HNO_3 and digested by boiling for 10 minutes to ensure that all species are converted to their highest oxidation state, As(V) for example. The samples were then cooled and made up to 10 mL using 1.0 M ultra pure HCl. For metal analysis, the samples were then appropriately diluted with 1 M HCl and analyzed by direct current plasma–atomic emission spectrometry (DCP-AES). To suppress matrix interference, a single metal cassette was employed for each target metal analysis. Standard reference samples in the blank extract solutions were within 3% of the target values.

The gastric extract (in µg / g) is calculated by the following equation:

$$Gastric\ extract\ (\mu g / g) = \mathrm{DF}\,\frac{21}{9}\,\mathrm{x}\,10\,\mathrm{C} \qquad \text{(C2-1)}$$

Where: C = test metal concentration (µg / mL)
DF = dilution factor

The gastric extraction of antimony, arsenic, copper, nickel, and zinc was determined in this manner. The relative bioaccessibility, estimated as percent (%), was calculated as the ratio of the gastrict extract (µg/g) and the total arsenic extract (also in µg/g) of the corresponding one step extraction procedure.

C2.2 BAKER CREEK SEDIMENT INVESTIGATION

C2.2.1 Sediment Sampling Program

The Baker Creek sediment-sampling locations and the sampling protocol are described in detail by Dillon Consulting (2004). Triplicate sediment samples were collected at seven locations using a modified trowel during a field campaign in August 2003. The samples were placed in sterilized sediment sampling jars and refrigerated immediately after collection. Samples were sent by Dillon Consulting to Taiga Environmental Laboratories, Yellowknife, NWT, for total metal and size distribution (percent sand, silt, and clay) analysis.

In May 2004, 14 samples comprising two sets of the triplicate samples collected on the seven sites were sent to SENES Consultants for further analysis. These samples were logged and renumbered as indicated in Table C2.2-1 prior to shipment to SGS Lakefield Research Ltd. for total metal analyses and sequential extraction test work. A sub-set of the samples was sent to the

Department of Chemical Engineering, University of Waterloo, Waterloo, Ontario, for gastric fluid extraction test work.

TABLE C2.2-1
BAKER CREEK SEDIMENT LOG

Site	Taiga ID	SENES ID	Observations
1	233004	A,1,8,9	high bioturbation, sandy, low organics, water cover
2	233005	B,2	Dark grey, sandy, organics, water cover
3	230006	C,3	dark grey, consolidated,, high organics water cover
4	230007	D,4	Dark grey, sand/silt, high organics, water cover
5	230008	E,5,10,11	bioturbation, gray, silt/clay, no organics, water cover
6	230009	F,6	gray, silt/clay, no organics, oxidation rind, water cover on 6
7	230010	G,7	gray, silt/clay, no organics, oxidation rind, water cover on 7

The letter designations (A to G) and the number designations (1 to 7) represent duplicate samples (field duplicates) taken from the same locations (e.g. samples A and 1 were collected at the same location). Number designations 8 to 11 represent multiple analyses (lab duplicates) performed on the same samples. Thus, four sets of analytical results were generated for sampling locations 1 and 5 and two sets for the other locations. The types of tests performed including total metals, total organics, partitioned metals, and bioaccessible metals are provided below in Table C2.2-2.

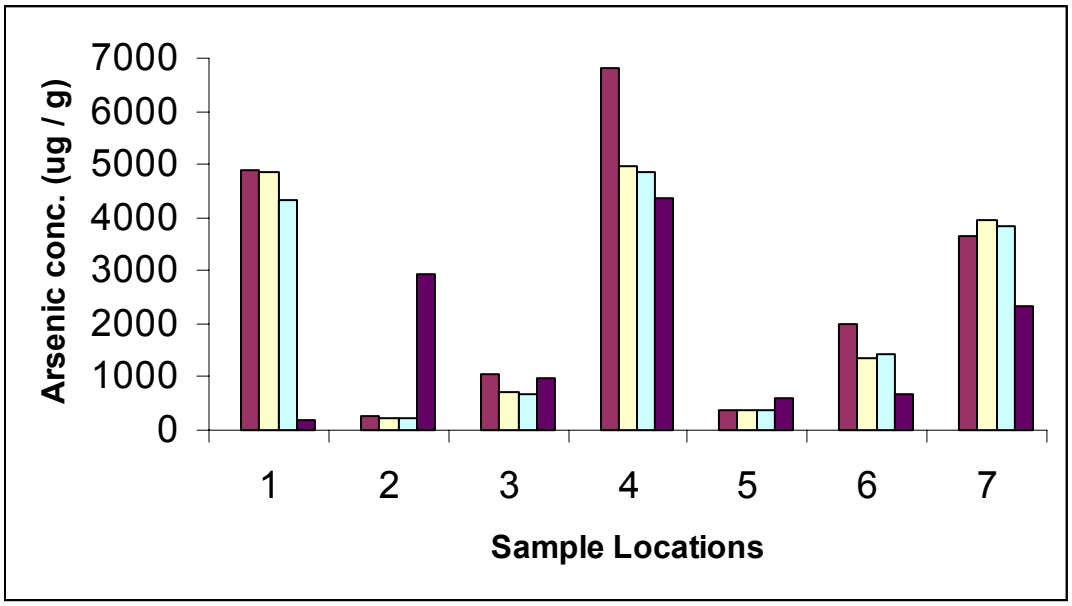
TABLE C2.2-2
SEDIMENT TESTING SCHEDULE

Sample	Testing	Laboratory
A – G	Total metals – aqua regia	SGS Lakefield
A – G	Total organic content	SGS Lakefield
1 – 11	Total metals – aqua regia	SGS Lakefield
1 – 11	Partitioned metals – Tessier sequential	SGS Lakefield
1 – 11	Bioaccessible metals – gastric fluid	Univ. of Waterloo
1 – 11	Total organic content	SGS Lakefield

C2.2.2 Sediment Analytical Results

The results of the total arsenic measurements on the samples from the 7 locations on Baker Creek are summarized in Figure C2.2-1. The first column for each location refers to the lettered samples (A to G). The second column presents the analytical results for the numbered samples (1 to 7). The concentrations at location 1 and location 5 are average values of the triplicate analyses. The third column is the sum of the sequential extraction tests while the fourth column is the analytical results reported by Dillon (2004) on one set of the triplicate samples which was analyzed by Taiga Laboratories.

FIGURE C2.2-1
TOTAL ARSENIC CONCENTRATIONS IN BAKER CREEK SEDIMENTS



It is evident that the total arsenic level varies substantially from one sample location to the next. Generally, highest arsenic levels were found at location 4 while lowest levels were observed at locations 2 and 5. The relationship between the analytical results at the seven locations was analyzed by calculating the Pearson correlation matrix (Legendre and Legendre, 1998) and are reported in Table C2.2-3.

TABLE C2.2-3
PEARSON CORRELATION MATRIX OF MEASURED TOTAL ARSENIC CONCENTRATIONS

	SET 1	SET 2	SET 3	SET 4
SET 1	1	0.961	0.971	0.424
SET 2	0.961	1	0.997	0.297
SET 3	0.971	0.997	1	0.345
SET 4	0.424	0.297	0.345	1

Where:

SET 1 = total arsenic results of the lettered sample set (A – G)
SET 2 = total arsenic results of the numerical sample set (1 – 7)
SET 3 = sum of sequential extraction arsenic results of the numerical sample set (1 – 7)
SET 4 = total arsenic results reported by Dillon (2004).

The critical value of the Pearson Correlation coefficient for Table C2.2-3 (p = 0.05, degrees of freedom = 5) is 0.875. It is evident from Table C2.2-3 that the correlations between the three sets of data reported by SENES are highly significant. However, none of the datasets correlated with the analytical results as reported by Dillon (2004).

To test the number of mismatched data pairs, a paired comparison between total arsenic reported for SET 2 and SET 4 (from Dillon, 2004) was made. Since the pairs of data representing different locations are highly variable (see Figure C2.2-1), the use of parametric tests developed for normally distributed populations (ANOVA, for example) is not valid. This is supported by estimates of the highly variable variances (σ^2) for the four measurements at two locations (i.e. σ^2 = 4758 at location 1 and σ^2 = 283 at location 5). For this reason, pair-wise comparison using the chi squared test (Montgomery *et. al*., 2001) was performed. This is a non-parametric test comparing matched pairs of data. It is designed to determine whether two measurements of a matched pair are sufficiently different to be considered as statistically significant. The results of the statistical analysis indicated that the current set of measurements and those reported by Dillon (2004) were significantly different at all 7 locations at the 95% confidence level. In contrast, none of matched data pairs (i.e. SET 2 and SET 3) of the present study were found to be significantly different using the same statistical method. Consequently, the analytical results and conclusions by Dillon regarding total arsenic levels in sediments were not considered further.

The analytical results of arsenic extraction in the present program are detailed in Table C2.2-4. The total arsenic content determined as the sum of the sequential extraction test was found to match almost perfectly with the one-step extraction using aqua regia. The Pearson correlation coefficient of 0.997 (see Table C2.2-3) is significant on a better than 99% significance level. The combination of the first three fractions of the sequential extraction studies comprises the leach fraction known as the acid extract. This includes the ion exchangeable, carbonate and oxide/hydroxide bound arsenic moieties. This fraction has been used by some in the past as an *in-vitro* estimate of bioaccessible arsenic. It is believed to mimic metal extraction in the acidic environment of the adult human stomach (MOE, 2001). The wide experimental range of 5% to 66% acid extraction is in line with the 19% to 51% range reported by the Ontario Ministry of the Environment (MOE, 2001).

TABLE C2.2-4
ANALYTICAL RESULTS OF ARSENIC EXTRACTION FROM BAKER CREEK SEDIMENTS

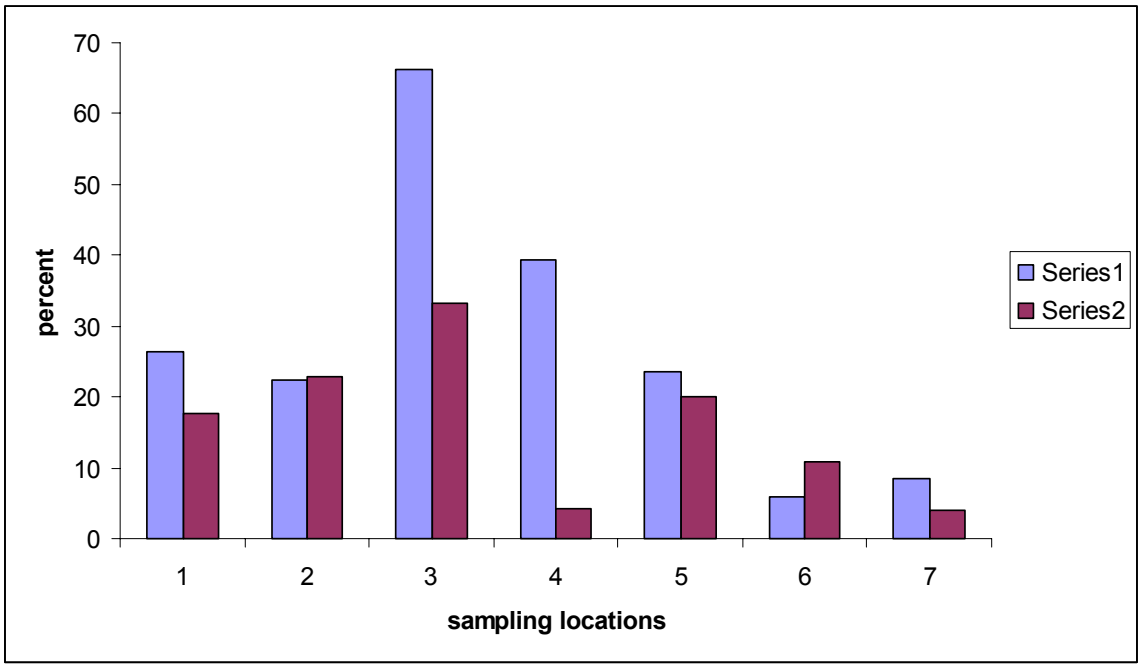
ARSENIC Sequential Extraction (µg/g)											
EXTRACT #	Sample 1	Sample 2	Sample 3	Sample 4	Sample 5	Sample 6	Sample 7	Sample 8	Sample 9	Sample 10	Sample 11
1	71	< 15	30	46	< 15	< 15	< 15	58	82	< 15	< 15
2	128	< 15	37	47	< 15	22	35	125	129	< 15	< 15
3	1020	51	371	1820	< 15	60	288	976	888	96	82
4	2190	140	180	2480	270	980	2000	2340	2220	210	220
5	937	36	44	478	110	352	1530	1110	952	75	71
Total	4346	227	662	4871	380	1414	3853	4609	4271	381	373
TOTAL ARSENIC One-step Extraction (µg/g)											
Aqua regia	4920	240	720	4960	370	1350	3970	4850	4770	390	350
ARSENIC Gastric Fluid Extraction (µg/g)											
Bioaccessibility	803.6	54.6	238.8	214.4	67.4	147.4	155.6	893.4	887.4	82	67.2
ARSENIC Sequential Extraction (%)											
EXTRACT #	Sample 1	Sample 2	Sample 3	Sample 4	Sample 5	Sample 6	Sample 7	Sample 8	Sample 9	Sample 10	Sample 11
1	1.6	<5.0	4.5	0.9	<5.0	<1.0	<1.0	1.3	1.9	<5.0	<5.0
2	2.9	<5.0	5.6	1.0	<5.0	1.6	0.9	2.7	3.0	<5.0	<5.0
3	23.5	22.5	56.0	37.4	<5.0	4.2	7.5	21.2	20.8	25.2	22.0
4	50.4	61.7	27.2	50.9	71.1	69.3	51.9	50.8	52.0	55.1	59.0
5	21.6	15.9	6.6	9.8	28.9	24.9	39.7	24.1	22.3	19.7	19.0
cumulative (1-3)	**28.0**	**22.5**	**66.2**	**39.3**	<5.0	**5.8**	**8.4**	**25.1**	**25.7**	**25.2**	**22.0**
ARSENIC Gastric Fluid Extraction (%)											
Bioaccessibility	**16.3**	**22.8**	**33.2**	**4.3**	**18.2**	**10.9**	**3.9**	**18.4**	**18.6**	**21**	**19.2**

The results of the gastric fluid extraction test work carried out at the University of Waterloo are included in Table C2.2-4 for comparison with the other datasets. Comparison of the gastric fluid and total arsenic analyses by the aqua regia method were found to correlate well. The calculated Pearson correlation coefficient of 0.765 is significant at the 95% confidence level (critical value = 0.602, DF = 9, p = 0.05). This means that higher total arsenic level in the sample tends to yield higher gastric extract concentrations. A positive correlation between total arsenic levels and the gastric fluid extracts was also observed by Ollson *et. al*. (2003). In their case, the correlation was weaker and the authors warned about drawing general conclusions.

The relative bioaccessibility values based on the acid extraction and the gastric fluid extraction procedures are summarized in Figure C2.2-2. The results presented on the figure for the sequential extraction test are the cumulative totals of the first three extractions. With one notable exception (location 6), the acid extract-based bioaccessibility tends to be the same or higher than

bioaccessibility based on gastric extraction. On the average, the difference between acid and gastric extracts is nearly 10%. It is also evident that higher acid extraction values tend to result in higher gastric extraction values. Excepting sediment data from sampling location 4 (a possible anomaly), the Pearson correlation (r = 0.898) is highly significant. The bioaccessibility based on gastric extracts ranged between 3.9% and 33%. A somewhat larger range (1% to 60%) was reported in various Yellowknife NT soils by Ollson et al (2003). These results indicate mineralogical constraints on bioaccessibility.

FIGURE C2.2-2
RELATIVE BIOACCESSIBILITY OF BAKER CREEK SEDIMENTS



Series 1 – acid extractions test results for steps 1 to 3.
Series 2 – gastric fluid extraction test results.

Davis *et al.* (1996) also noted that the bioavalability of arsenic minerals is highly variable. They found that bioavailability was related, to a large extent, to mineral solubility at a given pH. The solubility, hence the bioavailability of metal-arsenic sulphides and phosphates is particularly low; thus bioavailability is controlled by the sparingly soluble nature of these minerals. This explains the relatively low bioaccessibility of arsenic in sulphide-rich tailings (2.9%) at Yellowknife as reported by Ollson *et al.* (2003). It is noteworthy that both the acid and the gastric extract values in sediments at locations 6 and 7 resemble those typically found in tailings. Silicate rinds in some samples may have provided kinetic hindrance to dissolution in addition to solubility limits (Davis *et al.,* 1996).

The solubility of many metal arsenates is known to be amphoteric with minimum solubility between pH 3.5 and pH 7. Thus, arsenates that dissolved at acid pH may form iron and other transition metal arsenate/hydroxide co-precipitates if the pH becomes neutral. For this reason, Turpeinen *et al.* (2003) cautioned about the use of acid extract data in bioavailability assessments. Using a luminescent bacterial sensor, the authors had shown that the acid soluble fraction may be an over-estimator of bioavailability. Precipitation and time-dependent aging of the precipitates have been identified as important processes to be considered. In the present study, secondary precipitation of dissolved arsenates may have been responsible for the relatively large differences between the acid and the (neutralized) gastric fluid extracts in sediments from locations 3 and 4.

Based on 23 determinations, the overall mean value of the relative bioaccessibility of sediments by the gastric fluid method was calculated to be 17.0 % with a standard error of the mean being equal to ± 2.5%. Applying the central limit theorem for this sample size (Montgomery *et. al*., 2001), the mean is expected to be t-distributed with a 95% confidence interval ranging from 11.8 % to 22.2 %. The average value found in this study is not unlike the 20% (± 11%) average bioaccessibility reported by Ollson *et al.* (2003) in organic mine soils or the 12.6% to 28.4% range observed by the MOE (2001) using the gastric fluid extraction method. The overall mean value of the 10 calculable acid extracts was 26.8% ± 5.3% (s.d). This corresponds to a 95% confidence interval of 14.8 % to 38.8 %. Obviously, the overlap between the two confidence intervals is substantial. It is concluded therefore that the acid extract data may be employed as a conservative estimate of the bioaccessibility in probabilistic assessments if gastric fluid extract data are not available.

C3.0 REFERENCES

Battelle 1996. *Determination of the Bioavailability of Arsenic in Slag Following Oral Administration in Microswine.* Battelle Laboratories, Columbus, OH. (referenced by Roberts *et al.)*

Borch, R. S., L. L. Hastings, T. N. Tingle, and K. L. Verosub 1994. *Speciation and in Vitro Gastrointestinal Extractability of Arsenic in Two California Gold Mine Tailings*, Eos, 75(16), 190.

Canivet, V., P. Chambon, and J. Gibert 2000. *Toxicity and Bioaccumulation of Arsenic and Chromium in Epigean and Hypogean Freshwater Macroinvertebrates*. Arch. Environ. Contam. Toxicol. 40, 345-354. October.

Carbonell, A.A., M.A. Aarabi, R.D. DeLaune, R.P. Gambrell and W.H. Patrick, Jr. 1998. *Bioavailability and Uptake of Arsenic by Wetland Vegetation: Effects on Plant Growth and Nutrition*. J. Environ. Sc. Health, A33(1): 45-66.

Carbonell-Barrachina, A.A., F. Burlo, D. Valero, E. Lopez, D. Martinez-Romero, and F. Martinez-Sanchez 1999. *Arsenic Toxicity and Accumulation in Turnip as Affected by Arsenic Chemical Speciation*. J. Agric, Food Chem., 47: 2288-2294.

Casteel, S.W., T. Evans, M.E. Dunsmore, C.P. Weis, B. Lavelle, W.J. Brattin, Hammon, K.M. Ellickson, R.J. Meeker, M.A. Gallo, B.T. Buckley, and P.J. Lioy 2000. *Oral Bioavailability of Lead and Arsenic from a NIST Standard Reference Soil Material*. Arch. Environ. Contam. Toxicol., 40: 128-135. June.

Casteel, S. W., L.D. Brown, M.E. Dunsmore, *et al*. 1998a. *Bioavailability of Lead in a Slag Sample from the Butte NPL Site*, Butte, Montana; Document control no. 04800-030-0165; U.S. Environmental Protection Agency: Region VIII, Denver, CO.

Casteel, S. W., L.D. Brown, M.E. Dunsmore, *et al*. 1998b. *Bioavailability of Lead in Soil and Mine Waste Form the California Gulch NPL Site*, Leadville, Colorado; Document control no. 04800-030-0178; U.S. Environmental Protection Agency: Region VIII, Denver, CO.

Casteel, S. W., L.D. Brown, M.E. Dunsmore, *et al*. 1998c. *Bioavailability of Lead in a Slag Sample from the Midvale Slag NPL Site,* Midvale, Utah; Document control no. 04800-030-0166; U.S. Environmental Protection Agency: Region VIII, Denver, CO.

Casteel, S. W., L.D. Brown, M.E. Dunsmore, *et al*. 1998d. *Bioavailability of Lead in Unweathered Galena-enriched Soils*; document control no. 04800-030-0171; U.S. Environmental Protection Agency: Region VIII, Denver, CO.

Casteel, S. W., L.D. Brown, M.E. Dunsmore, *et al*. 1998e. *Bioavailability of Lead in Paint*; document control no. 04800-030-0170; U.S. Environmental Protection Agency: Region VIII, Denver, CO.

Casteel, S. W., L.D. Brown, M.E. Dunsmore, *et al*. 1997a. *Relative Bioavailability of Arsenic in Mining Wastes*; Document control no. 4500-88-AORH; U.S. Environmental Protection Agency: Region VIII, Denver, CO.

Casteel, S. W., R. Guzman, M. Starost, C.P. Weis, *et al*. 1997b. *Bioavailability of Lead in Soil Samples from the Kennecott NPL Site*; Document control no. 04800-030-0179; Salt Lake City, UT. U.S. Environmental Protection Agency, Region VIII, Denver, CO, 1997.

Casteel, S. W., L.D. Brown, M.E. Dunsmore, *et al*. 1996a. *Bioavailability of Lead in Slag and Soil Samples from the Murray Smelter Superfund Site*; Document control no. 04800-030-0163; U.S. Environmental Protection Agency: Region VIII, Denver, CO.

Casteel, S. W., L.D. Brown, M.E. Dunsmore, *et al*. 1996b. *Bioavailability of Lead in Soil Samples from the New Jersey Zinc NPL Site*, Palmerton, Pennsylvania; Document control no. 04800-030-0159; U.S. Environmental Protection Agency: Region VIII, Denver, CO.

Casteel, S. W., L.D. Brown, M.E. Dunsmore, *et al.* 1996c. *Bioavailability of Lead in Soil Samples from the Jasper County, Missouri, Superfund Site*; Document control no. 04800-030-0161; U.S. Environmental Protection Agency: Region VIII, Denver, CO.

Casteel, S. W., L.D. Brown, M.E. Dunsmore, *et al*. 1996d. *Bioavailability of Lead in Soil Samples from the Smuggler Mountain NPL Site, Aspen, Colorado*; Document control no. 04800-030-0160; U.S. Environmental Protection Agency: Region VIII, Denver, CO.

Davis, A., Ruby, M.V., Bloom, M., Schoof, R., Freeman, G., and Bergstrom, P.D. 1996. *Minerologic Constraints on the Bioavailability of Arsenic in Smelter-Contaminated Soils*. Env. Sci. Technol. 30: 392-399.

Dillon Consulting Ltd. (2004). *Biological Sampling at Baker Creek 2003*. Technical Report to Indian and Northern Affairs Canada. 30 pp.

Dresler, J., Fisher, N., Hennigsen, G., Lanno, R., McGeer, J., Sappington, K. 2003. *Issue Paper The Bioavalability and Bioaccumulation of Metals (Draft).* U.S. Environmental Protection Agency, Risk Assessment Forum, Contract #68-C-98-148, Washington DC., 114 pp.

Ellickson, K.M., R.J. Meeker, M.A. Gallo *et al.* 2001. *Oral Bioavailability of Lead and Arsenic from a NIST Standard Reference Soil Material*. Arch. Environ. Contam. Toxicol., 40:128-135.

The Federal Insecticide, Fungicide, and Rodenticide Act, Scientific Advisory Panel (FIFRA-SAP) 2001. *Review of Dietary Exposure Evaluation Model (DEEM).* EPA Science Advisory Panel Report No. 2001-01c. Internet.

Freeman, G.B.; J.D. Johnson, J.M. Killinger, S.C. Liao, A.O. Davis, M.V. Ruby, R.L. Chaney, S.C. Lovre, and P.D. Bergstrom 1993. Fund. Appl. Toxicol., 21: 83-88.

Freeman, G.B., R.A. Schoof, M.V. Ruby, A.O. Davis, J.A. Dill, S.C. Liao, C.A. Lapin, and P.D. Bergstrom 1995. *Bioavailability of Arsenic in Soil and House Dust Impacted by Smelter Activities Following Oral Administration in Cynomolgus Monkeys*. Fundamental and Applied Toxicology, 28: 215-222. June.

Groen, K.H. Vaessen, J.J.G. Kliest, J.L.M. deBoer, T.V. Ooik, A. Timmerman and R.F. Vlug 1994. Environ. Health Perspect. 102(2): 182-184.

Hamel, S.C., Buckley, B., and Lioy, P.J. 1999. *Bioaccessibility of Metals for Different Liquid to Solid Ratios*. Environ. Sci. Technol. 32: 358-362.

International Programme for Chemical Safety (IPCS) 2000. *Short list definitions. Harmonization Project (Terminology)*. Exposure Assessment Planning Workgroup. (http://www.ipcsharmonize.org/).

Kapuska, L.A., Clements, W.H., Ziccardi, L., Paquin, P.R., Sprenger, M., Wall, D. 2003. *Issue Paper on the Ecological Effects of Metals (Draft).* U.S. Environmental Protection Agency, Risk Assessment Forum, Contract #68-C-98-148, Washington DC. 59 pp.

Langmuir, D., Chrostowski, P., Chaney, R., and Vigneault, B. 2003. *Issue Paper on the Environmental Chemistry of Metals (Draft)*. U.S. Environmental Protection Agency, Risk Assessment Forum, Contract #68-C-98-148, Washington DC. 104 pp.

Legendre, P. and Legendre, L. 1998. *Numerical Ecology, 2nd.* Ed., Elsevier Science BV, Amsterdam, 853 pp.

Lorenzana, R.M., B. Duncan, M. Ketterer, J. Lowry, J. Simon, M. Dawson, and R. Poppenga 1996. *Bioavailability of Arsenic and Lead in Environmental Substrates*. EPA 910/R-96-002. February.

Montgomery, C., Runger, G.C. and Hubele, N.F. 2001. Engineering Statistics. John Wiley and Sons. New York, NY

National Environmental Policy Institute (NEPI) 2000. *Assessing the Bioavailability of Metals in Soil for Use in Human Health Risk Assessments*. Bioavailability Policy Report Phase II: Metals Task Force Report. Summer.

Ng, J.C. and M.R. Moore 1996. *Bioavailability of Arsenic in Soils from Contaminated Sites Using a 96 Hour Rat Blood Model*. The Third National Workshop on the Health Risk Assessment and Management of Contaminated Sites.

Ng, J.C., S.M. Kratzmann, L. Qi, H. Crawley, B. Chiswell and N.R. Moore 1998. *Speciation and Absolute Bioavailability: Risk Assessment of Arsenic-Contaminated Sites in a Residential Suburb in Canberra*. The Analyst, Vol. 123, (889-892). May.

Ollson C A, K J Reimer, I Kock, and W R Cullen, 2003. *Contaminant Bioavailability and its Consequences for Ecological and Human Health Risk Assessment Case Study: Arsenic in Yellowknife, NWT, in Site Characterization I – Arctic – Session 3 (ARCSACC 2003)*, pp 129 – 138.

Ontario Ministry of the Environment (MOE) 2001. *Soil Investigation and Human Health Risk Assessment Report* for the Rodney St. Community, Port Colborne. October.

Oomen, A.G., Hack, A., Minekus, H., Zeijdner, E., Cornelis, C., Schoeters, G., Verstraete, W., Van de Wiele, T., Wragg, I., Rompelberg, C.I.M., Sips, A.J.A.M., and van Wijnen, J.H. 2002. *Comparison of Five in Vitro Digestion Models to Study the Bioaccesibility of Soil Contaminants*. Environ. Sci. Technol. 36: 3326-3334.

Roberts, S.M., W.R. Weimar, J.R.T. Vinson, J.W. Munson, and R.J. Bergeron 2001. *Measurement of Arsenic Bioavailiability in Soil Using a Primate Model.* Departments of Physiological Sciences and Medicinal Chemistry, J. Hillis Miller Health Science Center, University of Florida, Gainesville, Florida 32610.

Rodriguez, R.R. and N.T. Basta 1999. *An in Vitro Gastrointestinal Meted to Estimate Bioavailable Arsenic in Contaminated Soils and Solid Media.* Department of Plant and Soil Sciences, Oklahoma State University, Stillwater, Oklahoma 74078. Environ. Sci. Technol. 33: 643-649.

Ruby M.V., R. Schoof, W. Brattin, M. Goldade, G. Post, M. Harnois, D.E. Mosby, S.W. Casteel, W. Berti, M. Carpenter, D. Edwards, D. Cragin and W. Chappell 1999. *Advances in Evaluating the Oral Bioavailability of Inorganics in Soil for Use in Human Health Risk Assessment*. Environ. Sci. Technol., 33(21):3697–3705.

Tessier A., G.G.C. Campbell and M. Bisson 1979. *Sequential Extraction Procedure for the Speciation of Particulate Trace Metals.* Analytical Chemistry 51(7): 844-851.

Turpeinen, R., Virta, M., and Haggblom, M. M. 2003. *Analysis of Arsenic Bioavailability in Contaminated Soils.* Environ. Toxicol. Chem. 22:1-6.

United States Environmental Protection Agency (U.S. EPA) 2001. *Baseline Human Health Risk Assessment Vasquez Boulevard and I-70 Superfund Site Denver, CO.* August.

Williams, T.M., B.G. Rawlings, B. Smith and N. Breward 1998. *In-Vitro Determination of Arsenic Bioavailability in Contaminated Soil and Mineral Beneficiation Waste from Ron Phibun, Southern Thailand: A Basis for Improved Human Risk Assessment*. Environmental Geochemistry and Health (1998), 20: 169-177.

ATTACHMENT 1
ANALYTICAL RESULTS OF THE SEQUENTIAL EXTRACTION

				Aluminum Extraction (microgram/g))							
Extract #	Sample 1	Sample 2	Sample 3	Sample 4	Sample 5	Sample 6	Sample 7	Sample 8	Sample 9	Sample 10	Sample 11
1	5	7	6	10	2	< 2	< 2	6	7	< 2	3
2	26	15	13	16	11	13	16	20	22	19	9
3	425	292	159	447	35	193	380	499	478	561	464
4	2019	1650	1335	1278	1810	1246	679	2249	2085	2124	2061
5	5980	5540	5630	13800	12700	12200	17700	7060	6120	13800	11900
TOTAL	8455	7504	7143	15551	14558	13652	18775	9834	8712	16504	14437
				Aluminum Extraction (%)							
Extract #	Sample 1	Sample 2	Sample 3	Sample 4	Sample 5	Sample 6	Sample 7	Sample 8	Sample 9	Sample 10	Sample 11
1	0.1	0.1	0.1	0.1	0.0	ND	ND	0.1	0.1	ND	0.0
2	0.3	0.2	0.2	0.1	0.1	0.1	0.1	0.2	0.3	0.1	0.1
3	5.0	3.9	2.2	2.9	0.2	1.4	2.0	5.1	5.5	3.4	3.2
4	23.9	22.0	18.7	8.2	12.4	9.1	3.6	22.9	23.9	12.9	14.3
5	70.7	73.8	78.8	88.7	87.2	89.4	94.3	71.8	70.2	83.6	82.4
TOTAL	100.0	100.0	100.0	100.0	100.0	100.0	100.0	100.0	100.0	100.0	100.0
				Barium Extraction (microgram/g)							
Extract #	Sample 1	Sample 2	Sample 3	Sample 4	Sample 5	Sample 6	Sample 7	Sample 8	Sample 9	Sample 10	Sample 11
1	9.9	22	3.9	6.7	19	7.0	2.5	11	11	18	19
2	4.8	11	1.9	4.2	8.0	5.2	4.0	4.8	4.8	9.3	8.5
3	10	17	3.5	10	3.4	5.5	5.2	11	9.5	15	11
4	7.8	19	12	6.5	19	7.7	2.5	8.5	7.9	13	12
5	937	36	44	478	110	352	1530	1110	952	75	71
TOTAL	970	105	65	505	159	377	1544	1145	985	130	122
				Barium Extraction (%)							
Extract #	Sample 1	Sample 2	Sample 3	Sample 4	Sample 5	Sample 6	Sample 7	Sample 8	Sample 9	Sample 10	Sample 11
1	1.0	21.0	6.0	1.3	11.9	1.9	0.2	1.0	1.1	13.8	15.6
2	0.5	10.5	2.9	0.8	5.0	1.4	0.3	0.4	0.5	7.1	7.0
3	1.0	16.2	5.4	2.0	2.1	1.5	0.3	1.0	1.0	11.5	9.1
4	0.8	18.1	18.4	1.3	11.9	2.0	0.2	0.7	0.8	10.0	9.9
5	96.6	34.3	67.4	94.6	69.0	93.3	99.1	96.9	96.6	57.6	58.4
TOTAL	100.0	100.0	100.0	100.0	100.0	100.0	100.0	100.0	100.0	100.0	100.0

ATTACHMENT 1 (Cont'd)
ANALYTICAL RESULTS OF THE SEQUENTIAL EXTRACTION

					Cadmium Extraction (microgram/g)						
Extract #	Sample 1	Sample 2	Sample 3	Sample 4	Sample 5	Sample 6	Sample 7	Sample 8	Sample 9	Sample 10	Sample 11
1	< 0.9	< 0.9	< 0.9	< 0.9	< 0.9	< 0.9	< 0.9	< 0.9	< 0.9	< 0.9	< 0.9
2	1.0	< 0.9	< 0.9	< 0.9	< 0.9	< 0.9	< 0.9	1.0	1.1	< 0.9	< 0.9
3	10	< 0.9	3.5	16	< 0.9	< 0.9	2.5	8.9	7.4	< 0.9	< 0.9
4	18	1.1	1.4	22	1.9	7.6	18	19	18	1.8	1.8
5	7.1	< 0.9	< 0.9	3.8	< 0.9	2.5	11	7.9	7.0	< 0.9	< 0.9
Extract #	36	1.1	4.9	41.8	1.9	10.1	31.5	36.8	33.5	1.8	1.8
					Cadmium Extraction (%)						
Extract #	Sample 1	Sample 2	Sample 3	Sample 4	Sample 5	Sample 6	Sample 7	Sample 8	Sample 9	Sample 10	Sample 11
1	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
2	2.8	ND	ND	ND	ND	ND	ND	2.7	3.3	ND	ND
3	27.7	ND	71.4	38.3	ND	ND	7.9	24.2	22.1	ND	ND
4	49.9	100.0	28.6	52.6	100.0	75.2	57.1	51.6	53.7	100.0	100.0
5	19.7	ND	ND	9.1	ND	24.8	34.9	21.5	20.9	ND	ND
TOTAL	100.0	100.0	100.0	100.0	100.0	100.0	100.0	100.0	100.0	100.0	100.0
					Cobalt Extraction (microgram/g)						
Extract #	Sample 1	Sample 2	Sample 3	Sample 4	Sample 5	Sample 6	Sample 7	Sample 8	Sample 9	Sample 10	Sample 11
1	< 3	< 3	< 3	< 3	< 3	< 3	< 3	< 3	< 3	< 3	< 3
2	< 3	< 3	< 3	< 3	< 3	4	6	< 3	< 3	< 3	< 3
3	< 3	< 3	< 3	11	< 3	5	27	< 3	< 3	8	7
4	< 3	< 3	11	12	11	22	10	< 3	3	5	5
5	4	3	4	9	8	10	16	4	4	7	7
TOTAL	4	3	15	32	19	41	59	4	7	20	19
					Cobalt Extraction (%)						
Extract #	Sample 1	Sample 2	Sample 3	Sample 4	Sample 5	Sample 6	Sample 7	Sample 8	Sample 9	Sample 10	Sample 11
1	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
2	ND	ND	ND	ND	ND	9.8	10.2	ND	ND	ND	ND
3	ND	ND	ND	34.4	ND	12.2	45.8	ND	ND	40.0	36.8
4	ND	ND	73.3	37.5	57.9	53.7	16.9	ND	42.9	25.0	26.3
5	100.0	100.0	26.7	28.1	42.1	24.4	27.1	100.0	57.1	35.0	36.8
TOTAL	100.0	100.0	100.0	100.0	100.0	100.0	100.0	100.0	100.0	100.0	100.0

ATTACHMENT 1 (Cont'd)
ANALYTICAL RESULTS OF THE SEQUENTIAL EXTRACTION

				Chromium Extraction (microgram/g)							
Extract #	Sample 1	Sample 2	Sample 3	Sample 4	Sample 5	Sample 6	Sample 7	Sample 8	Sample 9	Sample 10	Sample 11
1	< 1	< 1	< 1	< 1	< 1	< 1	< 1	< 1	< 1	< 1	< 1
2	< 1	< 1	< 1	< 1	< 1	< 1	< 1	< 1	< 1	< 1	< 1
3	1	1	< 1	2	< 1	2	2	1	1	3	3
4	6	5	4	3	5	3	1	7	6	5	5
5	18	19	17	34	31	34	41	19	18	29	29
TOTAL	25	25	21	39	36	39	44	27	25	37	37
				Chromium Extraction (%)							
Extract #	Sample 1	Sample 2	Sample 3	Sample 4	Sample 5	Sample 6	Sample 7	Sample 8	Sample 9	Sample 10	Sample 11
1	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
2	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
3	4.0	4.0	ND	5.1	ND	5.1	4.5	3.7	4.0	8.1	8.1
4	24.0	20.0	19.0	7.7	13.9	7.7	2.3	25.9	24.0	13.5	13.5
5	72.0	76.0	81.0	87.2	86.1	87.2	93.2	70.4	72.0	78.4	78.4
TOTAL	100.0	100.0	100.0	100.0	100.0	100.0	100.0	100.0	100.0	100.0	100.0
				Copper Extraction (microgram/g)							
Extract #	Sample 1	Sample 2	Sample 3	Sample 4	Sample 5	Sample 6	Sample 7	Sample 8	Sample 9	Sample 10	Sample 11
1	0.8	< 0.7	2.6	< 0.7	< 0.7	< 0.7	1.8	1.3	0.8	< 0.7	< 0.7
2	1.7	< 0.7	5.8	0.7	< 0.7	4.0	20	2.1	2.4	0.7	< 0.7
3	4.1	< 0.7	9.1	9.9	< 0.7	9.0	44	6.0	8.2	5.9	5.3
4	180	7.6	860	93	49	121	274	197	176	48	42
5	18	3.2	43	17	12	20	41	19	16	16	13
TOTAL	204.6	10.8	920.5	120.6	61.0	154.0	380.8	225.4	203.4	70.6	60.3
				Copper Extraction (%)							
Extract #	Sample 1	Sample 2	Sample 3	Sample 4	Sample 5	Sample 6	Sample 7	Sample 8	Sample 9	Sample 10	Sample 11
1	0.4	ND	0.3	ND	ND	ND	0.5	0.6	0.4	ND	ND
2	0.8	ND	0.6	0.6	ND	2.6	5.3	0.9	1.2	1.0	ND
3	2.0	ND	1.0	8.2	ND	5.8	11.6	2.7	4.0	8.4	8.8
4	88.0	70.4	93.4	77.1	80.3	78.6	72.0	87.4	86.5	68.0	69.7
5	8.8	29.6	4.7	14.1	19.7	13.0	10.8	8.4	7.9	22.7	21.6
TOTAL	100.0	100.0	100.0	100.0	100.0	100.0	100.0	100.0	100.0	100.0	100.0

ATTACHMENT 1 (Cont'd)
ANALYTICAL RESULTS OF THE SEQUENTIAL EXTRACTION

				Iron Extraction (microgram/g)							
Extract #	Sample 1	Sample 2	Sample 3	Sample 4	Sample 5	Sample 6	Sample 7	Sample 8	Sample 9	Sample 10	Sample 11
1	12	10	4	8	10	< 1	2	11	13	8	6
2	31	13	17	44	48	107	89	27	31	55	41
3	1900	1690	640	3950	510	3150	8750	2140	1860	4190	3900
4	2230	3330	2350	6790	4430	4480	5630	2580	2480	2570	2720
5	9000	8100	8800	27200	19400	23600	41400	10600	9200	20900	18300
TOTAL	13173	13143	11811	37992	24398	31337	55871	15358	13584	27723	24967
				Iron Extraction (%)							
Extract #	Sample 1	Sample 2	Sample 3	Sample 4	Sample 5	Sample 6	Sample 7	Sample 8	Sample 9	Sample 10	Sample 11
1	0.1	0.1	0.0	0.0	0.0	ND	0.0	0.1	0.1	0.0	0.0
2	0.2	0.1	0.1	0.1	0.2	0.3	0.2	0.2	0.2	0.2	0.2
3	14.4	12.9	5.4	10.4	2.1	10.1	15.7	13.9	13.7	15.1	15.6
4	16.9	25.3	19.9	17.9	18.2	14.3	10.1	16.8	18.3	9.3	10.9
5	68.3	61.6	74.5	71.6	79.5	75.3	74.1	69.0	67.7	75.4	73.3
TOTAL	100.0	100.0	100.0	100.0	100.0	100.0	100.0	100.0	100.0	100.0	100.0
					Manganese Extraction (microgram/g)						
Extract #	Sample 1	Sample 2	Sample 3	Sample 4	Sample 5	Sample 6	Sample 7	Sample 8	Sample 9	Sample 10	Sample 11
1	108	243	31.8	163	41.7	138	70.4	125	115	42.6	44.7
2	14	30	7.2	37	10	78	94	13	11	11	10
3	75	274	22	195	33	108	319	83	75	48	40
4	30	75	19	44	28	34	55	33	30	29	26
5	73	67	67	147	154	159	184	76	71	143	141
TOTAL	300	689	147	586	267	517	722	330	302	274	262
					Manganese Extraction (%)						
Extract #	Sample 1	Sample 2	Sample 3	Sample 4	Sample 5	Sample 6	Sample 7	Sample 8	Sample 9	Sample 10	Sample 11
1	36.0	35.3	21.6	27.8	15.6	26.7	9.7	37.9	38.1	15.6	17.1
2	4.7	4.4	4.9	6.3	3.7	15.1	13.0	3.9	3.6	4.0	3.8
3	25.0	39.8	15.0	33.3	12.4	20.9	44.2	25.2	24.8	17.5	15.3
4	10.0	10.9	12.9	7.5	10.5	6.6	7.6	10.0	9.9	10.6	9.9
5	24.3	9.7	45.6	25.1	57.7	30.8	25.5	23.0	23.5	52.3	53.9
TOTAL	100.0	100.0	100.0	100.0	100.0	100.0	100.0	100.0	100.0	100.0	100.0

ATTACHMENT 1 (Cont'd)
ANALYTICAL RESULTS OF THE SEQUENTIAL EXTRACTION

				Strontium Extraction (microgram/g)							
Extract #	Sample 1	Sample 2	Sample 3	Sample 4	Sample 5	Sample 6	Sample 7	Sample 8	Sample 9	Sample 10	Sample 11
1	5.22	5.36	19.7	14.7	15.7	13.4	11.2	6.74	5.57	16.2	< 5
2	0.60	0.59	1.7	1.7	1.5	2.6	3.1	0.55	0.52	1.6	1.7
3	0.95	0.72	1.9	5.0	1.6	2.6	6.3	1.0	0.92	2.2	1.9
4	1.4	1.3	2.1	2.1	2.3	2.1	2.0	1.4	1.3	2.2	2.0
5	3.8	2.5	3.3	3.1	9.2	6.0	2.0	4.0	3.7	9.2	8.5
TOTAL	11.97	10.47	28.70	26.60	30.30	26.70	24.60	13.69	12.01	31.40	14.10
				Strontium Extraction (%)							
Extract #	Sample 1	Sample 2	Sample 3	Sample 4	Sample 5	Sample 6	Sample 7	Sample 8	Sample 9	Sample 10	Sample 11
1	43.6	51.2	68.6	55.3	51.8	50.2	45.5	49.2	46.4	51.6	ND
2	5.0	5.6	5.9	6.4	5.0	9.7	12.6	4.0	4.3	5.1	12.1
3	7.9	6.9	6.6	18.8	5.3	9.7	25.6	7.3	7.7	7.0	13.5
4	11.7	12.4	7.3	7.9	7.6	7.9	8.1	10.2	10.8	7.0	14.2
5	31.7	23.9	11.5	11.7	30.4	22.5	8.1	29.2	30.8	29.3	60.3
TOTAL	100.0	100.0	100.0	100.0	100.0	100.0	100.0	100.0	100.0	100.0	100.0
				Titanium Extraction (microgram/g)							
Extract #	Sample 1	Sample 2	Sample 3	Sample 4	Sample 5	Sample 6	Sample 7	Sample 8	Sample 9	Sample 10	Sample 11
1	< 0.2	< 0.2	< 0.2	< 0.2	< 0.2	< 0.2	< 0.2	< 0.2	< 0.2	< 0.2	< 0.2
2	0.6	< 0.2	< 0.2	0.2	< 0.2	< 0.2	< 0.2	0.3	0.4	0.3	< 0.2
3	1.0	1.0	1.3	0.9	< 0.2	0.5	< 0.2	0.9	1.1	2.3	1.9
4	100	90	58	28	23	14	8.2	119	107	98	96
5	260	238	231	297	566	457	185	262	253	511	488
TOTAL	361.6	329	290.3	326.1	589	471.5	193.2	382.2	361.5	611.6	585.9
				Titanium Extraction (%)							
Extract #	Sample 1	Sample 2	Sample 3	Sample 4	Sample 5	Sample 6	Sample 7	Sample 8	Sample 9	Sample 10	Sample 11
1	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
2	0.2	ND	ND	0.1	ND	ND	ND	0.1	0.1	0.0	ND
3	0.3	0.3	0.4	0.3	ND	0.1	ND	0.2	0.3	0.4	0.3
4	27.7	27.4	20.0	8.6	3.9	3.0	4.2	31.1	29.6	16.0	16.4
5	71.9	72.3	79.6	91.1	96.1	96.9	95.8	68.6	70.0	83.6	83.3
TOTAL	100.0	100.0	100.0	100.0	100.0	100.0	100.0	100.0	100.0	100.0	100.0

ATTACHMENT 1 (Cont'd)
ANALYTICAL RESULTS OF THE SEQUENTIAL EXTRACTION

				Vanadium Extraction (microgram/g)							
Extract #	Sample 1	Sample 2	Sample 3	Sample 4	Sample 5	Sample 6	Sample 7	Sample 8	Sample 9	Sample 10	Sample 11
1	< 0.2	< 0.2	< 0.2	< 0.2	< 0.2	< 0.2	< 0.2	< 0.2	< 0.2	< 0.2	< 0.2
2	< 0.7	< 0.7	< 0.7	< 0.7	< 0.7	< 0.7	< 0.7	< 0.7	< 0.7	< 0.7	< 0.7
3	1.5	1.3	0.8	2.4	< 0.7	2.4	3.2	1.5	1.5	6.8	6.3
4	4.1	4.0	2.7	2.9	9.0	3.8	2.1	4.9	4.3	5.8	5.0
5	13	11	15	38	31	36	54	13	13	28	27
TOTAL	18.6	16.3	18.5	43.3	40	42.2	59.3	19.4	18.8	40.6	38.3
				Vanadium Extraction (%)							
Extract #	Sample 1	Sample 2	Sample 3	Sample 4	Sample 5	Sample 6	Sample 7	Sample 8	Sample 9	Sample 10	Sample 11
1	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
2	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
3	8.1	8.0	4.3	5.5	ND	5.7	5.4	7.7	8.0	16.7	16.4
4	22.0	24.5	14.6	6.7	22.5	9.0	3.5	25.3	22.9	14.3	13.1
5	69.9	67.5	81.1	87.8	77.5	85.3	91.1	67.0	69.1	69.0	70.5
TOTAL	100.0	100.0	100.0	100.0	100.0	100.0	100.0	100.0	100.0	100.0	100.0
				Uranium Extraction (microgram/g)							
Extract #	Sample 1	Sample 2	Sample 3	Sample 4	Sample 5	Sample 6	Sample 7	Sample 8	Sample 9	Sample 10	Sample 11
1	< 10	< 10	< 10	< 10	< 10	< 10	< 10	< 10	< 10	< 10	< 10
2	< 10	< 10	< 10	< 10	< 10	< 10	< 10	< 10	< 10	< 10	< 10
3	33	40	42	27	25	26	27	25	25	19	30
4	< 10	< 10	< 10	< 10	< 10	< 10	< 10	< 10	< 10	< 10	< 10
5	< 10	< 10	< 10	< 10	< 10	< 10	< 10	< 10	< 10	< 10	< 10
TOTAL	33	40	42	27	25	26	27	25	25	19	30
				Uranium Extraction (%)							
Extract #	Sample 1	Sample 2	Sample 3	Sample 4	Sample 5	Sample 6	Sample 7	Sample 8	Sample 9	Sample 10	Sample 11
1	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
2	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
3	100	100	100	100	100	100	100	100	100	100	100
4	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
5	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
TOTAL	100.0	100.0	100.0	100.0	100.0	100.0	100.0	100.0	100.0	100.0	100.0

APPENDIX D

PATHWAYS AND RISK ASSESSMENT MODEL DESCRIPTION

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APPENDIX D – PATHWAYS AND RISK ASSESSMENT MODEL DESCRIPTION

D1.0 INTRODUCTION

The Canadian Council of Ministers of the Environment (CCME) has provided general guidance concerning their views on what constitutes an ecological risk assessment, ERA (CCME 1996, 1997). The framework provided is similar to that proposed by Environment Canada and the U.S. EPA (Environment Canada 1997, U.S. EPA 1992). The CCME proposes three levels of investigation:

1) Screening level assessment (SLA): essentially a qualitative assessment of potential risks to important ecological receptors.
2) Preliminary quantitative risk assessment (PQRA): focuses on filling gaps identified at the screening level.
3) Detailed quantitative risk assessment (DQRA): includes more detailed data and modelling.

As discussed in the main document, a screening level assessment has already been carried out for the assessment of arsenic trioxide management alternatives. This current assessment is a Tier 2 or preliminary quantitative risk assessment to determine if arsenic loads to the aquatic environment would result in potential impacts to ecological species or humans.

Ecological systems are inherently dynamic and complex. As such, complete characterization of ecological interactions among individual organisms, populations, and communities lies beyond current capabilities in ecology. Quantifying the environmental transport, distribution, and accumulation of chemical contaminants is equally complicated. As a result of these kinds of uncertainties, the assessment is often undertaken within a probabilistic framework. This study models environmental transport and pathways in a probabilistic framework (Monte Carlo) and compares the results to given toxicological benchmarks.

As with any predictive assessment, probabilistic analysis involves a mathematical expression (model) of some physical system. The model is a mathematical expression to which information is supplied in the form of numerical values for one or more input variables and it produces the corresponding numerical values of one or more output variables. Most the input variables required by such a model are not known with certainty. The uncertainty arises because of normal experimental error in their measurement or because (sometimes) actual measurements are impossible and judgement must be used. Also, some input variable vary naturally from year to year. Consequently, it is impossible for most input variables to be adequately characterized by

any single number. To account for these uncertainties in the input variables (subjective), probability distributions can be assigned to the variable.

Having specified the input distributions, it is necessary to determine the corresponding values of the output variables. Because many of the input parameters are specified as distributions, rather than as single numbers, distributions, rather than single numbers will be obtained for the calculated variables. The distribution of calculated results is obtained by using a statistical sampling or Monte Carlo procedure which involves drawing many samples from each input parameter distribution. The procedure is a repetitive one. At each repetition, a single value is drawn for each input variable from its probability distribution. These values of the input variables are substituted into the mathematical model to obtain a single value for each output variable. This process, which is referred to as a trial, is then repeated as often as desired.

The output variable(s) from such an analysis can be considered as a sample(s) on an objective random variable(s) corresponding to that output variable(s). From the sample, the mean, variance, probability density function (pdf) or cumulative distribution function (cdf) may be estimated as desired. This information may be interpreted as describing the output variable in terms of subjective probability; telling us quantitatively what we are entitled to believe about the output variable. In the current assessment, the summary statistics most often referred to are the mean and 95^{th} percentile values.

D2.0 ECOLOGICAL CHARACTERIZATION

In general, the aquatic flora and fauna in the Yellowknife study area are typical of aquatic systems found in the Northwest Territories.

Great Slave Lake supports a number of fish species: northern pike, lake whitefish, white sucker to name a few. Baker Creek provides an excellent spawning habitat for white sucker; however, the shallowness of this creek may also be a limiting factor for the resident fish populations such as lake whitefish, northern pike and walleye.

The study area supports a good number of benthic invertebrates, although general productivity is considered to be low due to the arsenic contamination in both Baker Creek and Back Bay. Rawson (1953) found *Pontoporeia* to be the most abundant benthic group in Great Slave Lake including Yellowknife Bay. However, currently there is an absence of this species in the Back Bay area (Sutherland 1989).

There are large numbers of algae found in the water in Baker Creek and Yellowknife Bay. Additionally there are numerous zooplankton and phytoplankton species in Yellowknife Bay. Some of the predominant species of zooplankton are *Keratella cochlearis, Kellicottia longispera, Synchaeta stylata* and *Diaptomus ashlandi* to name a few (Moore *et al*. 1978). Predominant species of phytoplankton are *Asterionella formosa, Diatoma tenue, Melosira islandica, Dinobryon* and *Scensedesmus quadricauda* (Moore *et al*. 1978*)*.

Spruce grouse have been found in the area. Mallard and scaup are the most commonly observed waterfowl species (on-line website).

Small mammals are important food sources for birds, mammals and reptiles. A number of species such as red-backed vole, meadow vole, deer mouse, masked shrew and meadow jumping mouse have been found in the area (Hough *et al*. 2000). Furbearing mammals have also been found and include, muskrat, mink, red fox, red squirrel, snowshoe hare, and wolf (Hough *et al*. 2000). Moose, woodland caribou and deer have also been observed (Hough *et al*. 2000).

D3.0 ECOSYSTEMS

D3.1 GENERAL STRUCTURE AND FUNCTIONING

D3.1.1 Terrestrial Ecosystem

The soil is the foundation of the terrestrial ecosystem. The development of geological parent material into soil involves a complex variety of physical, biochemical and geochemical processes, including weathering, leaching, ammonification, nitrification, denitrification, respiration, nitrogen fixation, and decomposition (Larsen 1980). Once formed, soil acts as both a repository and a source of mineral nutrients, carbon compounds, and nitrogen compounds that form the major nutrient cycles in the ecosystem.

Atmospheric inputs of nutrients to terrestrial systems include oxygen, nitrogen, water, carbon dioxide, and air-borne particles that may contain minerals or other nutrient compounds. Lichens subsist almost entirely on air-borne nutrients, and rely on water vapour for moisture, taking little of either from the substrates on which they grow. Most other organisms cycle nutrients within the terrestrial system, but rely on atmospheric carbon dioxide for photosynthesis (green plants) and/or oxygen for respiration. The direct effect of climate (meteorological effect) is primarily on energy flow, through input of solar radiation and wind. However, climate also plays a major role in soil development and in the composition, abundance, and distribution of plant and animal species. Nutrients may be lost to the atmosphere through fire (in the form of smoke and ash) and as carbon dioxide and other gases produced in chemical cycles.

Nutrients input from biological sources originate from aquatic systems, such as through:

- the emergence of adult aquatic insects and amphibians after having spent egg and juvenile stages in the water;
- feeding (e.g. by herbivores such as beaver, muskrat, and moose that feed on submergent and emergent aquatic macrophytes, omnivores such as waterfowl that feed on benthic invertebrates and aquatic vegetation, and carnivores such as the bald eagle, osprey, and otter that capture and consume fish); or,
- the drinking of free water.

The amount of nutrient input from aquatic systems has not been quantified, but ranges from high along lake and river shores, to low in areas geographically or topographically removed from surface-water sources. Nutrients may also be lost to aquatic systems through litter fall into rivers and lakes, egg deposition by insects and amphibians, insects consumed by fish, tree and shrub cuttings taken to the water by beavers, and animal faecal material.

An important consideration in modelling uptake and transport of materials by birds is that many are migratory, and are only in the area for 4 to 5 months each year. The spruce grouse has been assumed to be a year-round resident.

D3.1.2 Aquatic Ecosystem

A lake ecosystem might be considered, to a large degree, a closed or self-contained ecosystem. However, nutrients and other substances move across the boundaries of the lake ecosystem along biological, geological, and meteorological pathways (Likens and Bormann 1972).

Exchanges between aquatic and surrounding ecosystems are usually multifaceted. Biological inputs include (1) fish moving into a lake, (2) deposition of eggs by terrestrial insects with aquatic larval stages, (3) tree branches from beaver cutting activity, and (4) waste material from terrestrial plants and animals. Biological outputs are composed of (1) fish moving out of a lake, (2) the emergence of aquatic insects, (3) predation by insect and fish-eating birds and mammals, and (4) grazing on aquatic macrophytes by birds and ungulates such as moose.

Geological inputs consist predominantly of nutrients dissolved in groundwater and in-flowing streams and particulate matter washed into the basin from the surrounding terrestrial watershed. Geological outputs include dissolved and particulate matter carried out of the lake by out-flowing waters and nutrients incorporated in deep sediments which may be removed from circulation for a long period of time. Meteorological inputs include wind-borne particulate matter, dissolved substances in rain and snow, and atmospheric gases. Outputs are generally small and consists largely of gases such as carbon dioxide and methane (Smith 1980; Vanriel 1989). Energy input into a lake ecosystem is predominantly sunlight and output is in the form of heat (consisting of solar energy not utilized by primary producers and total respiration losses of all biological components of the ecosystem). It is estimated that, in general, 99% of incoming solar energy is not utilized by primary producers (Cole 1983).

Like the terrestrial ecosystem, energy stored by plants (i.e., algae and aquatic macrophytes) is passed along through the ecosystem in a series of consumption steps in the food chain. Especially at the beginning of the food chain, resources are usually shared since no one organism lives wholly on another. Thus, food chains become inter-linked to form a food web, the complexity of which varies within and between ecosystems. It has been estimated that approximately 90% of biomass energy is dissipated as low grade energy (heat) during the transformation from lower to higher trophic levels (Harvey 1950; Smith 1980; Cole 1983). This limits the length of any food chain to four or five since the longer the food chain the less energy is available for the final members. Similar to terrestrial ecosystems, the detrital/decomposer loop dominates the ecosystem.

Food chains/webs of northern lake ecosystems are relatively simple when compared to such systems in more temperate and especially tropical regions. Nevertheless, relatively simple lake ecosystems, are still extremely complex. The trophic interactions especially between lower trophic levels are not perfectly linear; multiple pathways exist between aquatic macrophytes/benthic algae, benthic bacteria, and the variety of trophic life styles encountered in a benthic invertebrate community. Although most benthic invertebrates are likely detritivores, many benthic communities also contain herbivores and predatory species (Coffman 1978; Wiggins 1978; Fry 1982).

D3.2 ROLES OF SPECIFIC ECOLOGICAL RECEPTORS

D3.2.1 Terrestrial Species

Decomposers

Decomposers include soil organisms such as bacteria, fungi, earthworms, isopods, millipedes, and beetles. This group converts dead plant and animal material to simpler compounds and basic nutrients which then enter the carbon, nitrogen, and mineral cycles. The decomposers are essential to maintaining the fertility, and hence the productivity, of the ecosystem. Due to the difficulty in modelling and assessing (monitoring) of this group no ecological receptors were selected.

Primary Producers

Forest vegetation, including both vascular and most non-vascular plants, is important in creating and modifying macro- and micro-climatic conditions, providing forage for wildlife, producing materials used by people for subsistence, and producing commercial resources. VEC plant species predominately include those that provide food or that have cultural or medicinal significance to residents. Since there is no contribution from the air pathway, no plant species were included as ecological receptors in this assessment.

Nitrogen Fixers

In northern environments, plant growth may be limited by the availability of nitrogen. Most nitrogen in the soil is contained in organic matter that cannot be assimilated by higher plants, and during each growing season, only a small proportion of that material is broken down to produce available nitrogen (Larsen 1980). To cope with this limitation, some bacteria and some higher plants have developed a means of converting atmospheric nitrogen to organic nitrogen compounds ("fixing" nitrogen). Those organisms increase the productivity of the ecosystem by increasing available nitrogen in the soil. Due to the difficulties in modelling and assessing (monitoring) of this group, no ecological receptors were selected.

Herbivores

Herbivores convert vegetable matter to animal protein, and in turn are consumed by omnivores and carnivores. They also are trapped or hunted by residents for furs or food. The herbivore layer spans a wide range of species, including microorganisms, invertebrates, and vertebrates. Vertebrate herbivores considered as ecological receptors comprise; spruce grouse, snowshoe hare, and moose. Spruce grouse, snowshoe hare, muskrat, moose and barrenground caribou have been chosen as representative species for this group.

Omnivores

Omnivores consume both plant and animal matter. Vertebrate omnivores include waterfowl, which feed on aquatic vegetation and aquatic invertebrates. Waterfowl are hunted by some residents for food. As waterfowl are expected to be the highest exposed ecological receptors these birds were chosen for the assessment. Waterfowl were subdivided into mallards (which consume a mixture of plankton from the water column and benthic invertebrates from the lake sediment), mergansers (which may consume primarily fish) and scaup (which may consume aquatic snails from lake sediment). The bear has also been considered because it consumes berries (a potential source of arsenic) and fish.

Carnivores

Predators represent the top level of the food chain. Predators interact with prey species (usually herbivores) and may influence population levels and even distribution of prey. Several are trapped for fur. Terrestrial predators considered to be ecological receptors include gray wolf and mink.

In summary, Table D3.2-1 shows the terrestrial receptors chosen for this assessment:

TABLE D3.2-1
TERRESTRIAL RECEPTORS CHOSEN FOR THE ASSESSMENT

Herbivores	Omnivores	Carnivores
Snowshoe hare	Mallard duck	Mink
Spruce grouse	Common merganser duck	Gray wolf
Muskrat	Scaup duck	
Moose	Black bear	
Barrenground caribou		

Tables D3.2-2 to D3.2-10 provide the characteristics of each of these terrestrial species.

TABLE D3.2-2
SNOWSHOE HARE CHARACTERISTICS

Parameter Description	Units	Normal Value	Distribution	Reference
Food ingestion rate for hare[a]	g(wet)/d	300	C (300)	Pease *et al.* 1979
Fraction of food from reference site	-	1.0	C (1.0)	Assumed home range within site
Fraction of food that is browse[b]	-	0.6	T (0.4, 0.6, 0.8)	U.S. EPA 1993
Fraction of food that is herbaceous[b]	-	0.38	Calculated as reminder of diet	U.S. EPA 1993
Fraction of food that is soil	-	0.02	C (0.02)	Beyer *et al*. 1994
Water ingestion rate[d]	m^3/d	1.3×10^{-4}	C ($1.3x10^{-4}$)	U.S. EPA 1993
Food-to-hare meat transfer coefficient for: Arsenic	µg/g (wet wt) µg/d	2.0×10^{-6}	LN ($2.0x10^{-6}$, 2.5, $1.3x10^{-7}$, $3.1x10^{-5}$)	(c)

Notes:

a) This value is consistent with the value obtained for an allometric equation for herbivores given in U.S. EPA (1993).
b) Based on the dietary composition of Eastern Cottontail Rabbit from U.S. EPA (1993).
c) Based on information available in IAEA (1994), U.S. NCRP 1996, Baes *et al*. (1984), U.S. EPA (1998) and CSA (1987)
d) Based on allometric equation for water intake from U.S. EPA (1993) and using the body weight of a hare of 1.4 kg from U.S. EPA (1993).

TABLE D3.2-3
SPRUCE GROUSE CHARACTERISTICS

Parameter Description	Units	Normal Value	Distribution	Reference
Food ingestion rate[a]	g(wet wt.)/d	120	C (120)	U.S. EPA 1993
Fraction of food from reference site[b]	-	1.0	C (1.0)	Assumed
Fraction of food that is browse[c]	-	0.83	Calculated as reminder of diet	U.S. EPA 1993
Fraction of food that is berries[c]	-	0.15	T (0.02, 0.15, 0.3)	U.S. EPA 1993
Fraction of food that is soil	-	0.02	C (0.02)	Beyer *et al*. 1994
Water ingestion rate[d]	m^3/d	3.6×10^{-5}	C ($3.6x10^{-5}$)	U.S. EPA 1993
Fraction of water from reference site[b]	-	1.0	C (1.0)	Assumed
Food-to-grouse flesh transfer coefficient for: Arsenic	$\frac{\mu g/g \text{ (wet wt.)}}{\mu g/d}$	1.0×10^{-3}	LN ($1.0x10^{-3}$, 5.0, $4.0x10^{-5}$, $2.5x10^{-2}$)	(e)

Notes:

a) Based on the allometric equation provided in U.S. EPA (1993) and a body weight of 475 g for a grouse (Newfoundland Government 2001).

b) Because grouse do not migrate and are likely to be resident year around, it was assumed that they would obtain all their food and water from the reference site.

c) Based on breakdown of food intake by a quail in U.S. EPA (1993).

d) Based on allometric equation provided in U.S. EPA (1993).

e) Based on information available in IAEA (1994), Baes *et al*. (1984), U.S. EPA (1998) and CSA (1987).

TABLE D3.2-4
MOOSE CHARACTERISTICS

Parameter Description	Units	Normal Value	Distribution	Reference
Food ingestion rate[a]	g(wet wt.)/d	23×10^3	C (15×10^3, 23×10^3, 30×10^3)	Canadian Wildlife Service 1997
Fraction of terrestrial vegetation consumed from reference site[b]	-	0.25	C (0.25)	Assumed
Fraction of food that is browse	-	0.9	T (0.85, 0.9, 0.95)	Belovsky *et al*. 1973
Fraction of food that is aquatic vegetation	-	0.052	Calculated as reminder of diet	Belovsky *et al*. 1973
Fraction of food that is sediment	-	0.01	C (0.01)	Beyer *et al*. 1994
Water ingestion rate[c]	m^3/d	0.031	C (0.021, 0.031, 0.041)	U.S. EPA 1993
Fraction of water and aquatic vegetation from reference site[b]	-	0.25	C (0.25)	Assumed
Food-to-moose meat transfer coefficient for: Arsenic	µg/g (wet wt.) µg/d	2×10^{-6}	LN (2.0×10^{-6}, 2.5, 1.3×10^{-7}, 3.1×10^{-5})	(d)

Notes:

a) The Canadian Wildlife Service (CWS) report that moose eat 15 – 20 kg/d twigs and shrubs in the winter and 20 – 30 kg/d forage consisting of twigs, leaves, shrubs, upland and water plants in the summer.

b) Moose do not migrate and can be resident year around. Their home range is, on average, 5 to 10 square kilometres (Wilson and Ruff 1999). They prefer forested areas by lakes, ponds or swamps. Based on this information, it was assumed that they could obtain 25% of their food and water from the reference site.

c) Based on the allometric equation provided in U.S. EPA 1993 and a body weight of 600 kg for moose (CWS 1997).

d) Based on information available in IAEA 1994, U.S. NCRP 1996, Baes 1984, U.S. EPA 1998 and CSA 1987.

TABLE D3.2-5
DUCK CHARACTERISTICS

Parameter Description	Units	Normal Value	Distribution	Reference
Food ingestion rate[a]				
mallard,	g(wet wt.)/d	250	C (250)	Environment Canada 1999
common merganser		370	C (370)	
scaup		255	C (255)	
Fraction of time spent in study area[b]				
mallard, common merganser, scaup	-	0.5	U (0.33, 0.66)	U.S. EPA 1993
Fraction of food that is fish[c]				
common merganser	-	0.996	C (0.996)	Andress and Parker 1995
mallard, scaup		0.0	C (0.0)	U.S. EPA 1993
Fraction of food that is benthic invertebrates[c]				
scaup	-	0.89	C (0.89)	U.S. EPA 1993
mallard,		0.74	Calculated	U.S. EPA 1993
common merganser		0.0	C (0.0)	
Fraction of food that is pondweed[c]				
mallard	-	0.25	C (0.25)	U.S. EPA 1993
scaup		0.09	Calculated	U.S. EPA 1993
common merganser		0.0	C (0.0)	
Fraction of food that is sediment				
mallard	-	0.01	LN (0.01, 3.56, 0.001, 0.13)	Beyer *et al* 1994
scaup		0.02	C (0.02)	Beyer *et al* 1994
common merganser		0.004	C (0.004)	Beyer *et al* 1994
Water ingestion rate[d]				
mallard	m^3/d	6.2×10^{-5}	C (6.2×10^{-5})	U.S. EPA 1993
common merganser		7.6×10^{-5}	C (7.6×10^{-5})	U.S. EPA 1993
scaup		5.2×10^{-5}	C (5.2×10^{-5})	U.S. EPA 1993
Food-to-duck flesh transfer coefficient for:				
Arsenic	$\frac{\mu g/g \text{ (wet wt.)}}{\mu g/d}$	1.0×10^{-3}	LN (1.0×10^{-3}, 5.0, 4.0×10^{-5}, 2.5×10^{-2})	(e)

Notes:

a) Taken from Environment Canada (1999) along with body weights of 1082g for a mallard, 820g for a scaup and 1470g for common merganser.

b) Based on information that scaup and mallards migrate and spend 4 – 8 months away from this area.

c) Food types consumed assumed to comprise fish, benthic invertebrates and pondweed.

d) Based on the allometric equation published in U.S. EPA (1993).

e) Based on information available in IAEA (1994), Baes *et al*. (1984), U.S. EPA (1998) and CSA (1987).

TABLE D3.2-6
MINK CHARACTERISTICS

Parameter Description	Units	Normal Value	Distribution	Reference
Food ingestion rate[a]	g(wet)/d	220	C (220)	U.S. EPA 1993
Fraction of food from reference site[b]	-	1.0	C (1.0)	Assumed
Fraction of food that is hare[c]	-	0.05	C (0.05)	U.S. EPA 1993
Fraction of food that is fish[c]	-	0.65	C (0.65)	U.S. EPA 1993
Fraction of food that is benthic invertebrates[c]	-	0.09	C (0.09)	U.S. EPA 1993
Fraction of food that is pondweed[c]	-	0.05	C (0.05)	U.S. EPA 1993
Fraction of food that is duck[c]	-	0.05	C (0.05)	U.S. EPA 1993
Fraction of food that is muskrat	-	0.1	C (0.1)	U.S. EPA 1993
Fraction of food that is sediment	-	0.01	C (0.01)	Beyer *et al.* 1994
Water ingestion rate[d]	m^3/d	9.9×10^{-5}	C (9.9×10^{-5})	U.S. EPA 1993
Fraction of water from reference site[b]	-	1.0	C (1.0)	Assumed
Food-to-mink meat transfer coefficient for: Arsenic	µg/g (wet wt.) / µg/d	2.0×10^{-6}	LN (2.0×10^{-6}, 2.5, 1.3×10^{-7}, 3.1×10^{-5})	(e)

Notes:

a) Based on the food intake information provided in U.S. EPA (1993) and a body weight of 1kg for a mink (U.S. EPA 1993).
b) Because mink are territorial, it was assumed that they would obtain all their food and water from the reference site.
c) Based on the diet of mink provided in U.S. EPA (1993).
d) Based on the water intake of a mink provided in U.S. EPA (1993).
e) Based on information available in IAEA (1994), U.S. NCRP (1996), Baes *et al.* (1984), U.S. EPA (1998) and CSA (1987).

TABLE D3.2-7
GRAY WOLF CHARACTERISTICS

Parameter Description	Units	Normal Value	Distribution	Reference
Food ingestion rate[a]	g(wet)/d	5500	T (4000, 5500, 7000)	Fuller and Keith 1980
Fraction of food from reference site[b]	-	0.25	U (0.0, 0.5)	Assumed
Fraction of food that is moose[c]	-	0.4	C (0.4)	U.S. EPA 1993
Fraction of food that is caribou[c]	-	0.4	C (0.4)	U.S. EPA 1993
Fraction of food that is hare[c]	-	0.19	C (0.19)	U.S. EPA 1993
Soil ingestion rate	g/d	42.3	C (42.3)	Beyer *et al.* 1994
Water ingestion rate[d]	m^3/d	2.9×10^{-3}	C (2.9×10^{-3})	U.S. EPA 1993
Fraction of water from reference site[b]	-	0.25	U (0.0, 0.5)	Assumed
Food-to-wolf meat transfer coefficient for: Arsenic	µg/g (wet wt) µg/d	2.0×10^{-6}	LN ($2.0\text{x}10^{-6}$, 2.5, $1.3\text{x}10^{-7}$, $3.1\text{x}10^{-5}$)	(e)

Notes:

a) Based on study of Fuller and Keith (1980) which estimate that gray wolf in northeastern Alberta eat 5.5 kg/d. The Canadian Wildlife Service (1993b) estimate that gray wolves eat about 4 – 10 kg/d.

b) Because the gray wolf has a large home range, it was assumed that they would obtain one quarter to a maximum of half their food and water from the reference site.

c) Based on the intake of foxes from U.S. EPA (1993) and wolves from the Canadian Wildlife Service (1993b).

d) Based on the allometric equation provided in U.S. EPA (1993) and a body weight of 43 kg for a gray wolf (Schmidt and Gilbert 1978).

e) Based on information available in IAEA (1994), U.S. NCRP (1996), Baes *et al*. (1984), U.S. EPA (1998) and CSA (1987).

TABLE D3.2-8
BARRENGROUND CARIBOU CHARACTERISTICS

Parameter Description	Units	Normal Value	Distribution	Reference
Food ingestion rate	g(wet)/d	8000	T (5500, 8000, 10500)	based on Allaye-Chan *et al*. 1990; Holleman *et al* 1990; Garner 1972
Fraction of food from reference site[a]	-	0.1	C (0.1)	Assumed
Fraction of food that is lichen	-	0.91	Calculated as reminder of diet	based on Thomas & Barry 1991
Fraction of food that is summer forage[b]	-	0.01	T (0.0, 0.01, 0.02)	based on Thomas & Barry 1991
Fraction of food that is browse[b]	-	0.05	T (0.0, 0.05, 0.1)	based on Thomas & Barry 1991
Fraction of food that is soil	-	0.03	C (0.03)	Beyer *et al.* 1994
Water ingestion rate	m^3/d	$9.5x10^{-3}$	T ($4.75x10^{-3}$, $9.5x10^{-3}$, $14.2x10^{-3}$)	based on Kirk 1977 and Wales *et al*. 1975
Fraction of water from reference site[a]	-	0.1	C (0.1)	Assumed
Food-to-caribou meat transfer coefficients for Arsenic	$\frac{\mu g/g \text{ (wet wt)}}{\mu g/d}$	2×10^{-6}	LN ($2x10^{-6}$, 2.5, $1.3x10^{-7}$, $3.1x10^{-5}$)	(c)

Notes:

a) Body weight of 105 kg was assumed (Schmidt and Gilbert 1978). Because caribou have a very large home range and migrate to different areas in the north, it was assumed that they would obtain 10% of their food and water from the reference site. They are generally present in the Yellowknife area from October to December.

b) Browse is made up of shrubs and conifers and make up 5.3% of the diet, forage is approximately 1% and the remainder of the food is assumed to be lichen.

c) Based on information available in IAEA 1994, U.S. NCRP 1996, Baes 1984, U.S. EPA 1998 and CSA 1987.

TABLE D3.2-9
DEFAULT BLACK BEAR MODEL INPUT PARAMETERS

Parameter Description	Units	Normal Value	Distribution	Reference
Food ingestion rate[a]	g(wet)/d	19650	C (19650)	U.S. EPA 1993
Fraction of food from reference site[b]	-	0.25	C (0.25)	Assumed
Fraction of food that is herbaceous	-	0.33	C (0.33)	Holcroft and Herrero 1991
Fraction of food that is berries	-	0.40	C (0.40)	Holcroft and Herrero 1991
Fraction of food that is fish[c]	-	0.15	C (0.15)	Canadian Wildlife Service 1993a
Fraction of food that is caribou[c]	-	0.05	C (0.05)	Canadian Wildlife Service 1993a
Fraction of food that is moose[c]	-	0.05	C (0.05)	Canadian Wildlife Service 1993a
Fraction of food that is soil	-	0.02	C (0.02)	Beyer *et al*. 1994
Water ingestion rate[d]	m^3/d	0.013	C (0.013)	U.S. EPA 1993
Fraction of water from reference site[b]	-	0.25	C (0.25)	Assumed
Food-to-bear meat transfer coefficient for: Arsenic	$\frac{\mu g/g \text{ (wet wt)}}{\mu g/d}$	2.0×10^{-6}	LN ($2.0x10^{-6}$, 2.5, $1.3x10^{-7}$, $3.1x10^{-5}$)	(e)

Notes:

a) Based on the allometric equation provided in U.S. EPA (1993) and a body weight of 225 kg for a black bear.

b) Because bear have a large home range, it was assumed that they would obtain 25% of their food and water from the reference site.

c) The Canadian Wildlife Service (1993a) report that 75% of the diet of a black bear consists of berries, flowers and grasses where as 25% comprises fish and meat.

d) Based on the allometric equation provided in U.S. EPA (1993) and a body weight of 225 kg for a black bear.

e) Based on information available in IAEA (1994), U.S. NCRP (1996), Baes *et al*. (1984), U.S. EPA (1998) and CSA (1987).

TABLE D3.2-10
DEFAULT MUSKRAT MODEL INPUT PARAMETERS

Parameter Description	Units	Normal Value	Distribution	Reference
Food ingestion rate[c]	g(wet)/d	360	C (360)	U.S. EPA 1993
Fraction of food from reference site[a]	-	1.0	C (1.0)	Assumed
Fraction of food that is aquatic vegetation	-	0.98	C (0.98)	U.S. EPA 1993
Fraction of food that is sediment	-	0.01	LN (0.01, 3.56, 0.001, 0.13)	Beyer *et al*. 1994
Fraction of food that is snail	-	0.01	C (0.01)	U.S. EPA 1993
Water ingestion rate	m^3/d	1.2×10^{-4}	C (1.2×10^{-4})	U.S. EPA 1993
Fraction of water from reference site[a]	-	1.0	C (1.0)	Assumed
Food-to-muskrat meat transfer coefficient for: Arsenic	µg/g (wet wt) / µg/d	2.0×10^{-6}	LN (2.0×10^{-6}, 2.5, 1.3×10^{-7}, 3.1×10^{-5})	(b)

Notes:

a) Muskrats have a small home range (0.17 ha; U.S. EPA 1993). Therefore, it was assumed that muskrat will obtain all their food, water and soil from the reference site.
b) Based on information available in IAEA (1994), U.S. NCRP (1996), Baes *et al*. (1984), U.S. EPA (1998) and CSA (1987).
c) Based on information from U.S. EPA (1993) on food intake rates for muskrat and the body weight (1.2 kg).

D3.2.2 Aquatic Species

The Valued Ecosystem Components (VECs) discussed below by trophic level are those aquatic species considered important: (1) in the functioning of the ecosystem; (2) in the production of food for subsistence; or (3) due to their cultural or medicinal significance. No new studies specific to ecological receptors were performed for this assessment.

Primary Producers

i) *Aquatic Plants and Benthic Algae*

Aquatic plants and benthic algae in most lake ecosystems usually constitute the majority of the primary producer biomass. Aquatic plants are often consumed by moose, thereby forming a link between aquatic and terrestrial ecosystems. Aquatic plants are recognized as VECs due to their importance as food resources for moose and habitat to aquatic organisms. Benthic algae are consumed primarily by aquatic invertebrates, but are less important than the detritus.

Primary Consumers

i) Benthic Invertebrates

Chironomidae (midge) larvae are usually the most abundant benthic invertebrate taxa present in aquatic ecosystems. Midge larvae are selected as an ecological receptor due to the important role they play in aquatic ecosystems. Many species are detritivorous and thereby form an important link between the decomposer level and primary consumers. Furthermore, midge larvae are a main food source for small/juvenile fish and larger omnivorous fish. The adults are capable of flight and are frequently consumed by birds and bats. This life stage provides an important link between aquatic and terrestrial ecosystems in the region.

Due to the association of benthic invertebrates with sediments in aquatic ecosystems, they possess the greatest risk in terms of sediment contamination. Benthic invertebrates both live and feed within sediments and therefore may be exposed to contaminants through ingestion of sediment bound contaminants and also through exposure to interstitial waters within the sediment. Because of this strong relationship with sediments, benthic invertebrates are the primary ecological receptor when assessing the potential risk of sediment contaminant loading.

Secondary Consumers

Ecological receptors at the secondary consumer level include lake whitefish and sucker (both white sucker and longnose sucker). These three species feed largely on benthic invertebrates and smaller individuals are an important food source of larger predatory fishes. Larger lake whitefish are a valued subsistence and commercial fisheries species and larger suckers are frequently captured by bears and predatory birds during spawning, thereby providing a link with the surrounding terrestrial environment and humans. Lake whitefish and white suckers were chosen as ecological receptors in this study. Lake whitefish were only assessed in Back Bay and Yellowknife Bay since Baker Creek does not support this species.

Tertiary Consumers

Tertiary or terminal trophic level consists of larger predatory fish species which include northern pike, walleye, and lake trout. All three are considered ecological receptors from both an ecological and socio-economic perspective. Lake trout are found in most northern lakes; however, walleye and lake trout are less widespread. Lake trout are found in Great Slave Lake and are therefore, the only tertiary consumer considered as an ecological receptor in the assessment.

Table D3.2-11 provides a summary of the aquatic species used in this assessment.

TABLE D3.2-11
AQUATIC SPECIES SELECTED
FOR ECOLOGICAL RISK ASSESSMENT

Aquatic Species
Phytoplankton
Benthic Invertebrates
Zooplankton
Predator Fish
Bottom Feeder Fish

D4.0 CHARACTERIZATION OF EXPOSURE

The estimation of exposure of ecological receptors to potentially hazardous contaminants involves the identification of potential contaminants of concern and predicting the exposure considering temporal and spatial changes while incorporating the uncertainty involved.

D4.1 EXPOSURE PATHWAYS

Ecologically, exposure to arsenic occurs both directly and indirectly. Ecological receptors can be exposed directly to arsenic through environmental processes that determine contaminant concentrations in air, water, soils, and sediments. In this assessment, ingestion of water and soil/sediment is the primary route of direct exposure.

Indirect exposure results from ingestion of contaminated food (e.g. vegetation for herbivores) or prey organisms (e.g. omnivores, carnivores). Thus, the different patterns of exposure and differential sensitivity to elements of concern by organisms in complex food webs can result in ecological impacts that cannot be estimated accurately from considerations of direct effects alone.

Estimates of exposure can be provided by direct measurement or estimated using environmental transport models of varying complexity (e.g. Bartell *et al*. 1992; Burns *et al*. 1982; Mackay 1991). In this assessment, estimates of potential exposure were performed by environmental pathways modelling.

Exposures were predicted considering five different loadings of arsenic from the underground vaults into Baker Creek and the three segments of Yellowknife Bay.

D4.2 PATHWAYS MODEL

Exposure, via aquatic pathways, of human or terrestrial receptors can occur by consumption of water from Back Bay or Yellowknife Bay and consumption of fish taken from these bays. In addition, exposure can occur indirectly through consumption of terrestrial game (i.e. moose, ducks) which have consumed water or aquatic vegetation or benthic organisms from Baker Creek, Back Bay or Yellowknife Bay. Terrestrial species may also inadvertently ingest whole harvesting aquatic plants or benthic invertebrates.

Modelling the uptake of arsenic by aquatic biota can be quite involved and requires consideration of several interactions; for example, direct uptake from water by aquatic vegetation, sedimentation and deposition of aquatic vegetation to lake sediment, decomposition and release

of contaminants from lake sediments to the water phase, harvesting of aquatic vegetation by invertebrates, predation of large fish on small fish, etc. The bioaccumulation factors used in the aquatic biota model are an aggregate representation of numerous pathways and mechanisms of arsenic transfer through the aquatic environment.

Exposure to arsenic can also occur from terrestrial pathways as a result of ingestion of soils and terrestrial vegetation. Transfer of arsenic to terrestrial vegetation occurs through deposition on foliage from the atmosphere and uptake from soil through the root system.

The INTAKE pathways model, which was used in the application, was first developed by SENES (SENES 1985, 1986, 1987) to simulate the movement of contaminants released from uranium mining operations through the environment. The model predicts concentrations of contaminants in air, water, sediment and soil as well as uptake by aquatic and terrestrial biota. The model can also be used to estimate dose to humans resulting from exposure to contaminants via several potential pathways, specifically inhalation and ingestion.

The INTAKE model was developed within a probabilistic (Monte Carlo) framework which permits the user to explicitly take into account uncertainty in the input parameter values. This uncertainty is taken into account by specifying a probability distribution from each parameter that best describes the nature of variability found in environmental monitoring data, or where data are not available based on professional judgement. Table D4.2-1 lists parameter descriptions defining distributions used in the uncertainty analysis.

Input parameters, default values and distributions for the aquatic models are listed on Table D4.2-2. These parameters were obtained from site-specific data for Baker Creek and Yellowknife Bay as discussed in Appendix A.

Table D4.2-3 shows a comparison of historical predicted and measured fish, snail and aquatic vegetation concentrations in the Giant Mine area. It can be seen that the predicted concentrations generally compare well with the measured concentrations indicating that the model is performing as expected.

TABLE D4.2-1
DESCRIPTORS OF PARAMETER DISTRIBUTIONS USED IN UNCERTAINTY ANALYSIS

DISTRIBUTION	DISTRIBUTION DESCRIPTORS
Constant (C)	Constant
Normal (N)	Mean Standard Deviation Minimum Maximum
Lognormal (LN)	Geometric Mean Geometric Standard Deviation Minimum Maximum
Uniform (U)	Minimum Maximum
Log-uniform (LU)	Minimum Maximum
Triangular (T)	Minimum Mode Maximum
Beta (B)	Minimum Maximum Alpha shape parameter Beta shape parameter

TABLE D4.2-2
DEFAULT AQUATIC MODEL INPUT PARAMETERS

Parameter Description	Units	Normal Value	Default Parameter Distribution	Reference
Water-to-game fish flesh transfer coefficient:				
Arsenic	g/g (wet wt) g/m^3	5.2×10^{-5}	LN (5.2×10^{-5}, 2.8, 6.6×10^{-6}, 4.1×10^{-4})	Site-Specific
Water-to-lake trout flesh transfer coefficient:				
Arsenic	g/g (wet wt) g/m^3	5.2×10^{-5}	LN (5.2×10^{-5}, 2.8, 6.6×10^{-6}, 4.1×10^{-4})	Site-Specific
Water-to-whitefish flesh transfer coefficient:				
Arsenic	g/g (wet wt) g/m^3	5.2×10^{-5}	LN (5.2×10^{-5}, 2.8, 6.6×10^{-6}, 4.1×10^{-4})	Site-Specific
Water-to-aquatic vegetation transfer coefficient:				
Arsenic	g/g (wet wt) g/m^3	6.3×10^{-5}	LN (6.3×10^{-5}, 1.8, 1.9×10^{-5}, 2.0×10^{-4})	Site-Specific
Water-to-snail (benthic invertebrate) transfer coefficient:				
Arsenic	g/g (wet wt) g/m^3	6.3×10^{-5}	LN (3.2×10^{-5}, 3.5, 2.6×10^{-6}, 3.9×10^{-4})	Site-Specific

Note:

Distribution Types:
- C Constant (value).
- U Uniform (minimum, maximum).
- T Triangular (minimum, mode, maximum).
- LN Lognormal (geometric mean, geometric standard deviation, minimum, maximum).

TABLE D4.2-3
COMPARISON OF PREDICTED AND MEASURED CONCENTRATIONS IN AQUATIC BIOTA

Biota	Year	Predicted Value		Year	Measured Value	
		Mean	Range[d]		Mean	Range[d]
Fish[a] mg/kg (ww)	1990	0.187	(0.015, 1.0)	1992-1993	0.160	(0.01, 1.11)
	1995	0.144	(0.015, 0.79)			
	2010	0.114	(0.009, 0.63)	2003	0.15	(0.04-0.42)
Snail[b] mg/kg (dw)	1990	72	(3, 532)	1997	82.5	(82, 83)
	1995	48	(2, 377)			
Aquatic Vegetation[c] mg/kg (dw)	1995	73	(16-232)	2000	58	(0.52-260)

Note:

a –predicted concentrations for fish from North Yellowknife Bay and measured concentrations for fish from Yellowknife Bay and Back Bay (Jackson *et al.* 1996). Data from 2003 summarizes measurements of fish collected from Back Bay (de Rosemond *et al.* 2004)

b –predicted concentrations for snails from Baker Creek and measured concentrations from Baker Creek (Koch, 1998)

c – predicted concentration for aquatic vegetation from North Yellowknife Bay and measured concentrations from Baker Creek and Yellowknife Bay (Koch *et al.* 2000; Dillon 2002)

d – (minimum, maximum)

D4.2.1 Fish

The concentration of arsenic in game fish flesh is calculated as a function of the concentration in the water phase from the relationship:

$$[F]_n = B_f [R]_n \quad \text{(D4-1)}$$

where:

- $[F]_n$ = concentration of arsenic in game fish flesh at segment 'n' in the watershed (g/g (wet));
- B_f = the water-to-game fish flesh bioaccumulation factor of arsenic (g/g (wet) per g/m^3);
- $[R]_n$ = concentration of arsenic at segment 'n' in the watershed (g/m^3).

D4.2.2 Aquatic Vegetation

Arsenic levels in aquatic vegetation are calculated from the following aggregate model:

$$[C]_{av} = B_{av} [R_{av}] \quad \text{(D4-2)}$$

where:

- $[C]_{av}$ = concentration of arsenic in aquatic vegetation (g/g (wet));
- B_{av} = the water-to-aquatic vegetation transfer coefficient of arsenic (g/g (wet) per g/m^3);
- $[R_{av}]$ = average concentration of arsenic in the waters in the given segment (g/m^3).

The model calculates an average arsenic concentration in the aquatic vegetation.

C4.2.3 Aquatic Biota

Arsenic concentrations in plankton are calculated from the following aggregate model:

$$[C]_{bs} = k_D [R]_n \quad \text{(D4-3)}$$

where:

- $[C]_{bs}$ = concentration of arsenic in plankton (g/g (wet));
- k_D = distribution coefficient for arsenic between sedimenting material (both organic and inorganic fractions) and lake water (m^3/g);
- $[R]_n$ = concentration of arsenic at segment “n” in the watershed (g/m^3).

Arsenic levels in snails are calculated from the following aggregate model:

$$[C]_{sn} = B_{sn} [R]_n \quad \text{(D4-4)}$$

where:

- $[C]_{sn}$ = concentration of arsenic in snails (g/g (wet));

B_{sn} = the water-to-snail transfer coefficient of arsenic (g/g (wet) per g/m^3).

The model calculates the arsenic content in snails (benthic invertebrates) for each segment in the watershed.

D4.3 SOIL MODEL

The soil model simulates the movement in the soil of arsenic that is deposited as particulate matter from the air. The soil model includes atmospheric deposition, surface runoff, and downward transport of arsenic into the soil profile. The arsenic concentration in soil also be used to predict the dose due to consumption of vegetation grown in the soil via root uptake.

In this application arsenic concentrations in soil were directly input into the model as a distribution function. The data used to characterize arsenic levels in soil in the study area are presented in Appendix A. The data were evaluated to determine probability distributions of arsenic levels in soils specific to each receptor location. The distributions are included in Appendix A.

D4.4 TERRESTRIAL VEGETATION MODEL

In the terrestrial vegetation model, air and soil concentrations are used to model the concentration of arsenic in six classes of terrestrial vegetation: above ground leafy vegetables; below ground vegetables; summer forage; browse; lichen; and berries. One basic model is used for all types of terrestrial vegetation, which is described below.

Arsenic can be transferred to the edible portions of fruits and vegetables consumed by people or to forage consumed by animals either by direct foliar retention or root uptake. The following equation was adapted from work presented by the U.S. NRC (1982) to calculate the average annual concentration of arsenic in each vegetation type:

$$C_v = C_a V_d F_{in} F_{rv} E_v \left[\frac{1 - \exp(-\lambda_{wv} t_v)}{Y_v \lambda_{wv}} \right] + \frac{B_v C_g}{\rho_g} \qquad \text{(D4-5)}$$

where,

C_v = the resulting concentration of arsenic in each specific vegetation type (g/g (wet weight))

C_a = the average concentration of arsenic in the air (g/m^3)

V_d = the deposition velocity (m/s)

F_{in} = the fraction of the deposition that is intercepted by each specific vegetation type (0 to 1)

F_{rv} = the fraction of the total deposition retained on the plant surface of each specific vegetation type (0 to 1)

E_v = the fraction of the foliar deposition retained on edible portions of each specific vegetation type (0 to 1)

λ_{wv} = a decay constant accounting for weathering losses from each specific vegetation type (s^{-1})

t_v = the duration of the plant exposure to atmospheric deposition (s)

Y_v = the yield density for each specific vegetation type (g (wet weight) /m^2)

B_v = the soil-to-plant transfer coefficient for arsenic and each specific vegetation type (g/g (wet weight) plant per g/g (dry weight) soil))

C_g = the average concentration of arsenic in the root zone of the soil (g/m^3) (calculated in soil model)

ρ_g = density of soil (g (dry weight) /m^3)

The first term on the right hand side of equation (D4-5) represents direct foliar retention; the second term represents root uptake. Root uptake is considered to be insignificant for lichen (due to the limited root structure) and berries (negligible translocation to berries from root uptake). Irrigation is not included in this application. Note also that time dependence is due only to the time dependence of the air and soil concentrations and vegetation is calculated based only on the conditions at each time step.

The model assumes that the system is in equilibrium. For the foliar deposition model, this means that the weathering half-life is sufficiently less than the period of deposition. Therefore the concentration of arsenic reaches a steady state level. For vegetation that has a lifespan of one growing season, this is a valid assumption.

Vegetation model default input parameters are summarized in Table D4.4-1. Site-specific transfer factors for arsenic for above ground and below ground vegetation were obtained from data in the Yellowknife area as discussed in Appendix A. Further discussion of the vegetation model can be found in the UTAP Version 3 Component Model Documentation (SENES 1987).

Table D4.4-2 provides a comparison between measured lichen data (Koch et al. 2000) and the predicted concentrations. A moisture content of 40% was assumed to allow a comparison of the modelled concentrations to the measured levels in mg/kg dw. The comparison shows that the predicted values are reasonable, however the extreme value is not captured.

TABLE D4.4-1
TERRESTRIAL VEGETATION MODEL INPUT PARAMETERS

Parameter Description	Units	Normal Values	Distribution	Reference
fraction of deposition intercepted by plant type (F_{in})				
above ground vegetables	-	1.0	C (1.0)	Assumed
below ground vegetables		1.0	C (1.0)	Assumed
summer forage		1.0	C (1.0)	Assumed
browse		1.0	C (1.0)	Assumed
lichen		1.0	C (1.0)	Assumed
berries		0.027	T (0.02, 0.027, 0.034)	Baes *et al*. 1984
fraction of total deposition retained on plant surface (F_{rv})				
above ground vegetables	-	0.2	U (0.1, 0.3)	SENES 1987
below ground vegetables		0.2	U (0.1, 0.3)	SENES 1987
summer forage		0.2	U (0.1, 0.3)	SENES 1987
browse		0.2	U (0.1, 0.3)	SENES 1987
lichen		0.95	U (0.8, 1.0)	SENES 1987
berries		0.2	U (0.1, 0.3)	Beak 1987
fraction of plant deposition retained on edible portions (E_v)				
above ground vegetables	-	1.0	U (0.8, 1.0)	SENES 1987
below ground vegetables		0.1	U (0.05, 0.15)	SENES 1987
summer forage		1.0	U (0.8, 1.0)	SENES 1987
browse		1.0	U (0.8, 1.0)	SENES 1987
lichen		1.0	U (0.8, 1.0)	SENES 1987
berries		1.0	U (0.8, 1.0)	Beak 1987
duration of exposure to deposition (t_v)				
above ground vegetables	s	5.2×10^6	T (2.6x106, 5.2x106, 7.8x106)	SENES 1987
below ground vegetables		5.2×10^6	T (2.6x106, 5.2x106, 7.8x106)	SENES 1987
summer forage		2.6×10^6	U (1.3x106, 3.9x106)	SENES 1987
browse		7.8×10^6	U (6.8x106, 8.8x106)	Beak 1987
lichen		1.0×10^{10}	C (1x1010)	SENES 1987
berries		7.8×10^6	U (6.8x106, 8.8x106)	Beak 1987
yield density (Y_v)				
above ground vegetables	g(wet)/ m^2	2.0×10^3	U ($1.0x10^3$, $3.0x10^3$)	SENES 1987
below ground vegetables		2.0×10^3	U ($1.0x10^3$, $3.0x10^3$)	SENES 1987
summer forage		7.5×10^2	U ($5.0x10^2$, $1.0x10^3$)	SENES 1987
browse		7.5×10^2	U ($5.0x10^2$, $1.0x10^3$)	SENES 1987
lichen		5.0×10^2	U ($4.0x10^2$, $6.0x10^2$)	SENES 1987
berries		1.0×10^2	U (75, 125)	Beak 1987

TABLE D4.4-1 (CONT'D)
TERRESTRIAL VEGETATION MODEL INPUT PARAMETERS

Parameter Description	Units	Normal Values	Distribution	Reference
decay constant accounting for weathering loss (λ_{wv})				
above ground vegetables	s^{-1}	5.73×10^{-7}	U ($2.9x10^{-7}$, $8.6x10^{-7}$)	SENES 1987
below ground vegetables		5.73×10^{-7}	U ($2.9x10^{-7}$, $8.6x10^{-7}$)	SENES 1987
summer forage		5.73×10^{-7}	U ($2.9x10^{-7}$, $8.6x10^{-7}$)	SENES 1987
browse		5.73×10^{-7}	U ($2.9x10^{-7}$, $8.6x10^{-7}$)	SENES 1987
lichen*		7.3×10^{-9}	U ($3.65x10^{-9}$,$1.1x10^{-10}$)	SENES 1987
berries		5.73×10^{-7}	U ($2.9x10^{-7}$, $8.6x10^{-7}$)	Beak 1987
soil-to-plant transfer coefficient (B_v)				
above ground vegetables				
Arsenic	µg/g (wet) / µg/g (dry)	$2.6x10^{-3}$	β (0.0002, 0.018, 0.772, 4.634)	Site-Specific
below ground vegetables				
Arsenic	µg/g (wet) / µg/g (dry)	$1.0x10^{-3}$	β (0.0003, 0.003, 2.84, 6.040)	Site-Specific
summer forage				
Arsenic [a]	µg/g (wet) / µg/g (dry)	$2.6x10^{-3}$	β (0.0002, 0.018, 0.772, 4.634)	Site-Specific
browse				
Arsenic [a]	µg/g (wet) / µg/g (dry)	$2.6x10^{-3}$	β (0.0002, 0.018, 0.772, 4.634)	Site-Specific

Note:

a It was assumed that the data for above ground vegetation could applied to browse and summer forage.

Distribution Types:

C Constant (value).
U Uniform (minimum, maximum).
T Triangular (minimum, mode, maximum).
β Beta (geometric mean, geometric standard deviation, alpha, beta).

TABLE D4.4-2
COMPARISON OF PREDICTED AND MEASURED LICHEN CONCENTRATIONS (mg/kg (dw))

Predicted Value		Measured Value	
Median	Range[a]	Geometric Mean	Range[a]
28	(2.8-117)	56	(6.4-2300)

a – (minimum-maximum)

D4.5 TERRESTRIAL BIOTA MODELS

The terrestrial biota model calculates the concentration of arsenic in animal flesh.

D4.5.1 Terrestrial Herbivores

For herbivores, the calculation of the concentration of arsenic may include many possible subsets of browse, forage, berries, lichen, above ground vegetables, water, or aquatic vegetation-to-herbivore flesh pathways.

Arsenic in terrestrial and aquatic vegetation and water may be eaten by herbivores, which may in turn be eaten by omnivores, carnivores or people. The transfer of arsenic is modelled as follows:

$$[C]_h = \left(\sum \left([C]_v F_{h,v}\right) Q_h + [C]_w Q_{h,w}\right) F_L T_h \qquad \text{(D4-6)}$$

where:

- $[C]_h$ = annual average concentration of arsenic in herbivore (type "h") flesh (g/g (wet weight));
- $[C]_v$ = average concentration of arsenic in vegetation type "v" (g/g (wet weight));
- $F_{h,v}$ = fraction of feed for herbivore type "h" that is vegetation type "v";
- Q_h = average daily feed intake of herbivore type "h" (g (wet weight)/d);
- $[C]_w$ = average concentration of arsenic in water (g/m^3);
- $Q_{h,w}$ = average daily water intake of herbivore type "h" (m^3/d);
- T_h = feed-to-herbivore (type "h") flesh transfer coefficient for arsenic (g/g (wet weight) per g/d);
- FL = herbivore type "h" fraction of year in area (default = 1.0).

D4.5.2 Omnivores

In addition to vegetation and drinking water, omnivores may consume aquatic and/or terrestrial herbivores, omnivores or carnivores. The transfer of arsenic to omnivores is modelled as follows:

$$[C]_o = \left\{\left(\sum \left([C]_v F_{o,v}\right) Q_{o,f} + \left(\sum [C]_m F_{o,m}\right) Q_{o,m}\right) T_{o,f} + \left([C]_w Q_{o,w} T_{w,o}\right)\right\} F_L \qquad \text{(D4-7)}$$

where:

- $[C]_o$ = annual average concentration of arsenic in omnivore (type "o") flesh (g/g (wet weight));
- $[C]_v$ = average concentration of arsenic in vegetation type "v" (g/g (wet weight));
- $F_{o,v}$ = fraction of feed for omnivore type "o" that is vegetation type "v";
- $Q_{o,f}$ = average daily feed intake of omnivore type "o" (m^3/d);

$[C]_m$ = average concentration of arsenic in meat flesh type "m" (g/g (wet weight));
$F_{o,m}$ = fraction of feed for omnivore type "o" that is meat flesh type "m";
$T_{f,o}$ = feed-to-omnivore (type "o") flesh transfer coefficient for arsenic (g/g (wet weight) per g/d);
$[C]_w$ = average concentration of arsenic in water ((Bq/m^3) or (g/m^3));
$Q_{o,w}$ = average daily water intake of omnivore type "o" (m^3/d);
$T_{w,o}$ = water-to-omnivore (type "o") flesh transfer coefficient (g/g (wet weight) per g/d);
F_L = omnivore type "o" fraction of year in area (default = 1.0).

D4.5.3 Carnivores

Carnivores consume both drinking water and aquatic and/or terrestrial herbivores, omnivores or carnivores. The transfer for each constituent to carnivores is modelled as follows:

$$[C]_c = \left\{\left(\sum [C]_m F_{c,m}\right) Q_{c,m} T_{c,m} + \left([C]_w Q_{c,w} T_{w,c}\right)\right\} F_L \qquad \text{(D4-8)}$$

where:

$[C]_c$ = annual average concentration of arsenic in carnivore (type "c") (g/g (wet weight));
$Q_{c,m}$ = average daily feed intake of carnivore type "c" (m^3/d);
$[C]_m$ = average concentration of arsenic in meat flesh type "m" (g/g (wet weight));
$F_{c,m}$ = fraction of feed for carnivore type "c" that is meat flesh type "m";
$T_{c,m}$ = feed-to-carnivore (type "c") flesh transfer coefficient (g/g (wet weight) per g/d);
$[C]_w$ = average concentration of arsenic in water (g/m^3);
$Q_{c,w}$ = average daily water intake of carnivore type "c" (m^3/d);
$T_{w,c}$ = water-to-carnivore (type "c") flesh transfer coefficient (g/g (wet weight) per g/d);
F_L = carnivore type "c" fraction of year in area (default = 1.0).

The intake rates and transfer co-efficients are all provided in Tables D3.2-2 to D3.2-10.

D4.5.4 Comparison to Predicted Values

The predicted terrestrial biota concentrations were compared to available data on arsenic levels in terrestrial biota. The comparison, shown on Table D4.5-1, is limited due to the detection limits for analysis of arsenic. In general, the predicted values compare well to the measured data with the upper range of the predicted values at a much higher concentrations than the maximum

measured values. This suggests that the model provides a reasonable, but cautious estimate of the exposure for terrestrial biota.

TABLE D4.5-1
COMPARISON OF MEASURED AND MODELLED CONCENTRATIONS FOR TERRESTRIAL RECEPTORS

	Basis	Measured Concentration[a] (mg/kg)	Modelled Concentration (mg/kg)	
			Mean	Range[b]
Caribou	DW	<0.2	0.15*	(0.003-1.56)*
Moose	DW	<0.2	0.36*	(0.01-3.67)*
Grouse	WW	<0.07	0.58	(0.01-6.76)
Mallard	WW	<0.03-0.11	0.13**	(0.004-3.6)
Scaup	WW	<0.03-0.426	0.21**	(0.003-3.02)
Merganser	WW	<0.03-0.232	0.12	(0.001-3.2)

DW - dry weight WW - wet weight
* converted to a dry weight basis based on an assumed 70% moisture content
[a] See Section 2.3.3 of the main document for a discussion of the measured data.
[b] (minimum, maximum) ** - modelled concentrations for North Yellowknife Bay

D4.6 UNCERTAINTIES INVOLVED

Physical, chemical, and biological processes can contribute to uncertainties in characterizing exposure to chemicals of concern. Incomplete data or understanding of physical transport processes or physical processes that cause chemicals to concentrate or degrade can result in inaccurate or imprecise estimates of exposure.

Complex environmental chemistry determines the speciation of chemicals in the environment; the proportions of different ions, isomers, valence states, complexes, etc., can vary significantly in time and space. Rarely is this detailed information available for estimating exposure. Indeed, even the best models and data are imperfect reflections of the real system. Necessary simplifying assumptions introduce bias and imprecision in estimates of exposures. In the case of

novel chemicals, estimates of exposure may have to rely on extrapolations from structurally similar compounds (i.e. quantitative structure activity relations, QSARs).

The processes and dynamics of chemical uptake, accumulation, transformation, and metabolism differ among organisms and among chemicals. The assimilation of chemicals from consuming and digesting contaminated food items remains poorly quantified. This incomplete understanding introduces further uncertainty into estimates of exposure. In this analysis, such uncertainties were taken into account by assigning ranges of values to reflect environmental fate and uptake.

D5.0 ESTIMATED INTAKE OF ARSENIC FROM MARKET (STORE-BOUGHT) FOODS

An estimate of the arsenic intake from the food purchased at the grocery store (market foods) was obtained for use in determining the total exposure to arsenic. An estimate of the intake of market foods depends on the typical arsenic levels in food and general intake rates.

D5.1 Arsenic Concentration in Market Foods

A study was conducted to measure the level in arsenic in market foods across Canada (Dabeka *et al*. 1993). The study included 112 food composites from 6 Canadian studies. The results of the study were grouped into several categories, which are summarized in Table D5.1-1.

TABLE D5.1-1
ARSENIC LEVELS IN FOOD

Food Category	Arsenic Concentrations (ng/g)		
	Mean	Minimum	Maximum
Milk and dairy products	3.8	0.4	26
Meat and poultry	24.3	1.3	536
Fish and shellfish	1662.4	77	4830
Soups	4.2	0.2	11
Bakery goods and cereals	24.5	0.1	365
Vegetables	7	0.1	84
Fruit and fruit juices	4.5	0.1	37
Fats and oils	19	1	57
Sugar and candies	10.9	1.4	105
Beverages	3	0.4	9
Miscellaneous	12.5	0.8	41

Source: Dabeka *et al.* 1993

D5.2 Typical Intake Rates

The intake rates for the general Canadian population were obtained from Health Canada (1994). This reference was selected over other compilations such as Richardson (1997) because Health Canada (1994) had intakes for food categories which matched the categories in Dabeka *et al.* (1993).

Table D5.2-1 provides the intake rates for the general Canadian population for an adult and a child.

TABLE D5.2-1
TYPICAL INTAKE RATES FOR CANADIAN POPULATION

Food Category	Mean Intake Rate (g/d)	
	Child	Adult
Milk and dairy products	609	283
Meat and poultry	112	169
Fish and shellfish	8.37	14
Soups	82	100
Bakery goods and cereals	300	247
Vegetables	198	250
Fruit and fruit juices	202	186
Fats and oils	21	25
Sugar and candies	57	57
Beverages	196	255
Miscellaneous	13	12

Source: Health Canada 1994

The information in Table D5.2-1 was used for the estimate of the intake of market food for Receptors 1 and 3. The intake of market foods was determined as follows:

Intake of Market Foods = Total Consumption (Table D5.2-1) - Intake of Food from Giant Mine site and Yellowknife Area (Section 4)

That is, the amount of food consumed by the receptor that was accounted for in the modeling (i.e. food from the Giant Mine site and Yellowknife area) was subtracted from the total consumption rate to provide an estimate of the amount of food not included in the detailed modeling.

For Receptors 2 and 4 information on the intake of market foods were obtained from data presented in Receveur *et al.* (1996) for the Dogrib region. This report provided a breakdown of the market foods for adults. The intakes were grouped into the same categories as shown in Table D5.2-1. The intake for a child was estimated assuming that the ratio of child to adult of a particular category of food for the general population (Table D5.2-1) could be applied to the information for the Dogrib region. The intake of market foods for Receptors 2 and 4 is provided in Table D5.2-2.

TABLE D5.2-2
INTAKE OF MARKET FOODS BASED ON THE DOGRIB REGION

Food Category	Mean Intake Rate of Market Foods (g/d)	
	Child	Adult
Milk and dairy products	82.5	38.3
Meat and poultry	112.8	170.9
Fish and shellfish	0.6	1.0
Soups	63.8	77.8
Bakery goods and cereals	228.5	188.1
Vegetables	103.6	130.8
Fruit and fruit juices	77.7	71.5
Fats and oils	12.6	15.0
Sugar and candies	30.3	30.3
Beverages	1470.4	1913.0
Miscellaneous	33.0	30.5

Source: Receveur *et al.* 1996

A comparison between Table D5.2-1 for the general population and Table D5.2-2 based on the Dogrib region shows a lower consumption of milk and dairy products and fruits and vegetables. This is not surprising given that these types of foods are not readily available in this region. The beverage intake, however for the Dogrib region is much higher than the general population. As a significant amount of the beverage intake is coffee and tea, the beverage intake for the child is likely overstated.

D5.3 INTAKE RATES OF ARSENIC FROM MARKET FOODS

The intake of arsenic from market foods was calculated by multiplying the intake of a specific food group (using the information in Tables D5.2-1 and D5.2-2) by the concentration of arsenic in the given group (Table D5.1-1) and dividing by the appropriate body weight (70 kg for an adult and 35 kg for a child). The results of this calculation are shown in Table D5.3-1.

TABLE D5.3-1
ESTIMATED INTAKE OF ARSENIC IN MARKET FOODS

Food Category	Background Intake (mg/(kg d))			
	Receptor 1 and 3		Receptor 2 and 4	
	Adult	Child	Adult	Child
Milk and dairy products	1.5×10^{-5}	6.6×10^{-5}	2.1×10^{-6}	9.0×10^{-6}
Meat and poultry	4.5×10^{-5}	5.8×10^{-5}	5.9×10^{-5}	7.8×10^{-5}
Fish and shellfish	0	0	2.4×10^{-5}	2.9×10^{-5}
Soups	6.0×10^{-6}	9.8×10^{-6}	4.7×10^{-6}	7.7×10^{-6}
Bakery goods and cereals	8.6×10^{-5}	2.1×10^{-4}	6.6×10^{-5}	1.6×10^{-4}
Vegetables	2.4×10^{-5}	3.8×10^{-5}	1.3×10^{-5}	2.1×10^{-5}
Fruit and fruit juices	1.1×10^{-5}	2.5×10^{-5}	4.6×10^{-6}	1.0×10^{-5}
Fats and oils	6.8×10^{-5}	1.1×10^{-5}	4.1×10^{-6}	6.9×10^{-6}
Sugar and candies	8.9×10^{-5}	1.8×10^{-5}	4.7×10^{-6}	9.4×10^{-6}
Beverages	1.1×10^{-5}	1.7×10^{-5}	8.2×10^{-5}	1.3×10^{-4}
Miscellaneous	2.1×10^{-6}	4.6×10^{-6}	5.4×10^{-6}	1.2×10^{-5}
TOTAL	2.0×10^{-4}	4.4×10^{-4}	1.8×10^{-4}	3.3×10^{-4}

Table D5.3-1 shows that for all receptors the intake of bakery goods and cereals represents a significant contribution to the intake of arsenic from market foods. As well, vegetables are also a significant contributor. For Receptors 2 and 4 the intake of beverages represents a significant source of the arsenic intake from market foods.

In addition to this deterministic calculation, the estimation of the arsenic intake from market foods was completed in a probabilistic manner within the INTAKE Model. The concentration of arsenic in food was assigned a triangular distribution between the minimum, mean and maximum shown in Table D5.1-1. The intake rates shown in Tables D5.2-1 and D5.2-2 were assigned lognormal distributions and a geometric standard deviation of 1.5 was applied to all food categories. This approximation is based on the information provided by Richardson (1997). The estimate of intake from market foods was included in the calculation of total dose for each of the receptors.

D6.0 REFERENCES

Allaye-Chan, A.C., R.G. White, D.F. Holleman and D.F. Russell 1990. *Seasonal Concentrations of Cesium0137 in Rumen Content, Skeletal Muscles and Faeces of Caribou from the Porcupine Herd: Lichen Ingestion Rates and Implication for Human Consumption.* Rangifer Spec. Issue 3:17-34.

Andress, S. and G.H. Parker 1995. *Radium-226 Levels in Eggs and Body Tissues of Herring Gulls and Common Mergansers Nesting Near the Elliot Lake Uranium Mine Tailings.* Presentation at the Environmental Assessment Panel Reviewing Decommissioning Proposals for Elliot Lake Uranium Mine Tailings Management Areas (TMAs). Transcript of Proceedings. 16 November. Volume 3.

Baes, C.F. R.D. Sharp, A.L. Sjoreen, and R.W. Shar 1984. *A Review and Analysis of Parameters for Assessing Transport of Environmentally Released Radionuclides Through Agriculture.* ORNL-5786.

Bartalucci, A., B. Weinstein and T. Dewey 2000. "Alces Alces" (On-Line), Animal Diversity Web. Accessed August 20, 2004 at http://animaldiversity.ummz.umich.edu/site/accounts/information/Alces_alces.html.

Bartell, S.M., R.H. Gardner and R.V. O'Neill 1992. *Ecological Risk Estimation.* Lewis Publishers, Inc., Chelsea, Michigan.

Beak Consultants Ltd. 1987. *Application of the National Uranium Tailings Program Probabilistic System Model (UTAP) to the Rabbit Lake Uranium Tailings Site*. Research Report Prepared for the National Uranium Tailings Program, CANMET, EMR, Ottawa.

Belovsky, G.E., P.A. Jordan, and D.B. Botkin 1973. *Summer Browsing by Moose in Relation to Preference, and Animal Density: A New Quantitative Approach.* Proceedings of the 9th North American Moose Conference and Workshop, Quebec, City, March.

Beyer, W. N., E. Connor, and S. Gerould 1994. *Survey of Soil Ingestion by Wildlife*. Journal of Wildlife Management, 58:375-382.

Burns, L.A., D.M. Cline, and R.R. Lassiter 1982. *Exposure Analysis Modelling System (EXAMS): User Manual and System Documentation.* EPA/600/3-82/023. U.S. Environmental Protection Agency, Environmental Research Laboratory, Athens, Georgia.

Canadian Council of Ministers of the Environment (CCME) 1997. *A Framework for Ecological Risk Assessment: Technical Appendices.* March.

Canadian Council of Ministers of the Environment (CCME) 1996. *A Framework for Ecological Risk Assessment: General Guidance.* March.

Canadian Standards Association (CSA) 1987. *Guidelines for Calculating Derived Release Rate Limits for Radioactive Material in Airborne and Liquid Effluents for Normal Operation of Nuclear Facilities.* CAN ICSA-N288.1-M87.

Canadian Wildlife Service (CWS) 1997. *Hinterland Who's Who: Moose On-Line Report.* Environment Canada.

Canadian Wildlife Service (CWS) 1993a. *Hinterland Who's Who: Bear On-Line Report.* Environment Canada.

Canadian Wildlife Service (CWS) 1993b. *Hinterland Who's Who: Wolf On-Line Report.* Environment Canada.

Coffman, W.P. 1978. *Chironomidae, An Introduction to the Aquatic Insects of North America.* In R.W. Merritt and K.W. Cummins [eds.] Kendall/Hunt Publishing Co., Dubuque., 345-376.

Cole, G.A. 1983. *Textbook of Limnology.* The C.V. Mosby Co., St. Louis. 401 p.

Dabeka, R.W., A.D. McKenzie, G.M.A. Lacroix, C. Cleroux, S. Bowe, R.A. Graham, H.B.S. Conacher and P. Verdier 1993. *Survey of Arsenic in Total Diet Food Composites and Estimation of the Dietary intake of Arsenic by Canadian Adults and Children*. Journal of AOAC International. Vol. 76. No. 1. pp14-25.

de Rosemond, S., K. Liber and Q. Xie 2004. *Arsenic Concentration and Speciation in Fish from Back Bay Near Yellowknife, NT.* Draft Report Submitted to Department of Indian Affairs and Northern Development, Yellowknife. June.

Dillon Consulting Limited 2002. *Biological Sampling at Baker Creek – Summary Report.* Department of Indian and Northern Affairs Development. April 2002. 01-9670-3000.

Environment Canada 1999. *Canadian Environmental Quality Guidelines.* Table 1 Body Weights and Daily Food Ingestion Rates of Avian Species that Consume Aquatic Biota. www.ec.gc.ca/ceqg-rcque/English/html/Table1_tissue.cfm

Environment Canada 1997. *Environmental Assessments of Priority Substances Under the Canadian Environmental Protection Act.* Guidance Manual Version 1.0. March. EPS/2/CC/3E.

Fry, J.C. 1982. *Interactions Between Bacteria and Benthic Invertebrates*, 171-201. In D.B Nedwell and C.M. Brown [eds.] Sediment microbiology. Academic Press, London., 171-201.

Fuller, T.K. and L.B. Keith 1980. *Wolf Population Dynamics and Prey Relationships in Northeastern Alberta.* J. Wildl. Mange. 44:583-602.

Garner, R.J. 1972. *Transfer of Radioactive Materials form the Terrestrial Environment to Animals and Man.* CRC Press. The Chemical Rubber Company.

Harvey, H.W. 1950. *On the Production of Living Matter in the Sea Off Plymouth.* Journ. Mar. Biol. Assoc. 29: 7-137.

Health Canada 1994. *Human Health Risk Assessment for Priority Substances.* Canadian Environmental Protection Act

Holcroft, A.C. and S. Herrero 1991. *Black Bear, Ursus americanus, Food Habits in Southwestern Alberta.* An. Field-Nat. 105:335-345.

Holleman, D.F., R.G. White and A.C. Allaye-Chan 1990. *Modelling of Radiocesium Transfer in the Lichen-Reindeer/Caribou-Wolf Food Chain.* Rangifer Spec. Issue 3:39-42.

Hough, C., C. Ollson, and I. Koch 2000. *Animal Trapping Report: Bioavailability of Arsenic in Yellowknife, NWT.* Prepared by Environmental Sciences Group Royal Military College, Kingston, Ontario. RMC-CCS-CE-00-21. November.

International Atomic Energy Agency (IAEA) 1994. *Handbook of Parameter Values for the Prediction of Radionuclide Transfer in Temperature Environments.* International Atomic Energy Agency, Vienna.

Jackson, F.J., D.N. Lafontaine and J. Klaverkamp 1996. *Yellowknife – Back Bay Study on Metal and Trace Element Contamination of Water, Sediment and Fish.* November.

Kirk, R.W. (ed.), 1977. *Current Veterinary Therapy.* W.B. Sauders Co., Toronto, pp 1418.

Koch, I. 1998. *Arsenic and Antimony Species in the Terrestrial Environment.* Thesis. University of Waterloo.

Koch, I., L. Wang, C.A. Ollson, W.R. Cullen and K.J.Reimer 2000. *The Predominance of Inorganic Arsenic Species in Plants from Yellowknife, Northwest Territories, Canada.* Environmental Science and Technology. 34(1): 22-26.

Larsen, J.A. 1980. *The Boreal Ecosystem*. Academic Press, New York. 500 pp.

Likens, G.E. and F.H. Bormann 1972. *Nutrient Cycling in Ecosystems*, pp. 25-67. In J.A. Weins [ed.] Ecosystem structure and function. Oregon State University Press, Orvallis.

Mackay, D. 1991. *Multimedia Environmental Models: The Fugacity Approach.* Lewis Publishers, Chelsea, Michigan.

Moore, J.W. S.J. Wheeler and D.J. Sutherland 1978. *The Effects of Metal Mines on Aquatic Ecosystems in the Northwest Territories II.* Giant Yellowknife Mines Limited. Report EPS 5-NW-78-9. June.

Newfoundland Government 2001. Salmonier Nature Park Website, Animals: Spruce Grouse On-line Report.

Pease, J.L., R.H. Vowles and L.B. Keith 1979. *Interaction of Snowshoe Hares and Woody Vegetation.* J. Wildl. Manage, 43:43-60.

Rawson, D.S. 1953. *The Bottom Fauna of Great Slave Lake.* Journal of the Fisheries Research Board of Canada, 10(8): 486-520.

Receveur, O., M. Boulay, C. Mills, W. Carpenter and H. Kuhnlein 1996. *Variance in Food Use in Dene/Metis Communities.* Centre for Indigenous Peoples' Nutrition and Environment (CINE). October.

Richardson, G.M. 1997. *Compendium of Canadian Human Exposure Factors for Risk Assessment.* O'Connor Associates.

Schmidt, J.L. and D.L. Gilbert 1978. *Big Game of North America, Ecology and Management.* Stackpole Books, Harrisburg, PA 17105.

SENES Consultants Limited 1987. *Uranium Tailings Assessment Program (UTAP.3): Component Model Documentation*. Research Report Prepared for the National Uranium Tailings Program, CANMET, EMR, under Supply and Services Canada Contract No. 0SQ85-00182.

SENES Consultants Limited 1986. *Probabilistic Model Development for the Assessment of the Long-Term Effects of Uranium Mill Tailings in Canada – Phase III.* Research Report Prepared for the National Uranium Tailings Program, CANMET, EMR, under Supply and Services Canada Contract No. 0SQ85-99182.

SENES Consultants Limited 1985. *A Study to Develop and Demonstrate Techniques for the Probabilistic Assessment of the Long-Term Effects of Uranium Mill Tailings in Canada – Phase II.* Research Report Prepared for the National Uranium Tailings Program, CANMET, EMR, under Supply and Services Canada Contract No. 0SQ84-00207.

Smith, R.L. 1980. *Ecology and Field Biology.* Harper and Row, Publishers, New York. 835 p.

Sutherland, D. 1989. *Assessment of Gold Mine Impacts on the Benthic Environment of Yellowknife Bay, NWT.* March.

Thomas, D.C. and S.J. Barry 1991. *Microhistological Analyses of Caribou Diet: Fecal Versus Rumen Samples and Other Variables.* Pages 516-529 in Proc. Fourth North Amer. Caribou Workshop. C.E. Buttler and S.P. Mahoney (eds). Newf.-Lab. Wild1. Div., St. John's, Newfoundland.

United States Environmental Protection Agency (U.S. EPA) 1998. *Human Health Risk Assessment Protocol for Hazardous Waste Combustion Facilities.* Peer Review Draft. EPA 530-D-98-001.

United States Environmental Protection Agency (U.S. EPA) 1993. *Wildlife Exposure Factors Handbook. Volume I and II.* Office of Research and Development, Washington DC.

United States Environmental Protection Agency (U.S. EPA) 1992. *Framework for Ecological Risk Analysis.* EPA/630/R-92-001. Washington DC.

United States National Council on Radiation Protection and Measurements (U.S. NCRP) 1996. *Screening Models for Release of Radionuclides to Atmosphere, Surface Water and Ground.* NCRP Report No. 123.

United States Nuclear Regulatory Commission (U.S. NRC) 1982. *Calculational Models for Estimating Radiation Doses to Man from Airborne Radioactive Materials Resulting from Uranium Milling Operations.* Reg. Guide, 3.51, March.

Vanriel, P. 1989. *Thermodynamic Relationships Between Trophic Levels in a Small Lacustrine Ecosystem in the Canadian Arctic.* M.Sc. thesis, University of Manitoba, Winnipeg. xxii + 380 p.

Wales, R.A., L.P. Milligan and E.H. McEwan 1975. *Urea Recycling in Caribou, Cattle and Sheep.* In First Int. Reinder and Caribou Sympo. J.R. Luick, P.C. Lent, D.R. Klein and R.G. White (eds), Univ. Alaska, Fairbanks, 297-307.

Wiggins, G.B. 1978. Trichoptera, p. 147-185. In R.W. Merritt and K.W. Cummins [eds.] *An Introduction to the Aquatic Insects of North America.* Kendall/Hunt Publishing Co., Dubuque.

Wilson, D. and S. Ruff 1999. *The Smithsonian Book of North American Mammals.* Washington: Smithsonian Institution Press. Cited by Bartalucci *et al*. 2000.

APPENDIX E

ANTIMONY ASSESSMENT

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APPENDIX E: ANTIMONY ASSESSMENT

E.1 INTRODUCTION

A screening level human health assessment (SLRA) was performed for the Giant Mine in November 2003 on behalf of DIAND (SENES 2003). This assessment included very cautious assumptions and used literature derived transfer factors. Besides arsenic, the SLRA showed that the estimated intakes of antimony by an adult and child were above the toxicity reference value for antimony indicating that further investigations were necessary. This appendix summarizes the results of a more detailed assessment for antimony using site-specific data.

E.2 SITE-SPECIFIC DATA CONSIDERED IN ASSESSMENT

Table E.2-1 provides a summary of data on antimony levels in the aquatic environment at the Giant Mine site that were considered in this assessment. As seen in the table, the data collected on antimony levels in aquatic plants, benthic invertebrates, fish sediments and water on Baker Creek were obtained between 2001 and 2004 (Dillon 2002a, b; 2004). These data were used to derive site-specific transfer factors.

TABLE E.2-1
FIELD SAMPLING AND MEASUREMENT ON BAKER CREEK

Media	Antimony Concentration	Unit	Sampling Location	Date	References
aquatic vegetation	22.6	mg/kg (dw)	BC-3.1	Oct-01	Dillon 2002a (April)
aquatic vegetation	22.5	mg/kg (dw)	BC-3.2	Oct-01	Dillon 2002a (April)
aquatic vegetation	16.2	mg/kg (dw)	BC-3.3	Oct-01	Dillon 2002a (April)
aquatic vegetation	16.6	mg/kg (dw)	BC-4.1	Oct-01	Dillon 2002a (April)
aquatic vegetation	0.66	mg/kg (dw)	BC-4.2	Oct-01	Dillon 2002a (April)
aquatic vegetation	18	mg/kg (dw)	BC-4.3	Oct-01	Dillon 2002a (April)
benthos	2.8	µg/g (dw)	BC-3	Jun-02	Dillon 2002b (Nov)
benthos	6.2	µg/g (dw)	BC-4	Jun-02	Dillon 2002b (Nov)
benthos	3.25	µg/g (dw)	3	Aug-04	Dillon 2004
benthos	1.05	µg/g (dw)	4	Aug-04	Dillon 2004
benthos	2.05	µg/g (dw)	5	Aug-04	Dillon 2004
benthos	2.39	µg/g (dw)	6	Aug-04	Dillon 2004
benthos	1.95	µg/g (dw)	7	Aug-04	Dillon 2004
fish - longnose sucker	<0.05	mg/kg (dw)	BC4	Jun-02	Dillon 2002b (Nov)
fish - longnose sucker	<0.05	mg/kg (dw)	BC4	Jun-02	Dillon 2002b (Nov)
fish - longnose sucker	<0.05	mg/kg (dw)	BC4	Jun-02	Dillon 2002b (Nov)
fish - longnose sucker	<0.05	mg/kg (dw)	BC4	Jun-02	Dillon 2002b (Nov)
fish - longnose sucker	<0.05	mg/kg (dw)	BC4	Jun-02	Dillon 2002b (Nov)
fish - longnose sucker	<0.05	mg/kg (dw)	BC4	Jun-02	Dillon 2002b (Nov)

TABLE E.2-1 (Cont'd)
FIELD SAMPLING AND MEASUREMENT ON BAKER CREEK

Media	Antimony Concentration	Unit	Sampling Location	Date	References
fish - longnose sucker	<0.05	mg/kg (dw)	BC4	Jun-02	Dillon 2002b (Nov)
fish - northern pike	<0.05	mg/kg (dw)	BC4	Jun-02	Dillon 2002b (Nov)
fish - northern pike	<0.05	mg/kg (dw)	BC4	Jun-02	Dillon 2002b (Nov)
fish - northern pike	<0.05	mg/kg (dw)	BC4	Jun-02	Dillon 2002b (Nov)
sediment	18	mg/kg (dw)	BC-3.1	Oct-01	Dillon 2002a (April)
sediment	12.7	mg/kg (dw)	BC-3.2	Oct-01	Dillon 2002a (April)
sediment	7	mg/kg (dw)	BC-3.3	Oct-01	Dillon 2002a (April)
sediment	19	mg/kg (dw)	BC-4.1	Oct-01	Dillon 2002a (April)
sediment	12.9	mg/kg (dw)	BC-4.2	Oct-01	Dillon 2002a (April)
sediment	12.9	mg/kg (dw)	BC-4.3	Oct-01	Dillon 2002a (April)
sediment	248	mg/kg (dw)	BC-3.1	Jun-02	Dillon 2002b (Nov)
sediment	222	mg/kg (dw)	BC-3.2	Jun-02	Dillon 2002b (Nov)
sediment	210	mg/kg (dw)	BC-3.3	Jun-02	Dillon 2002b (Nov)
sediment	145	mg/kg (dw)	BC-4.1	Jun-02	Dillon 2002b (Nov)
sediment	102	mg/kg (dw)	BC-4.2	Jun-02	Dillon 2002b (Nov)
sediment	7.4	mg/kg (dw)	BC-4.3	Jun-02	Dillon 2002b (Nov)
sediment	728	mg/kg (dw)	2	Aug-04	Dillon 2004
sediment	160	mg/kg (dw)	3	Aug-04	Dillon 2004
sediment	153	mg/kg (dw)	4	Aug-04	Dillon 2004
sediment	21.5	mg/kg (dw)	5	Aug-04	Dillon 2004
sediment	90.5	mg/kg (dw)	6	Aug-04	Dillon 2004
sediment	97.6	mg/kg (dw)	7	Aug-04	Dillon 2004
water	25	µg/L	BC-3	Oct-01	Dillon 2002a (April)
water	23	µg/L	BC-4	Oct-01	Dillon 2002a (April)
water	1.8	µg/L	BC-3	Jun-02	Dillon 2002b (Nov)
water	6.3	µg/L	BC-4	Jun-02	Dillon 2002b (Nov)
water	103	µg/L	3	Aug-04	Dillon 2004
water	222	µg/L	4	Aug-04	Dillon 2004
water	213	µg/L	5	Aug-04	Dillon 2004
water	203	µg/L	6	Aug-04	Dillon 2004
water	112	µg/L	7	Aug-04	Dillon 2004

The antimony and concentrations assumed to be present in each of the environmental media considered in the assessment are provided in Table E.2-2. These concentrations were input into the conservative pathways analysis used in the screening level assessment (SENES 2003). In the current analysis, an adjustment was made to account for soil being available for contact for only half the year (i.e. the fact that the ground is frozen and snow covered for over six months of the year was taken into account).

TABLE E.2-2
SUMMARY OF MEASURED AND ASSUMED DATA CONSIDERED IN ASSESSMENT

Media	Concentration	Comment
Water		
- Baker Creek	0.12 mg/L	Estimated worst-case
- Back Bay	0.003 mg/L	Estimated worst-case
Soil		
- Giant Mine site	127.7 mg/kg	95th percentile on-site
- Latham Island	69.6 mg/kg	95th percentile from ESG - RMC 2001
Sediment		
- Baker Creek	357 mg/kg	95th percentile, Jackson *et al*. 1996, Golder 2001, Mace 1998
- Back Bay	33 mg/kg	max, Jackson *et al*. 1996, Golder 2001, Mace 1998
Aquatic plant		
- Baker Creek	15.7 mg/kg (ww)	Estimated from water concentration and site-specific transfer factor based on Dillon (maximum in 2001 was 4.5 mg/kg ww, 22.6 mg/kg dw)
- Back Bay	0.46 mg/kg (ww)	Estimated from site-specific transfer factor
Benthic invertebrate		
- Baker Creek	8.9 mg/kg (ww)	Estimated from water concentration and site-specific transfer factor based on Dillon (maximum in 2002 was 1.2 mg/kg ww, 6.2 mg/kg dw)
- Back Bay	0.36 mg/kg (ww)	Estimated from site-specific transfer factor
Fish	<0.05 mg/kg (dw)	Dillon 2002 – antimony not detected in 10 samples collected from downstream locations in Baker Creek
Berry		
- Giant mine site	0.01 mg/kg (ww)	Estimated from soil concentration and transfer factor
- Latham Island	0.0056 mg/kg (ww)	Estimated from soil concentration and transfer factor
Lichen	0	No air
Forage	25.5 mg/kg (ww)	Giant Mine – estimated from soil concentration and transfer factor
Browse	25.5 mg/kg (ww)	Giant Mine – estimated from soil concentration and transfer factor
Hare	8.6×10^{-3} mg/kg (ww)	Giant Mine - estimated from intake of water, browse, forage and soil – assumed to be in study area year round
Grouse	1.31 mg/kg (ww)	Giant Mine - estimated from intake of water, browse, berries and soil – assumed to be in study area year round
Moose	0.63 mg/kg (ww)	Giant Mine - estimated from intake of water, browse, aquatic vegetation and sediment – assumed to be in study area year round
Caribou	2.7×10^{-3} mg/kg (ww)	Giant Mine - estimated from intake of water, browse, forage, lichen and soil – assumed to be in study area 10% of year
Mallard	0.03 mg/kg (ww)	Back Bay - estimated from intake of water – assumed to be in study area 50% of year

Note: ww – wet weight, dw-dry weight.

E.3 HUMAN HEALTH RISK ASSESSMENT

Table E.3-1 provides a breakdown of the total ingestion intake by pathway for an adult and child receptor at Latham Island. These receptors are assumed to reside at Latham Island, and obtain all of their drinking water, fish and mallard from Back Bay. It is also assumed that these receptors consume hare, moose, caribou, grouse from the Giant Mine site, and spend weekends (3 months of the year) at the Giant Mine site (soil and berries). Table E.3-1 shows that the ingestion of water, moose, grouse and soil are the primary pathways of exposure to the human receptors.

TABLE E.3-1
ANTIMONY INGESTION INTAKE BY PATHWAY

	Adult		Child	
	Ingestion (mg/(kg d))	**% of Total Intake**	**Ingestion (mg/(kg d))**	**% of Total Intake**
Water	7.30×10^{-5}	35.4%	8.36×10^{-5}	23.3%
Fish	2.36×10^{-5}	11.5%	3.76×10^{-5}	10.5%
Hare	8.86×10^{-7}	0.4%	1.41×10^{-6}	0.4%
Moose	5.29×10^{-5}	25.7%	8.42×10^{-5}	23.5%
Caribou	1.17×10^{-5}	5.7%	8.43×10^{-6}	2.4%
Grouse	3.16×10^{-5}	15.3%	5.02×10^{-5}	14.0%
Mallard	1.32×10^{-6}	0.6%	2.10×10^{-6}	0.6%
Berries	7.26×10^{-7}	0.4%	1.15×10^{-6}	0.3%
Soil	1.04×10^{-5}	5.1%	8.96×10^{-5}	25.0%
Total	2.06×10^{-4}		3.58×10^{-4}	

The human health assessment result of long-term exposure to antimony was expressed in terms of hazard quotient for a non-carcinogenic substance. The hazard quotient is defined as the ratio of predicted exposure (ingestion or inhalation dose) to a chronic reference dose (RfD) or long-term toxicity value. For this assessment the Toxicological Reference Value (TRV) used was an oral RfD of 4×10^{-4} mg/(kg d) as recommended by IRIS (U.S. EPA 2005). This TRV is based on changes in blood glucose levels in rats.

In general, regulatory agencies concur that a hazard quotient below one (1) does not represent a significant impact to the receptor, as the predicted exposure does not exceed the applicable benchmark. Health Canada allocates 20% of the Hazard Quotient for each exposure pathway. Additionally, Cadwell *et al*. (1993) suggests a value of 0.3, based on a conservative approach designed to account for chemical synergism. A hazard quotient of 0.5 was used in this assessment as all potentially important pathways were considered other than the contribution of supermarket foods.

A summary of total dose from ingestion and dermal contact pathways, as well as the hazard quotient for the adult and child receptors is provided in Table E.3-2. This table shows that the HQ is above the comparison point of 0.5 for both receptors.

TABLE E.3-2
SUMMARY OF TOTAL DOSE AND HAZARD QUOTIENT

	Ingestion	Dermal	Total Dose	Antimony Oral Toxicity Data	Reference of Toxicity Value	Hazard Quotient
	(mg/(kg d))	(mg/(kg d))	(mg/(kg d))	(mg/(kg d))		
Adult	$2.06x10^{-4}$	$9.65x10^{-6}$	$2.16x10^{-4}$	4×10^{-4}	IRIS	0.54
Child	$3.58x10^{-4}$	$1.17x10^{-5}$	$3.70x10^{-4}$	4×10^{-4}	IRIS	0.93

Note: IRIS – Integrated Risk Information System (U.S. EPA 2005)

The assumptions regarding the soil concentration at the Giant Mine site and Latham Island have a significant impact on the results. Antimony and arsenic occur together at the site, as seen in Table E.3-3 and Figure E.3-1.

TABLE E.3-3
SOIL CONCENTRATION DATA FOR ARSENIC AND ANTIMONY

Easting	Northing	Lab ID	Sample ID	Depth (m)	Date Sampled	Antimony (mg/kg)	Arsenic (Total) (mg/kg)
TABLE 3-2 AREA 1 (Mill and C-Shaft area)							
636319	6932883	L14671-1	PSO-01-2200-01	0.2	22-Jul-00	1.6	841
636390	6932985	L14671-2	PSO-02-2200-01	0.2	22-Jul-00	0.2	192
635932	6932428	L14671-3	PSO-11-2200-01	0.2	22-Jul-00	1.5	2440
635963	6932392	L14671-4	PSO-12-2200-01	0.2	22-Jul-00	2.5	3760
636012	6932595	L14671-44	PSO-13-2500-01	0.2	25-Jul-00	13	1970
636012	6932595	L14671-15	PSO-13-2500-01A	0.2	25-Jul-00	10.5	2370
635965	6932654	L14671-5	PSO-14-2200-01	0.2	22-Jul-00	27.7	6160
635965	6932654	L14671-6	PSO-14-2200-02	0.4	22-Jul-00	1.7	943
635838	6932567	L14671-7	PSO-15-2200-01	0.2	22-Jul-00	0.9	1830
635993	6932824	L14671-8	PSO-16-2200-01	0.2	22-Jul-00	0.5	753
636109	6932389	L14671-9	PSO-17-2200-01	0.15	22-Jul-00	3.4	1600

TABLE E.3-3 (Cont'd)
SOIL CONCENTRATION DATA FOR ARSENIC AND ANTIMONY

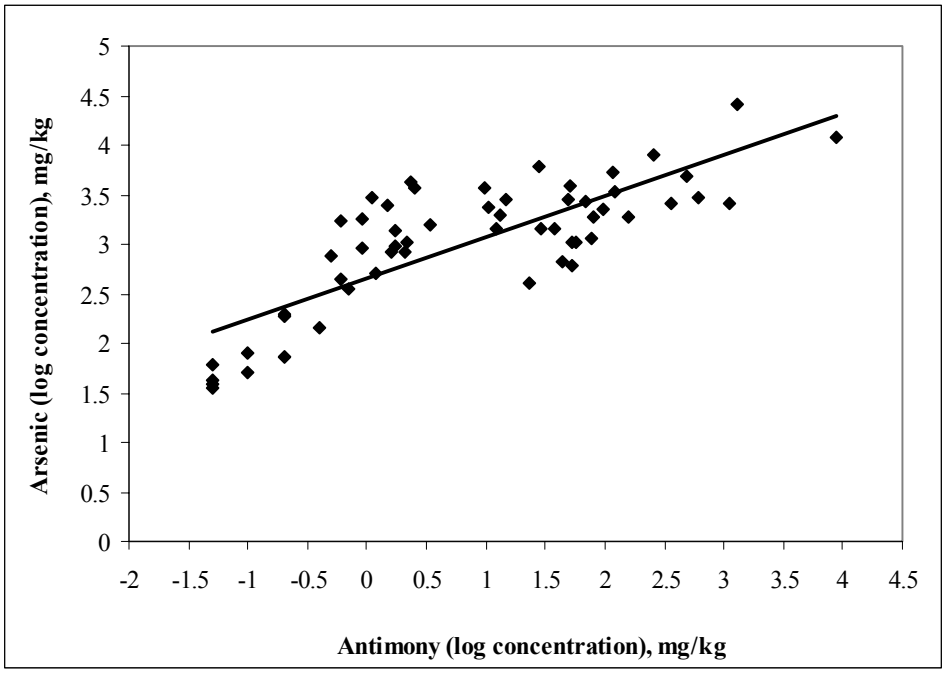
Easting	Northing	Lab ID	Sample ID	Depth (m)	Date Sampled	Antimony (mg/kg)	Arsenic (Total) (mg/kg)
636160	6932711	L14671-10	PSO-18-2200-02	0.2	22-Jul-00	1.7	1390
636160	6932711	L14671-11	PSO-18-2200-03	0.4	22-Jul-00	0.9	919
636111	6932792	L14671-12	PSO-19-2200-01	0.2	22-Jul-00	0.6	1720
636105	6932953	L14671-13	PSO-20-2200-01	0.2	22-Jul-00	0.2	199
636052	6932800	L14671-16	PSO-21-2500-01	0.2	25-Jul-00	2.3	4280
636052	6932800	L14671-18	PSO-21-2500-01A	0.2	25-Jul-00	9.7	3650
636052	6932800	L14671-17	PSO-21-2500-02	0.4	25-Jul-00	1.1	2910
636059	6932603		PS04-1-20-75	0.75	22-Jun-04	254	8040
636342	6932938		PS04-1-37-20	0.2	22-Jun-04	51	3930
636255	6932802		PS04-1-38-20	0.2	22-Jun-04	38	1410
636159	6932653		PS04-1-39-20	0.2	22-Jun-04	29	1440
635885	6932410		PS04-1-48-20	0.2	23-Jun-04	52	1070
635885	6932410		PS04-1-48-20	0.2	23-Jun-04	41	-
635885	6932410		PS04-1-48-75	0.75	23-Jun-04	49	2800
635885	6932410		PS04-1-48-75	0.75	23-Jun-04	58	-
635943	6932449		PS04-1-49-75	0.75	23-Jun-04	122	3360
635943	6932449		PS04-1-49-75	0.75	23-Jun-04	94	-
635913	6932642		PS04-1-51-75	0.75	23-Jun-04	116	5250
636046	6932862		PS04-1-52-200	2	23-Jun-04	97	2230
TABLE 3-3 AREAS 2, 3 and 4 (West of Central TCA, TRP, Settling Pond)							
636278	6933389	L14671-23	PSO-03-2200-01	0.1	22-Jul-00	14.7	2850
636352	6933522	L14671-24	PSO-05-2200-01	0.2	22-Jul-00	0.1	81.6
636231	6933715	L14671-25	PSO-06-2200-01	0.2	22-Jul-00	0.7	351
636292	6933892	L14671-26	PSO-08-2200-01	0.2	22-Jul-00	2.2	1060
636294	6933810	L14671-27	PSO-09-2200-01	0.2	22-Jul-00	2.1	824
636169	6933906	L14671-28	PSO-10-2200-01	0.2	22-Jul-00	0.1	51.5
636123	6933846		PS04-4-27-20	0.2	22-Jun-04	68	2650
636050	6933795		PS04-4-28-20	0.2	22-Jun-04	1110	2550
636064	6933711		PS04-4-29-20	0.2	22-Jun-04	1270	25500
636223	6933801		PS04-4-34-260	2.6	22-Jun-04	8860	12200
636304	6933786		PS04-4-35-20	0.2	22-Jun-04	357	2550
636274	6933863		PS04-4-36-75	0.75	22-Jun-04	599	2980
TABLE 3-4 AREA 5 (Propane Tank Farm)							
635917	6934254	L14671-29	PSO-30-2300-01	0.2	23-Jul-00	0.4	146
635731	6934287	L14671-30	PSO-31-2300-01	0.2	23-Jul-00	1.2	516
636015	6934351	L14671-31	PSO-32-2300-01	0.2	23-Jul-00	0.6	437

TABLE E.3-3 (Cont'd)
SOIL CONCENTRATION DATA FOR ARSENIC AND ANTIMONY

Easting	Northing	Lab ID	Sample ID	Depth (m)	Date Sampled	Antimony (mg/kg)	Arsenic (Total) (mg/kg)
TABLE 3-5 AREAS 6 and 7 (Townsite and Townsite Road)							
636125	6931513		PS04-6-42-75	0.75	23-Jun-04	23	400
636094	6931448		PS04-6-43-75	0.75	23-Jun-04	56	1050
636094	6931448		PS04-6-43-150	1.5	23-Jun-04	44	662
636038	6931360		PS04-6-45-20	0.2	23-Jun-04	483	4860
635924	6931271		PS04-6-46-20	0.2	23-Jun-04	53	602
635924	6931271		PS04-6-46-75	0.75	23-Jun-04	158	1880
TABLE 3-6 AREA 8 (Downstream of Dam 7 to Yellowknife Bay)							
636763	6932405		PS04-8-16-20	0.2	22-Jun-04	12	1420
636596	6932454		PS04-8-19-20	0.2	22-Jun-04	79	1870
TABLE 3-7 AREA 9 (Ingraham Trail at Yellowknife River)							
637453	6935045	L14671-35	PSO-33-2500-01	0.2	25-Jul-00	<0.1	60.9
637499	6935051	L14671-36	PSO-34-2500-01	0.2	25-Jul-00	0.2	71.6
637499	6935051	L14671-38	PSO-34-2500-01A	0.2	25-Jul-00	<0.1	38.4
637408	6935080	L14671-37	PSO-38-2500-01	0.2	25-Jul-00	0.2	189
TABLE 3-8 AREA 10 (East of Dam 3)							
637440	6934129		PS04-10-08-40	0.4	22-Jul-00	76	1150
TABLE 3-9 (Akaitcho Area)							
636116	6935695	L14671-32	PSO-39-2500-01	0.2	25-Jul-00	<0.1	41.7
636210	6935695	L14671-33	PSO-40-2500-01	0.2	25-Jul-00	<0.1	35.9
636405	6935418	L14671-34	PSO-41-2500-01	0.2	25-Jul-00	0.2	71.7

Source: Golder 2004.

FIGURE E.3-1
RELATIONSHIP OF ARSENIC AND ANTIMONY AT THE GIANT MINE SITE



Remedial work planned for the Giant Mine site to reduce the arsenic levels in soil will therefore result in a decrease in the antimony soil concentrations. It is estimated that antimony concentrations will be reduced to levels below the CCME residential parkland value of 20 mg/kg once the arsenic contaminated soils are remediated.

If it is assumed that the Latham Island receptor does not obtain water from Back Bay, but rather is connected to the municipal water supply and the soil at the Giant Mine site has an average antimony concentration of 20 mg/kg, then the HQs for the adult and child are 0.17 and 0.42 respectively. These values are below the selected comparison point and therefore there are not expected to be any human health issues associated with antimony exposure after remediation.

E.4 ECOLOGICAL RISK

A review was also conducted for potential ecological effects. The ecological receptors were chosen to represent a wide range of exposure and to ensure that the public concerns are addressed.

E.4.1 Aquatic Receptors

The lowest chronic values (Suter and Tsao 1996) EC_{20} (effects concentration for 20% of the population) were used in this assessment.

Fish	1.6 mg/L
Daphnids	5.4 mg/L
Aquatic plants	0.61 mg/L

By comparison the maximum expected antimony concentration is Baker Creek was determined to equal 0.12 mg/L which is well below the toxicity benchmarks identified above. Hence, it was concluded that the antimony level in Baker Creek will not cause adverse effects on aquatic populations.

E.4.2 Terrestrial Receptors

For terrestrial receptors a screening evaluation was completed using available information. The receptors selected in the assessment were either the most exposed or represented a dietary component for humans. A summary of ecological receptors characteristics is provided in Table E.4-1.

TABLE E.4-1
TERRESTRIAL RECEPTORS CHARACTERISTICS

Ecological Receptors	Body Weight (kg)	Fraction of Tme
caribou	105	0.1
moose	600	1
hare	1.4	1
grouse	0.475	1
muskrat	1.2	1

The toxicity data used in the assessment was based on a mouse (with allometric scaling) as discussed in Section 5 of the main document. The Lowest Adverse Effects Level (LOAEL) for a mouse was 1.25 mg/(kg d) based on an endpoint of lifespan (Sample et al.1996). Appropriate toxicity data for avian receptors was not located from the readily available information; therefore, ducks and grouse were not evaluated. The calculated intake concentration, the LOAEL toxicity reference values and screening index values are provided in Table E.4-2.

TABLE E.4-2
INTAKE CONCENTRATION AND SCREENING INDEX VALUES

	Intake (mg/(kg/d))	LOAEL (mg/(kg/d))	SI
Caribou	0.025	0.77	0.03
Moose	1.06	0.69	1.53
Hare	6.13	0.99	6.17
Grouse	3.28	na	na
Muskrat	5.85	1.00	5.83

Note: na – not available,. Shaded values exceed SI benchmark value of 1.

The results show that there is a potential issue for moose, hare and muskrat along Baker Creek. It should be noted that estimated intakes for moose and hare are affected by the soil concentration. As discussed previously, remedial activities planned at the site for arsenic will result in antimony levels being below the CCME residential/parkland value of 20 mg/kg. Thus, after remediation no adverse effects will be expected on moose and hare populations from exposure to antimony. The results of this assessment confirm that with a soil concentration of 20 mg/kg the intakes of moose and hare are below a level of concern. Muskrat have been observed in recent field studies on Baker Creek as described in the main document. The field investigation found higher concentrations of arsenic in muskrats downstream of the mine site than upstream. However, there is no evidence of effects on the muskrat populations from exposure to elevated levels of arsenic and antimony.

E.5 REFERENCES

Cadwell, R.D., B. Parkhurst, W. Warren-Hicks and J. Volosin 1993. *Aquatic Ecological Risk Assessment and Clean-up Goals for Metals Arising from Mining Operations.* pp 61-72. In Bender, E.S. and F.A. Jones (eds.) Applications of Ecological Risk Assessment to Hazardous Waste Site Remediation. Water Environment Federation, Alexandria, Virginia.

Dillon Consulting Limited 2004. *Biological Sampling at Baker Creek 2003 – Technical Report Final.* Indian and Northern Affairs Canada. April 2004. 03-2049.

Dillon Consulting Limited 2002a. *Biological Sampling at Baker Creek – Summary Report.* Department of Indian and Northern Affairs Development. April 2002. 01-9670-3000.

Dillon Consulting Limited 2002b. *Biological Sampling at Baker Creek 2002 –Technical Report-Final.* Prepared for Indian and Northern Affairs Canada. 02-0465. November.

Environmental Sciences Group, Royal Military College 2001. *Arsenic Levels in the Yellowknife Area: Distinguishing Between Natural Background and Anthropogenic Inputs.* Prepared for Yellowknife Arsenic Soil remediation Committee.

Golder Associates 2004. *Soil Concentration Data for Arsenic and Antimony.* Excel Spreasheet: "Report Tables Revision2 (November 26).

Golder Associates 2001. *Final Abandonment and Restoration Plan, Giant Mine.* Submitted to Miramar Giant Mine Ltd. September.

Jackson, F.J., C.N. Lafontaine and J. Klaverkamp 1996. *Yellowknife – Back Bay Study on Metal and Trace Element Contamination of Water, Sediment and Fish.* Department of Indian and Northern Affairs Canada. November.

Mace, I.S. 1998. *A Study of Arsenic Contamination from the Royal Oak Giant Mine, Yellowknife, Northwest Territories,* M.Eng. Thesis. May.

Sample, B.E., D.M. Opresko, G.W. Suter II 1996. Toxicological Benchmarks for Wildlife: 1996 Revision. June 1996.

SENES Consultants Limited 2003. *Human Health Screening Level Risk Assessment For Anvil Range Mine Sites.* Prepared for Department of Indian Affairs and Northern Development. October.

Suter II, G. W. and C. L. Tsao 1996. *Toxicological Benchmarks for Screening Potential Contaminants of Concern for Effects on Aquatic Biota: 1996 Revision.* ES/ER/TM-96/R2 Department of Energy, Oak Ridge National Laboratory. June.

United States Environmental Protection Agency (U.S. EPA) IRIS 2005. *Integrated Risk Information System - IRIS.* On-line database. http://www.epa.gov/iris/.

APPENDIX F

RESULTS OF NO REMEDIATION (BASE CASE) ASSESSMENT

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APPENDIX F RESULTS OF NO REMEDIATION (BASE CASE) ASSESSMENT

The water quality and pathways model described previously in this report was also used to assess the impact of a *"No Remediation (Base Case)"* scenario where it is assumed that the site was not remediated and pumping and treatment of mine water is terminated. This scenario is presented for illustration purposes only of the potential consequences of doing nothing at the Giant Mine site. It is stressed that the *"No Remediation"* scenario is not considered to be a potential option for the site, but rather has been assessed for illustrative purposes of the consequences of doing nothing.

The loads used in the assessment were provided by SRK and are summarized in Table F.1-0.

TABLE F.1-0
SUMMARY OF LOADS FOR NO REMEDIATION (BASE CASE) SCENARIO

	Current	Future	
From treatment plant to Baker Creek	290		kg/y
Upstream load to Baker Creek	220	220	kg/y
Tributary load to Baker Creek	70	70	kg/y
Surface load to Baker Creek	220	220	kg/y
Underground mine to Baker Creek		7100	kg/y
Surface load to Yellowknife Bay	110	110	kg/y
Total to Baker Creek	800	7610	
Total to Yellowknife Bay	110	110	
Total	910	7720	

In addition to these loads to the aquatic environment, it was assumed that no remediation of soils at the Giant Mine site or the Townsite would be undertaken. Existing on-site soil concentrations were used in the pathways model.

F.1 WATER AND SEDIMENT QUALITY

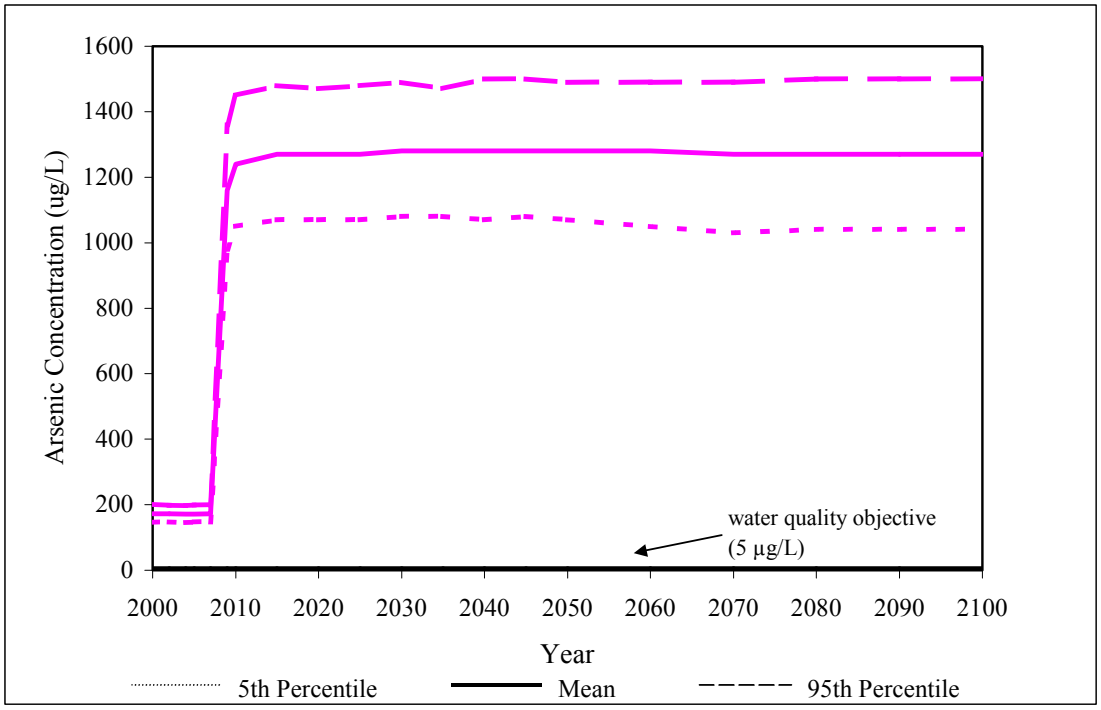
The prediction of water and sediment quality impacts was carried out using the lake dispersion model described in Appendix B. The model was run probabilistically for 100 trials to account for uncertainty and/or natural variability in many of the model input parameters. The simulations were run in one-year time steps commencing in 1950 to account for the effects of past operations on environmental levels of arsenic and continuing until 2100.

F.1.1 Water Quality Predictions

Baker Creek System

The predicted mean, 5th and 95th percentile arsenic levels in Baker Creek over the 2000 through 2100 period are shown in Figure F.1-1 for the No Remediation scenario. As shown, the arsenic level is seen to increase sharply and significantly between 2005 and 2010 then continue at this elevated concentration for the remainder of the simulation period. This sharp rise is a result of stopping treatment and allowing the load from the underground mine enter Baker Creek. For assessment purposes it was assumed that untreated mine water would flow into Baker Creek starting in 2005. In reality, it would take many years for the mine to flood before untreated mine water would reach Baker Creek.

FIGURE F.1-1
PREDICTED ARSENIC LEVELS – BAKER CREEK WATER
NO REMEDIATION SCENARIO



The predicted mean, 5th and 95th percentile arsenic concentrations for Baker Creek in 2010 are presented in Table F.1-1. As seen in both Figure F.1-1 and Table F.1-1, the predicted arsenic concentrations substantially exceed both the Canadian environmental quality guidelines for the protection of aquatic life (5 µg/L) (CCME 2002) and the new proposed guideline for drinking water (5 µg/L) (Federal-Provincial-Territorial Committee on Drinking Water, 2004).

TABLE F.1-1
PREDICTED ARSENIC CONCENTRATIONS IN THE FOUR STUDY SEGMENTS IN YEAR 2010 FOR THE NO REMEDIATION BASE CASE

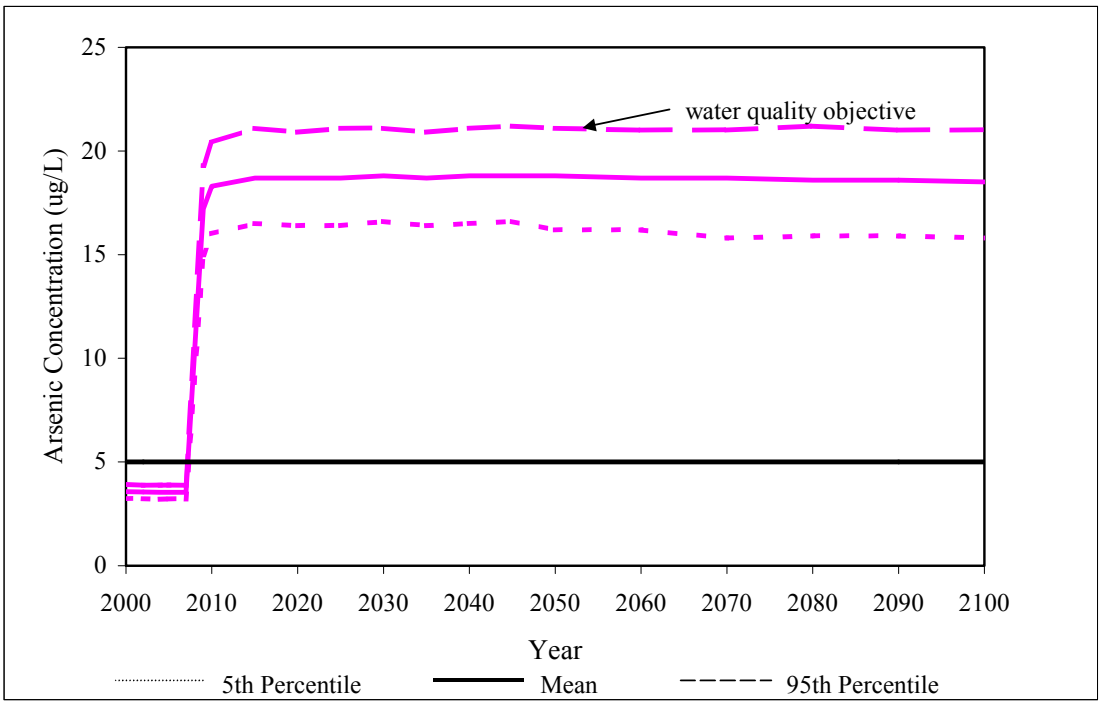
Study Segment	Water Arsenic Concentrations (µg/L)		
	5th Percentile	Mean	95th Percentile
Baker Creek	1040	1240	1440
Back Bay	16.1	18.3	20.5
North Yellowknife Bay	6.3	7.1	8.1
South Yellowknife Bay	1.7	1.9	2.2
CCME (2002) Guideline	5		
Federal-Provincial-Territorial Committee on Drinking Water (2004) proposed guideline	5		

Back Bay

The predicted arsenic levels in Back Bay for the No Remediation scenario is provided in Figure F.1-2. As shown in this figure, a sharp increase in the arsenic level between 2005 and 2010 is visible, reflecting the loading to Baker Creek from the underground mine, which was assumed to occur in 2005.

The predicted mean, 5th and 95th percentile values for Back Bay in 2010 are presented in Table F.1-1. As evidenced by the data presented on this table, as well as the data plotted in Figure 6.1-2, the arsenic concentration in the water column of Back Bay exceeds the proposed drinking water quality guideline of 5 µg/L and the surface water quality guideline of 5 µg/L for the protection of aquatic life by up to a factor of 4.

FIGURE F.1-2
PREDICTED ARSENIC LEVELS – BACK BAY WATER
NO REMEDIATION SCENARIO

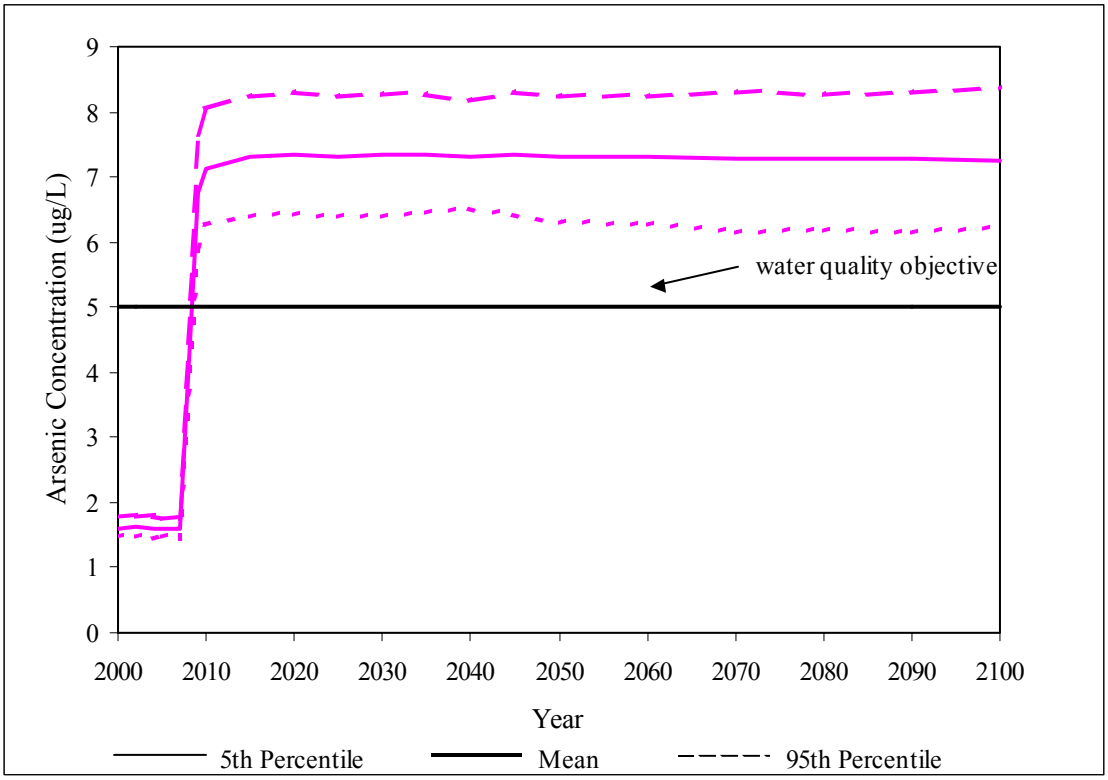


North Yellowknife Bay

The predicted arsenic levels in North Yellowknife Bay water are plotted in Figure F.1-3. A rise in the arsenic level in North Yellowknife Bay can be seen in the 2005 to 2010 period and then the concentration is predicted remain at this new level. The predicted mean, 5th and 95th percentile values for North Yellowknife Bay in 2010 are presented in Table F.1-1.

The predicted arsenic concentrations in the water column of North Yellowknife Bay for the No Remediation scenario are above both the proposed drinking water quality guideline and the current surface water quality guideline of 5 µg/L.

FIGURE F.1-3
PREDICTED ARSENIC LEVELS – NORTH YELLOWKNIFE BAY WATER
NO REMEDIATION SCENARIO

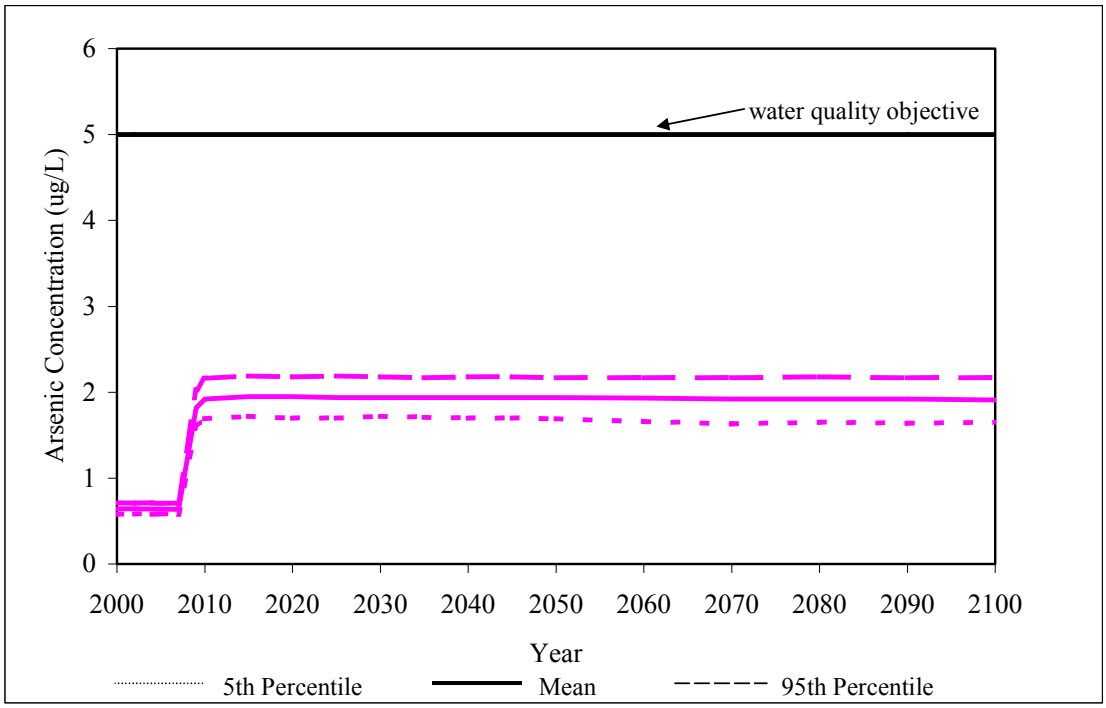


South Yellowknife Bay

The water quality predictions for arsenic in South Yellowknife Bay surface water are provided in Figure F.1-4. As seen from this figure, a slight rise in the arsenic concentration is predicted in South Yellowknife Bay.

The predicted mean, 5th and 95th percentile values for South Yellowknife Bay in 2010 are presented in Table F.1-1. The arsenic concentration in the water column in South Yellowknife Bay remain below the proposed drinking water guideline and the current surface water quality guideline. However these concentrations are higher than the baseline level of 0.3 µg/L measured in the Yellowknife River.

FIGURE F.1-4
PREDICTED ARSENIC LEVELS – SOUTH YELLOWKNIFE BAY WATER
NO REMEDIATION SCENARIO



F.1.2 Sediment Quality Predictions

Baker Creek System

Figure F.1-5 shows the predicted sediment concentrations in Baker Creek for the No Remediation scenario. Over time, a decreasing trend in arsenic concentrations in sediment is evident from the figure. Sediments in Baker Creek had an average arsenic content of approximately 2,340 µg/g in the early 1990's. In this no remediation scenario the arsenic concentrations in the creek sediments are expected to decrease as a result of arsenic diffusion from the sediment to the water phase.

FIGURE F.1-5
PREDICTED ARSENIC LEVELS – BAKER CREEK SEDIMENT
NO REMEDIATION SCENARIO

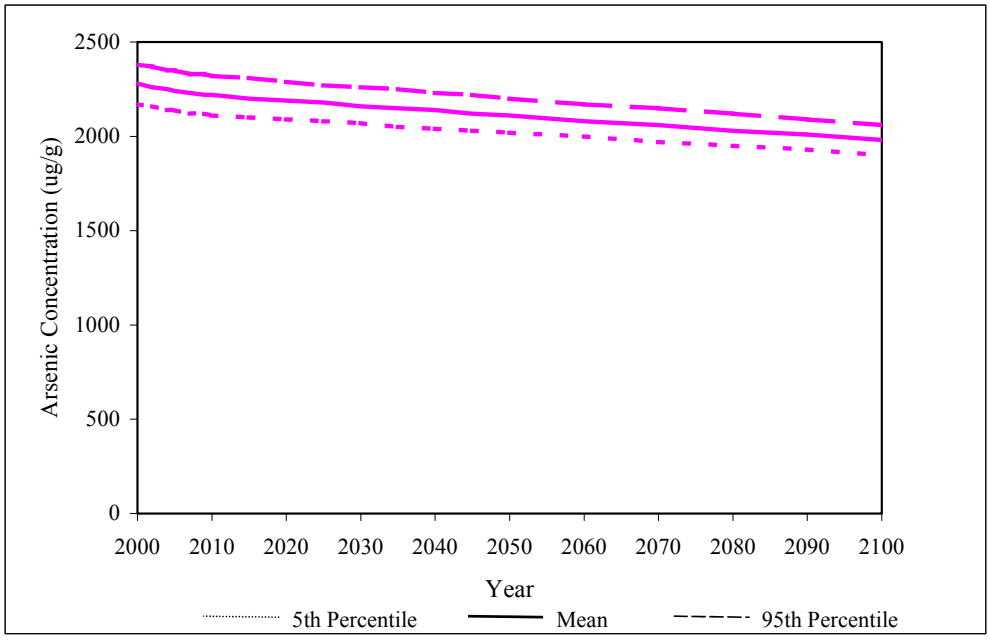


Table F.1-2 compares predicted mean sediment concentrations in 2010 and 2100 to toxicity reference values published by several different authors (discussed in Section 5.0 of the main report) for the No Remediation scenario. The screening index values presented in the table are the calculated ratios of the predicted sediment concentrations to the respective toxicity reference values, which are shown at the top of each column. A screening index value greater than 1 indicates that the predicted concentration exceeds the benchmark.

TABLE F.1-2
COMPARISON OF PREDICTED (MEAN) ARSENIC CONCENTRATIONS IN BAKER CREEK SEDIMENT TO TOXICITY REFERENCE VALUES

	Sediment Concentration (µg/g)	CCME 1999		Thompson *et al.* 2002		Liber and Sobey[a] 2000			
		TEL	PEL	LEL	SEL	NOEC	LOEC	Growth Effects IC_{25}	Growth Effects IC_{50}
Benchmark Values (µg/g)	-	5.9	17	10	346	-	39	174	342
Screening Index Values									
Base Case – 2010	2220	**376**	**131**	**222**	**6.4**	-	**57**	**13**	**6.5**
Base Case - 2100	1980	**336**	**116**	**198**	**5.7**	-	**51**	**11**	**5.8**

Notes:
TEL – Threshold Effect Level
PEL - Probable Effect Level
LEL - Lowest Effect Level
SEL - Severe Effect Level
ERL - Effects Range Low
ERM - Effects Range Medium
NOEC - No-observed-effect-concentration
LOEC – Lowest-observed-effect-concentration
Screening Index (SI) values equal the ratio of predicted arsenic concentrations to toxicity reference values.
Bold SI values indicate that the predicted concentration exceeds the toxicity reference value.

The results suggest that there is a potential for adverse effects on the benthic community in Baker Creek based on the predicted arsenic concentrations in sediment.

Back Bay

The arsenic concentrations in Back Bay sediment (Figure F.1-6) show a small decreasing trend. Table F.1-3 presents the screening index values calculated for the No Remediation case in years 2010 and 2100, based on the mean predicted concentrations divided by the toxicity reference values shown at the top of each column.

FIGURE F.1-6
PREDICTED ARSENIC LEVELS – BACK BAY SEDIMENT
NO REMEDIATION SCENARIO

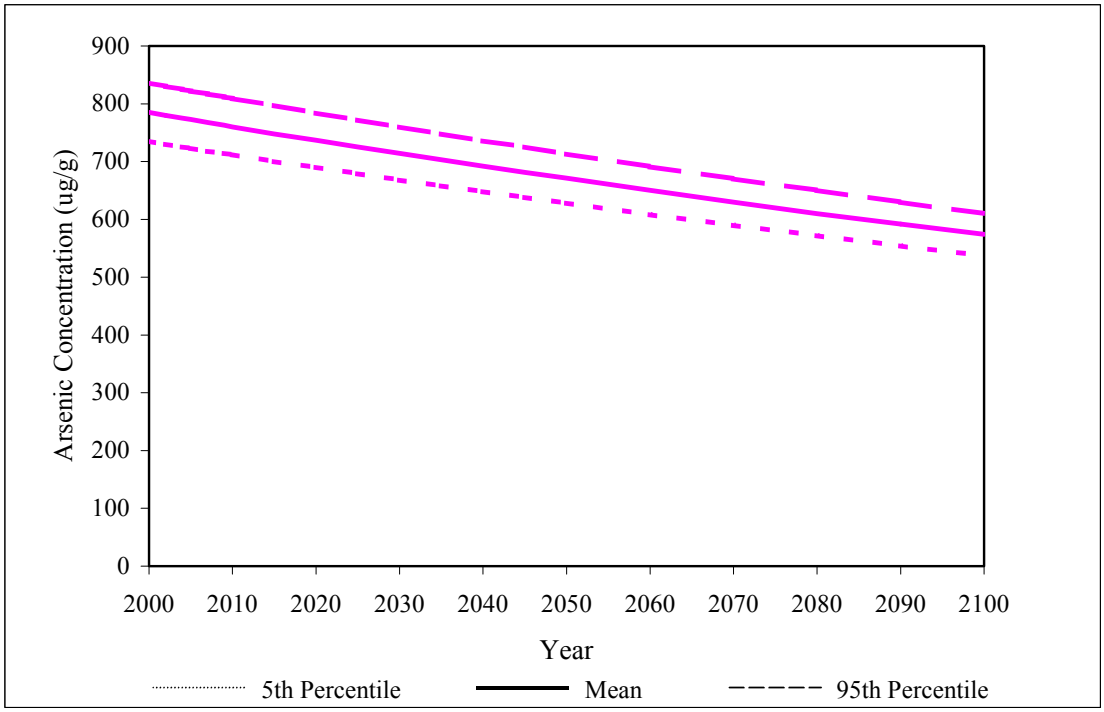


TABLE F.1-3
COMPARISON OF PREDICTED (MEAN) ARSENIC CONCENTRATIONS IN BACK BAY SEDIMENT TO TOXICITY REFERENCE VALUES

	Sediment Concentration (µg/g)	CCME 1999		Thompson *et al.* 2002		Liber and Sobey[a] 2000			
		TEL	PEL	LEL	SEL	NOEC	LOEC	Growth Effects	
								IC_{25}	IC_{50}
Benchmark Values (µg/g)	-	5.9	17	10	346	-	39	174	342
Screening Index Values									
Base Case – 2010	760	**129**	**45**	**76**	**2.2**	**-**	**19**	**4.4**	**2.2**
Base Case - 2100	574	**97**	**34**	**57**	**1.7**	**-**	**15**	**3.3**	**1.7**

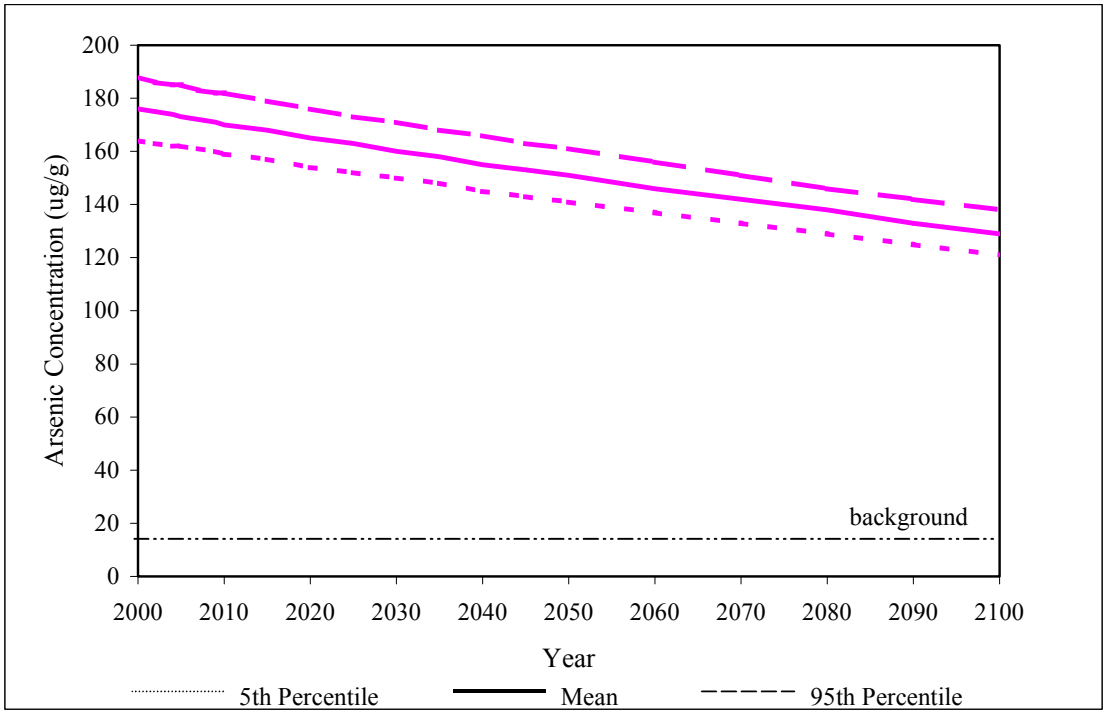
Notes:
LEL - Lowest Effect Level ERM - Effects Range Medium
TEL - Threshold Effect Level SEL - Severe Effect Level NOEC - No-observed-effect-concentration
PEL - Probable Effect Level ERL - Effects Range Low LOEC - Lowest-observed-effect-concentration
Screening Index (SI) values equal the ratio of predicted arsenic concentrations to toxicity reference values.
Bold SI values indicate that the predicted concentration exceeds the toxicity reference value.

The results suggest that there is a potential for adverse effects on the benthic community in Back Bay based on the predicted arsenic concentrations in sediment.

North Yellowknife Bay

The arsenic concentrations in North Yellowknife Bay sediment (Figure F.1-7) show a decreasing trend from the current level for the No Remediation (Base Case) scenario. As discussed for Baker Creek, this is due to lower arsenic loadings than were experienced historically. Table F.1-4 provides a comparison of predicted sediment concentrations in North Yellowknife Bay for the No Remediation scenario for the years 2010 and 2100 to a range of toxicity reference values.

FIGURE F.1-7
PREDICTED ARSENIC LEVELS – NORTH YELLOWKNIFE BAY SEDIMENT
NO REMEDIATION SCENARIO



The comparison, presented as screening index values in Table F.1-4, shows that a number of the sediment toxicity reference values are exceeded by the predicted sediment concentrations in North Yellowknife Bay (i.e. a number of the screening index values are greater than 1). Whether the predicted future sediment arsenic levels will have an affect on the health of the benthic community in North Yellowknife Bay is uncertain. However, it is not unreasonable to expect that as the sediment quality improves over time, if a diverse healthy community does not exist already, the benthic community would improve.

TABLE F.1-4
COMPARISON OF PREDICTED (MEAN) ARSENIC CONCENTRATIONS IN NORTH YELLOWKNIFE BAY SEDIMENT TO TOXICITY REFERENCE VALUES

	Sediment Concentration (µg/g)	CCME 1999		Thompson *et al.* 2002		Liber and Sobey[a] 2000			
		TEL	PEL	LEL	SEL	NOEC	LOEC	Growth Effects	
								IC_{25}	IC_{50}
Benchmark Values (µg/g)	-	5.9	17	10	346	-	39	174	342
Screening Index Values									
Base Case – 2010	170	**29**	**10**	**17**	0.5	-	**4.4**	0.98	0.5
Base Case - 2100	129	**22**	**7.6**	**13**	0.4	-	**3.3**	0.7	0.4

Notes:
LEL - Lowest Effect Level ERM - Effects Range Medium
TEL - Threshold Effect Level SEL - Severe Effect Level NOEC - No-observed-effect-concentration
PEL - Probable Effect Level ERL - Effects Range Low LOEC - Lowest-observed-effect-concentration
Screening Index (SI) values equal the ratio of predicted arsenic concentrations to toxicity reference values.
Bold SI values indicate that the predicted concentration exceeds the toxicity reference value.

South Yellowknife Bay

The arsenic concentrations in the sediment for South Yellowknife Bay are predicted to decline but remain above background in the No Remediation scenario over the simulation period. Figure F.1-8 shows the predicted sediment concentrations over time for South Yellowknife Bay and Table F.1-5 summarizes the screening index values calculated as the ratio of predicted sediment concentrations to the sediment toxicity reference values for the No Remediation scenario.

The background arsenic concentration in lake sediment has been estimated to equal 15 µg/g based on core sample results in Yellowknife Bay, as previously noted. It is noteworthy that this value exceeds the CCME (1999) TEL and the Thompson *et al*. (2002) benchmark values.

FIGURE F.1-8
PREDICTED ARSENIC LEVELS – SOUTH YELLOWKNIFE BAY SEDIMENT
NO REMEDIATION SCENARIO

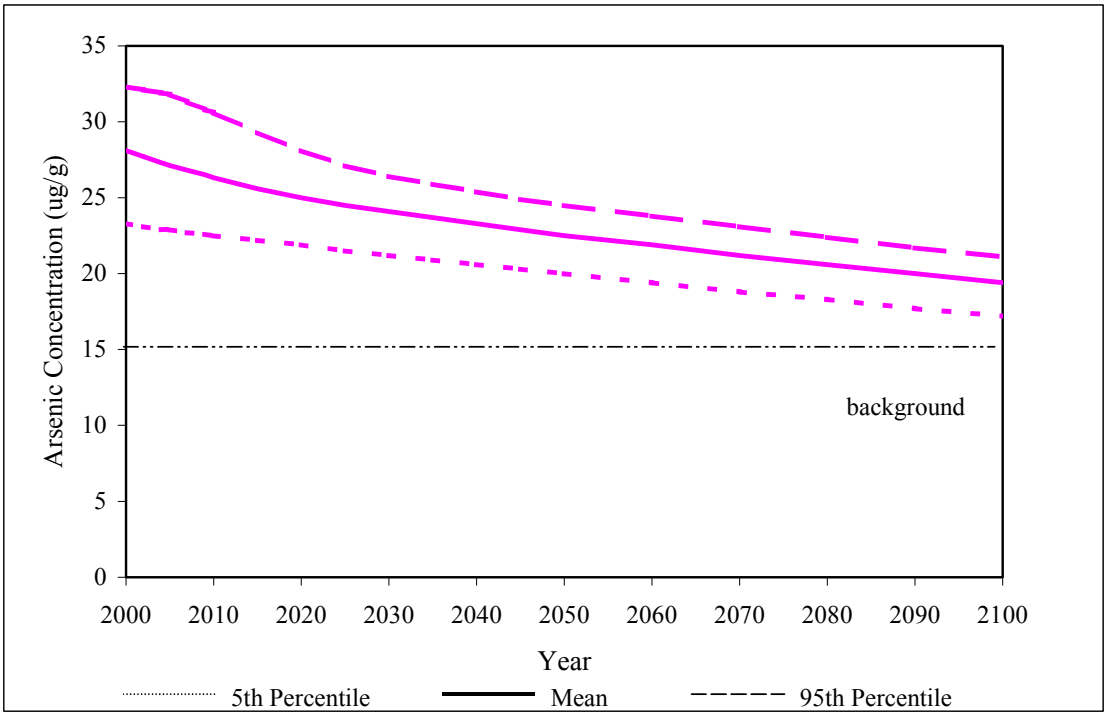


TABLE F.1-5
COMPARISON OF PREDICTED (MEAN) ARSENIC CONCENTRATIONS IN SOUTH YELLOWKNIFE BAY SEDIMENT TO TOXICITY REFERENCE VALUES

	Sediment Concentration (µg/g)	CCME 1999		Thompson *et al.* 2002		Liber and Sobey[a] 2000			
		TEL	PEL	LEL	SEL	NOEC	LOEC	Growth Effects	
								IC_{25}	IC_{50}
Benchmark Values (µg/g)	-	5.9	17	10	346	-	39	174	342
Screening Index Values									
Base Case – 2010	26.3	**4.5**	**1.5**	**2.6**	0.1	-	0.7	0.2	0.1
Base Case - 2100	19.4	**3.3**	**1.1**	**1.9**	0.1	-	0.5	0.1	0.1

Notes:

LEL - Lowest Effect Level ERM - Effects Range Medium

TEL - Threshold Effect Level SEL - Severe Effect Level NOEC - No-observed-effect-concentration

PEL - Probable Effect Level ERL - Effects Range Low LOEC - Lowest-observed-effect-concentration

Screening Index (SI) values equal the ratio of predicted arsenic concentrations to toxicity reference values.

Bold SI values indicate that the predicted concentration exceeds the toxicity reference value.

F.2 ECOLOGICAL ASSESSMENT

The ecological risk assessment for the No Remediation scenario was conducted in the same manner as outlined in the main report. Specifically, the risk assessment evaluated potential adverse effects of arsenic on:

- aquatic receptors (aquatic plants, benthic invertebrates, predatory fish and bottom-feeder fish) based on the predicted arsenic concentrations in surface water; and
- terrestrial receptors (bear, caribou, moose, wolf, snowshoe hare, spruce grouse, mink, muskrat and three types of ducks – common merganser, mallard and scaup) based on estimated exposure to arsenic in water, aquatic food sources, sediment, soil and terrestrial vegetation.

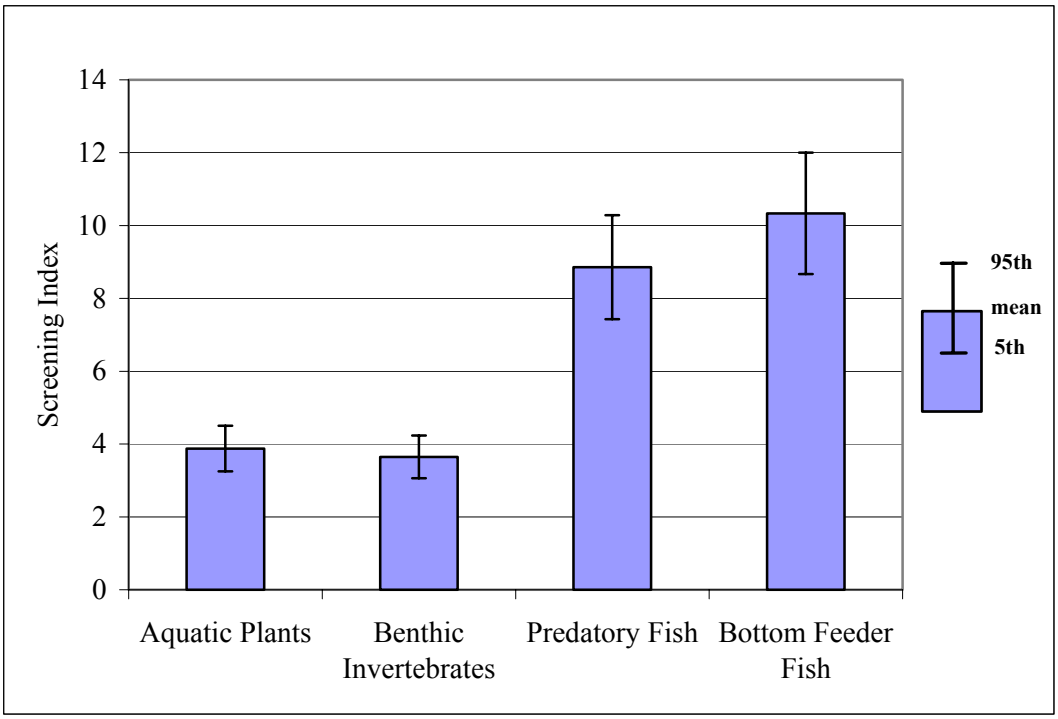
It should be noted that the screening index (SI) values reported in this section are not estimates of the probability of ecological impact. Rather, the index values are positively correlated with the potential of an effect, that is, higher index values imply greater potential of an effect. The benchmark SI value of 1.0 was selected for this assessment, as the calculations of arsenic in water and sediment incorporate background levels in addition to the source contributions.

F.2.1 Aquatic Impacts

The predicted concentrations of arsenic in the waters of Baker Creek, Back Bay and Yellowknife Bay were presented in Section F.1. Tables F.2-1 to F.2-4 summarize the screening index values calculated for the aquatic receptors identified above in Baker Creek, Back Bay and North and South Yellowknife Bay, respectively, for the No Remediation (Base Case) scenario in years 2010 and 2100. The tables show the screening index values based on the predicted mean concentrations, as well as uncertainty bounds in the mean estimates (i.e. the predicted 5^{th} and 95^{th} percentile concentrations). This same information is displayed on Figures F.2-1 to F.2-2 in the form of bar charts for the Baker Creek and Back Bay.

The results for Baker Creek (Table F.2-1 and Figure F.2-1) show that there are potential issues identified for the health of aquatic receptors. The issues are identified throughout the simulation period from 2010 to 2100.

FIGURE F.2-1
SCREENING INDICES FOR AQUATIC SPECIES – BAKER CREEK, 2010
NO REMEDIATION SCENARIO



SCREENING INDICES FOR AQUATIC SPECIES – BAKER CREEK, 2100
NO REMEDIATION SCENARIO

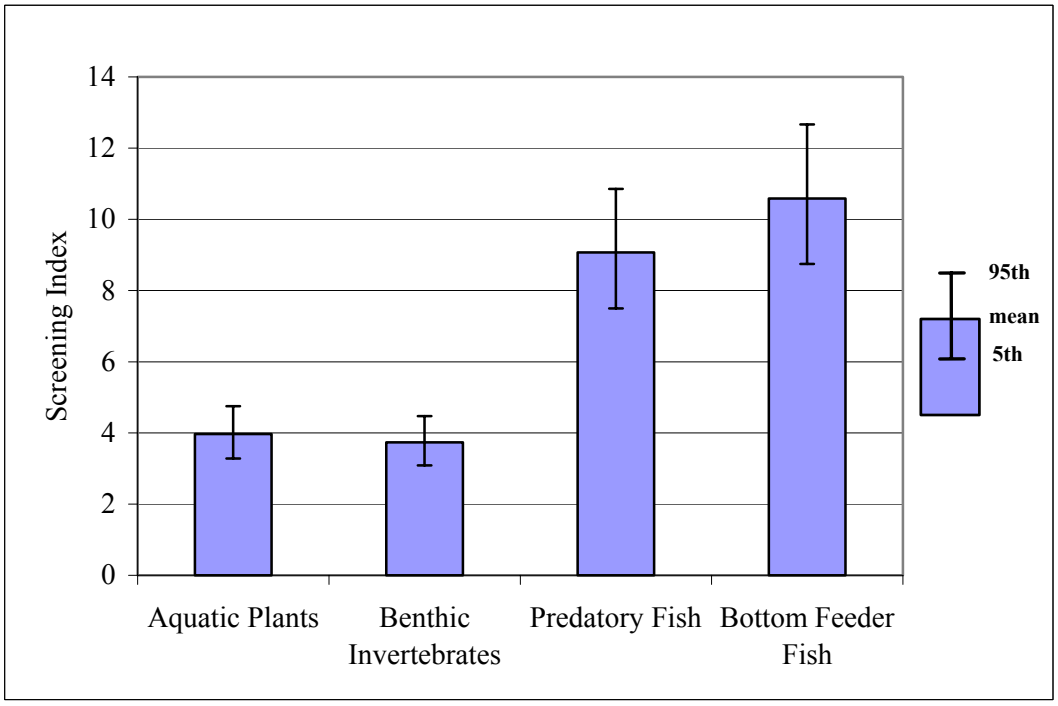
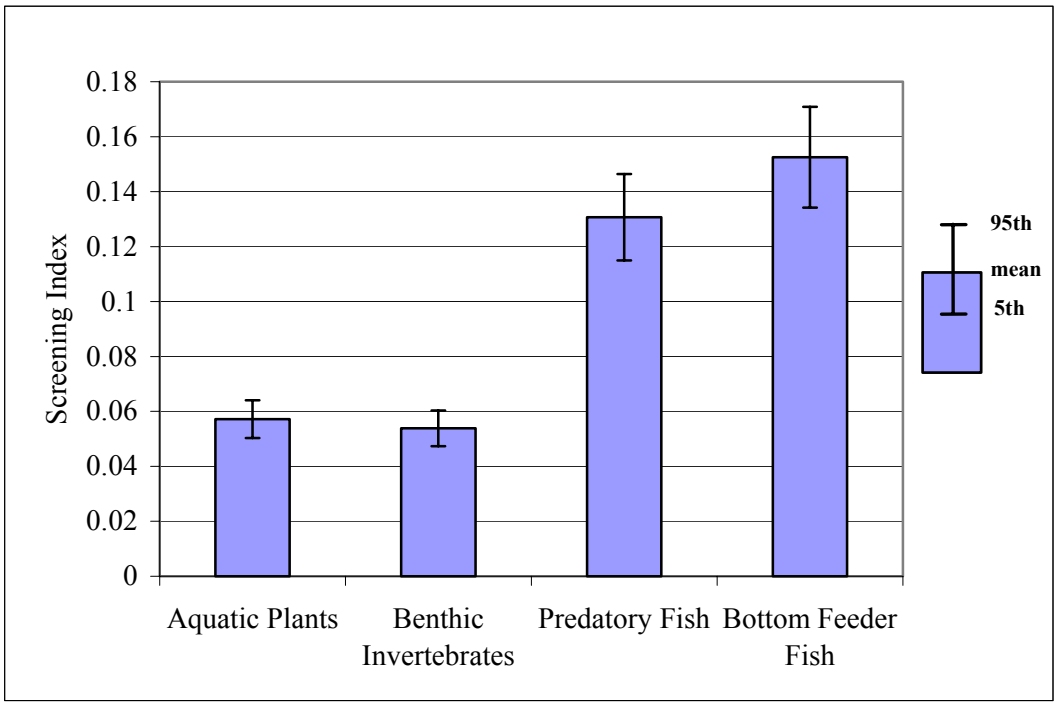
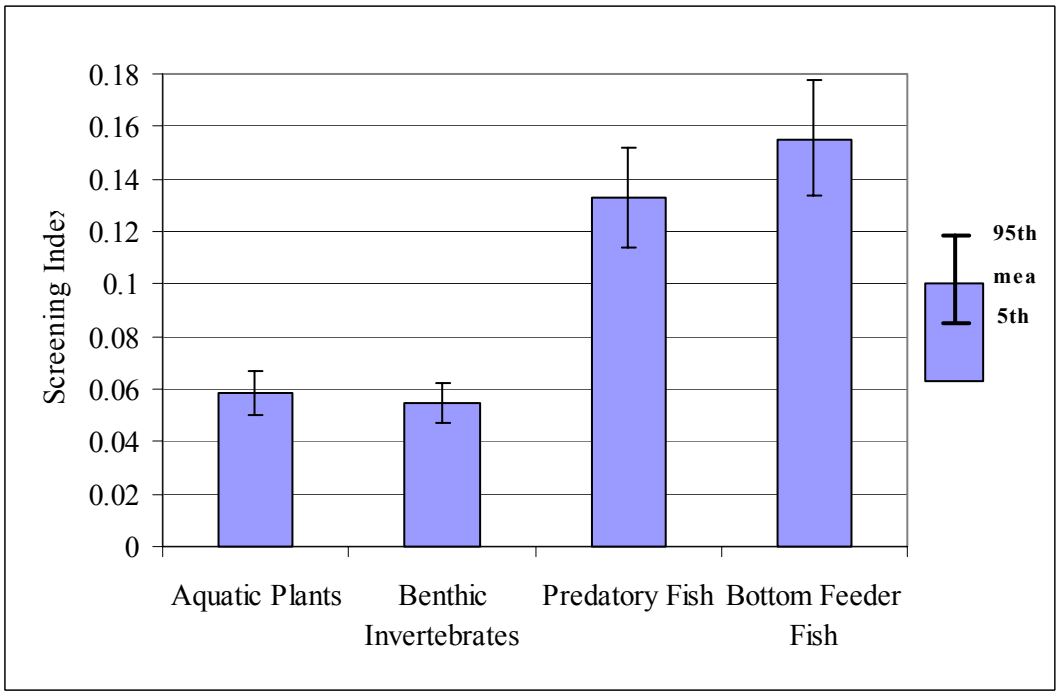


FIGURE F.2-2
SCREENING INDICES FOR AQUATIC SPECIES – BACK BAY, 2010
NO REMEDIATION SCENARIO



SCREENING INDICES FOR AQUATIC SPECIES – BACK BAY, 2100
NO REMEDIATION SCENARIO



In Back Bay (see Table F.2-2 and Figure F.2-2), the screening index values are all below 1 which suggests that none of the aquatic species are expected to be adversely affected at the predicted levels in the water column. Tables F.2-3 and F.2-4 for Yellowknife Bay also show that adverse effects are not expected.

TABLE F.2-1
SCREENING INDEX VALUES FOR AQUATIC SPECIES - BAKER CREEK
NO REMEDIATION SCENARIO

Benchmark Values (mg/L)	**Aquatic Plants**			**Benthic Invertebrates**			**Predatory Fish**			**Bottom-feeding Fish***		
	0.32			0.34			0.19			0.18		
Screening Index Values												
	5th	Mean	95th	5th	Mean	95th	5th	Mean	95th	5th	Mean	95th
2010	**3.3**	**3.9**	**4.5**	**3.1**	**3.6**	**4.3**	**7.5**	**8.9**	**10.4**	**8.8**	**10.3**	**12.1**
2100	**3.3**	**4.0**	**4.7**	**3.1**	**3.7**	**4.4**	**7.4**	**9.1**	**10.7**	**8.7**	**10.6**	**12.5**

Notes: n/a – Not applicable as the habitat in Baker Creek is not conducive to the presence of predatory fish.
* bottom-feeding fish such as lake whitefish and white sucker.
Screening Index (SI) values equal the ratio of predicted arsenic concentrations to toxicity reference values.
Bold SI values indicate that the predicted concentration exceeds the toxicity reference value.

TABLE F.2-2
SCREENING INDEX VALUES FOR AQUATIC SPECIES - BACK BAY
NO REMEDIATION SCENARIO

Benchmark Values (mg/L)	**Aquatic Plants**			**Benthic Invertebrates**			**Predatory Fish***			**Bottom-feeding Fish***		
	0.32			0.34			0.19			0.18		
Screening Index Values												
	5th	Mean	95th	5th	Mean	95th	5th	Mean	95th	5th	Mean	95th
2010	0.05	0.06	0.06	0.05	0.05	0.06	0.11	0.13	0.15	0.13	0.15	0.17
2100	0.05	0.06	0.07	0.05	0.05	0.06	0.11	0.13	0.15	0.13	0.15	0.18

Notes: * predatory fish such as lake trout; bottom-feeding fish such as lake whitefish and white sucker.
Screening Index (SI) values equal the ratio of predicted arsenic concentrations to toxicity reference values.
Bold SI values indicate that the predicted concentration exceeds the toxicity reference value.

TABLE F.2-3
SCREENING INDEX VALUES FOR AQUATIC SPECIES
NORTH YELLOWKNIFE BAY – NO REMEDIATION SCENARIO

Benchmark Values (mg/L)	Aquatic Plants			Benthic Invertebrates			Predatory Fish			Bottom-feeding Fish		
	0.32			0.34			0.19			0.18		
Screening Index Values												
	5th	Mean	95th	5th	Mean	95th	5th	Mean	95th	5th	Mean	95th
2010	0.02	0.02	0.03	0.02	0.02	0.02	0.04	0.05	0.06	0.05	0.06	0.07
2100	0.02	0.02	0.03	0.02	0.02	0.02	0.04	0.05	0.06	0.05	0.06	0.07

Notes: Screening Index (SI) values equal the ratio of predicted arsenic concentrations to toxicity reference values.
Bold SI values indicate that the predicted concentration exceeds the toxicity reference value.

TABLE F.2-4
SCREENING INDEX VALUES FOR AQUATIC SPECIES
SOUTH YELLOWKNIFE BAY - NO REMEDIATION SCENARIO

Benchmark Values (mg/L)	Aquatic Plants			Benthic Invertebrates			Predatory Fish			Bottom-feeding Fish		
	0.32			0.34			0.19			0.18		
Screening Index Values												
	5th	Mean	95th	5th	Mean	95th	5th	Mean	95th	5th	Mean	95th
2010	0.01	0.01	0.01	0.005	0.01	0.01	0.01	0.01	0.02	0.01	0.02	0.02
2100	0.01	0.01	0.01	0.005	0.01	0.01	0.01	0.01	0.02	0.01	0.02	0.02

Notes: Screening Index (SI) values equal the ratio of predicted arsenic concentrations to toxicity reference values.
Bold SI values indicate that the predicted concentration exceeds the toxicity reference value**.**

In summary, the results of the aquatic ecological risk assessment demonstrate that there is the potential that the predicted arsenic levels could adversely impact aquatic species in Baker Creek for the No Remediation scenario. This discharge scenario however, result in arsenic levels which would have an adverse effect on the aquatic communities in Back Bay or Yellowknife Bay.

F.2.2 Terrestrial Species

The potential adverse effects on terrestrial species were evaluated by comparing the intake of arsenic in various terrestrial receptors to lowest observable adverse effects level toxicity values (LOAELs). An exceedance of a LOAEL benchmark indicates the possibility of an adverse effect. In this assessment, it was assumed that all terrestrial receptors other than ducks were located along Baker Creek, which were considered to also be present in Back Bay and Yellowknife Bay. The results of the analysis for terrestrial species, with the exception of ducks, are provided in Figures F.2-3a to F.2-3h for Baker Creek.

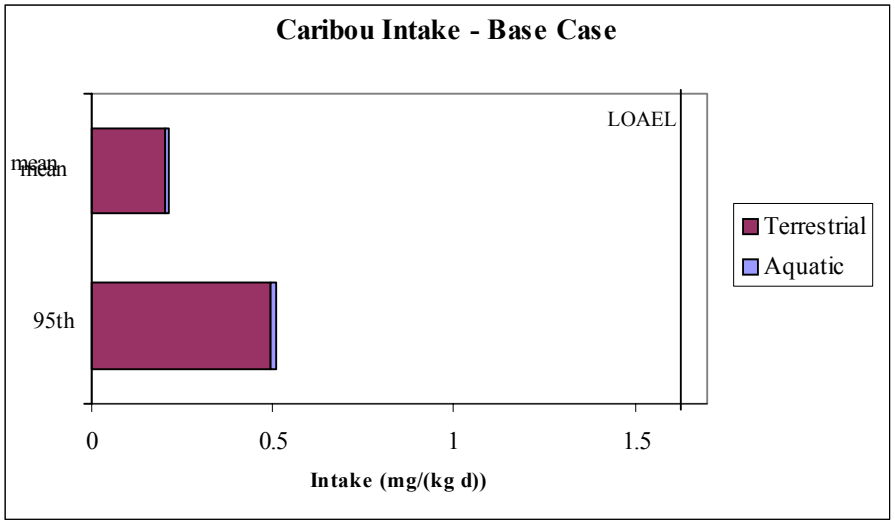
Figures F.2-3a to h provide the estimated arsenic intakes of the given ecological receptors for the No Remediation scenario. The intakes associated with the terrestrial and aquatic components of the animal diets are indicated on the figures to highlight whether aquatic or terrestrial pathways are the main contributors. The figures also provide the toxicity reference values used in the assessment.

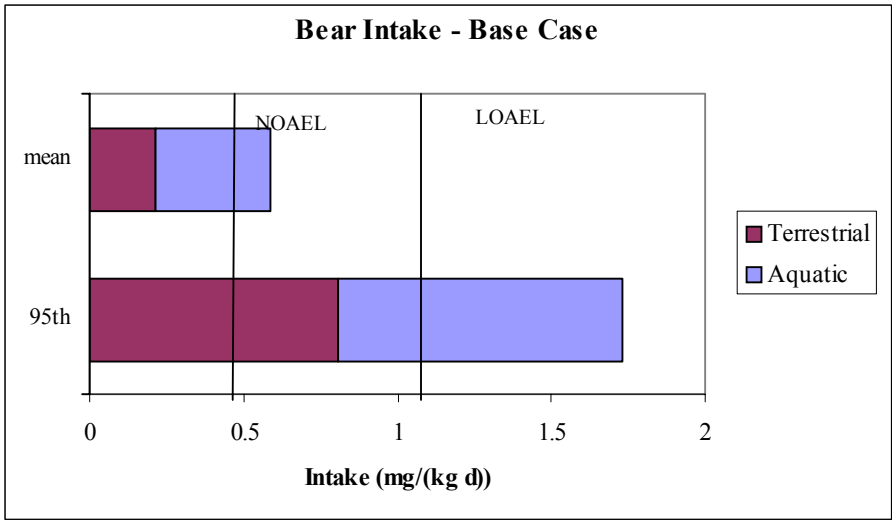
The figures indicate that arsenic in the terrestrial environment contributes to the majority of the estimated intake by caribou, grouse, hare and wolf, whereas the aquatic pathways are relatively minor. Estimated arsenic intakes by caribou and wolf, which were assumed to roam along Baker Creek and obtain their drinking water and food from the downstream watershed for the fraction of the year they are in the study area, are seen to be well below the respective toxicity reference values. It is concluded therefore, that these species are not expected to be adversely affected. For grouse, the mean intake level is predicted to be below the LOAEL however the 95th percentile is above. This suggests that grouse may be affected. The estimated arsenic intake by hare, which were assumed to obtain all their drinking water and vegetation from the Baker Creek watershed area downstream of the Giant Mine, were greater than the LOAEL toxicity reference. These results suggest that hare may potentially be adversely affected. As evident from examination of Figure F.2-3e, ingestion of terrestrial vegetation accounts for the majority of the arsenic intake by hare.

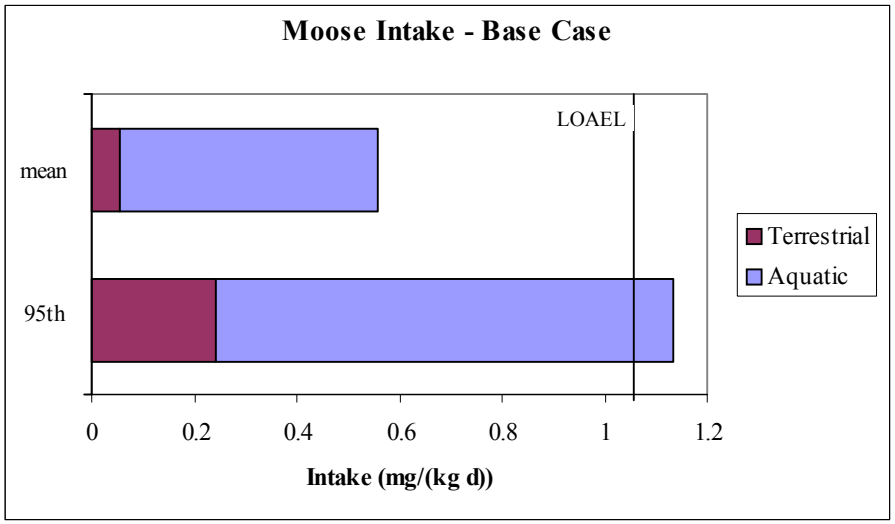
For mink, moose and muskrat, the aquatic pathways dominate the arsenic intakes by each species. For bear both the aquatic and terrestrial pathways are important. The predicted arsenic intake levels for these species indicate that they have the potential to adversely impacted.

Figures F.2-4a to F.2-4c summarize the estimated intakes by ducks on Baker Creek and Back Bay. In each case, the water pathways represent the total arsenic intake. From the figures, it can be seen that the No Remediation scenario results in intakes above the NOAELs and the LOAELs for scaup and merganser in Baker Creek. In Back Bay, the arsenic loadings for do not represent a cause for concern for ducks.

FIGURE F.2-3a,b,c,d
PREDICTED ARSENIC INTAKES BY CARIBOU, MOOSE, BEAR AND WOLF IN 2010







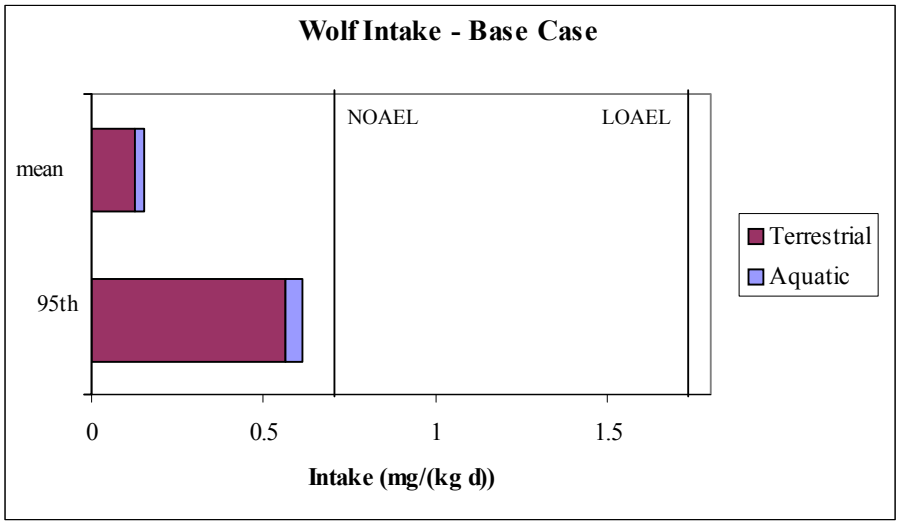


FIGURE F.2-3 e,f,g,h
PREDICTED ARSENIC INTAKES BY HARE, GROUSE, MUSKRAT AND MINK IN 2010

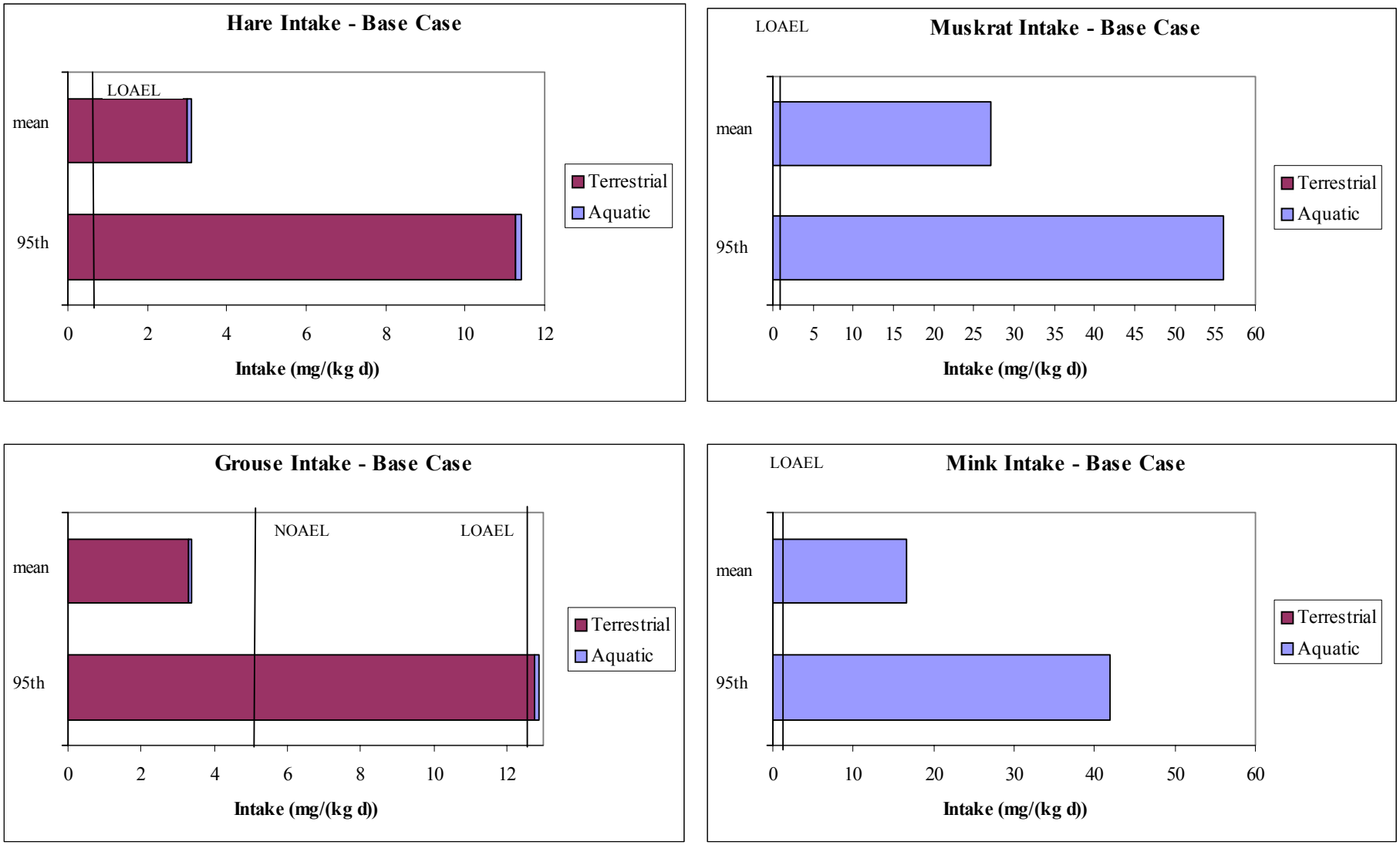
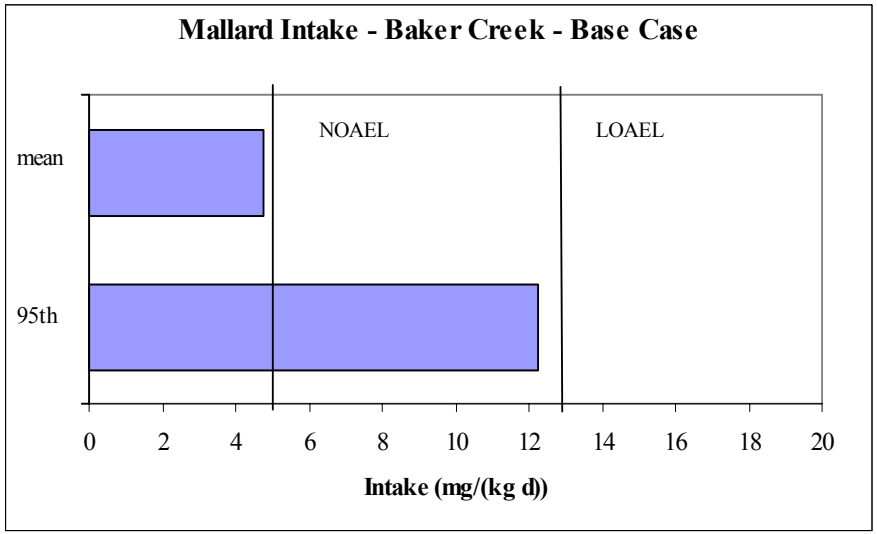
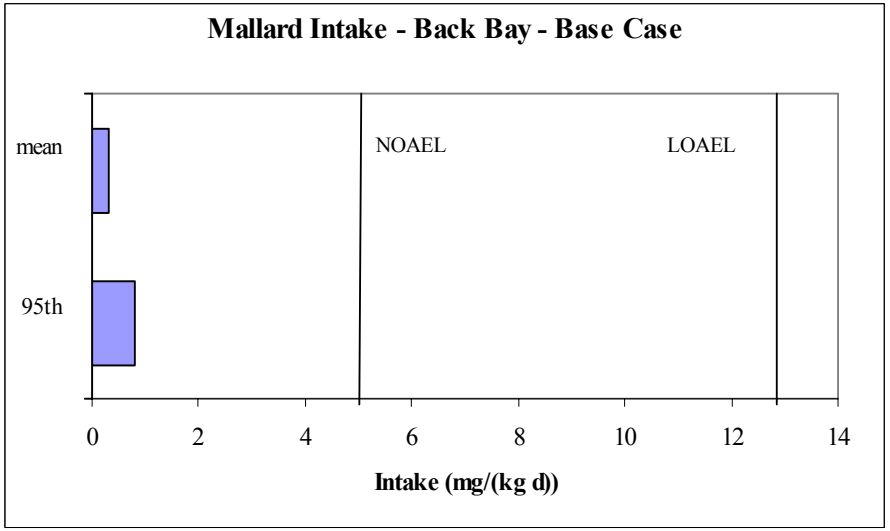
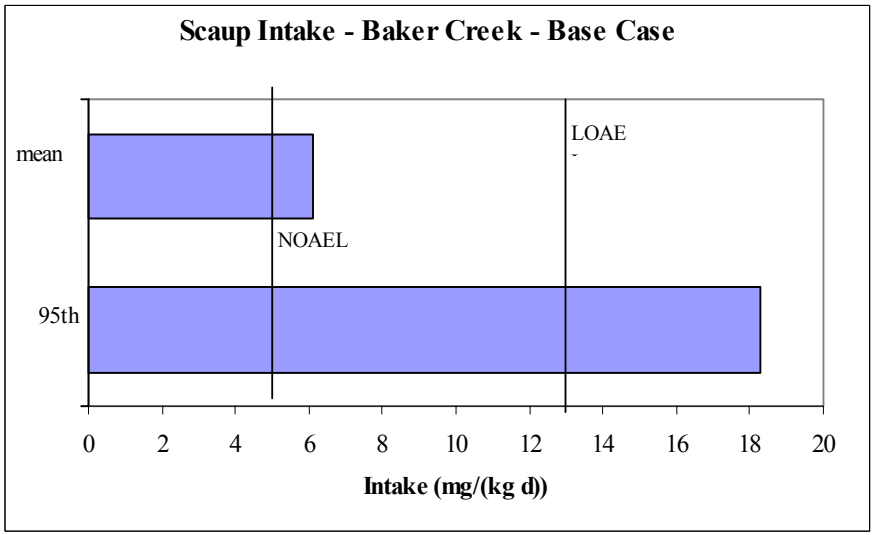


FIGURE F.2-4a,b
COMPARISON OF ARSENIC INTAKES FOR DUCKS IN 2010 TO ARSENIC TOXICITY REFERENCE VALUES







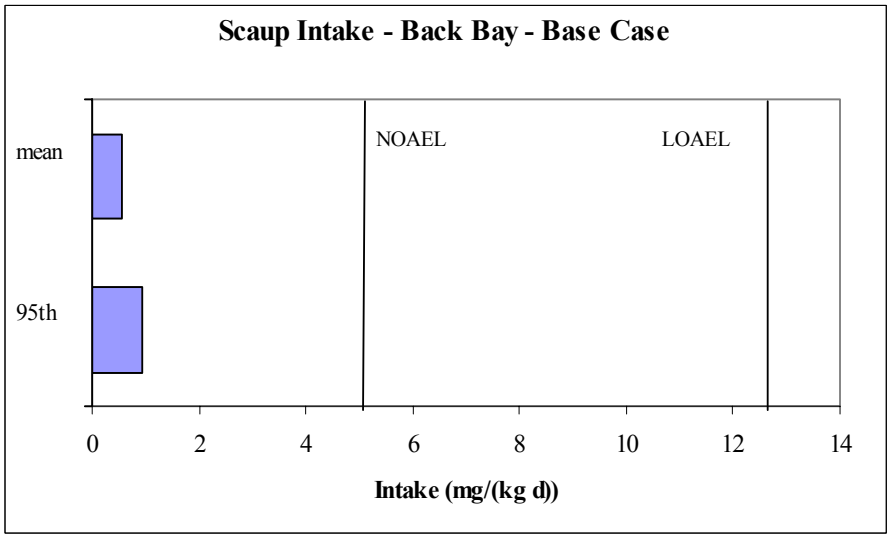
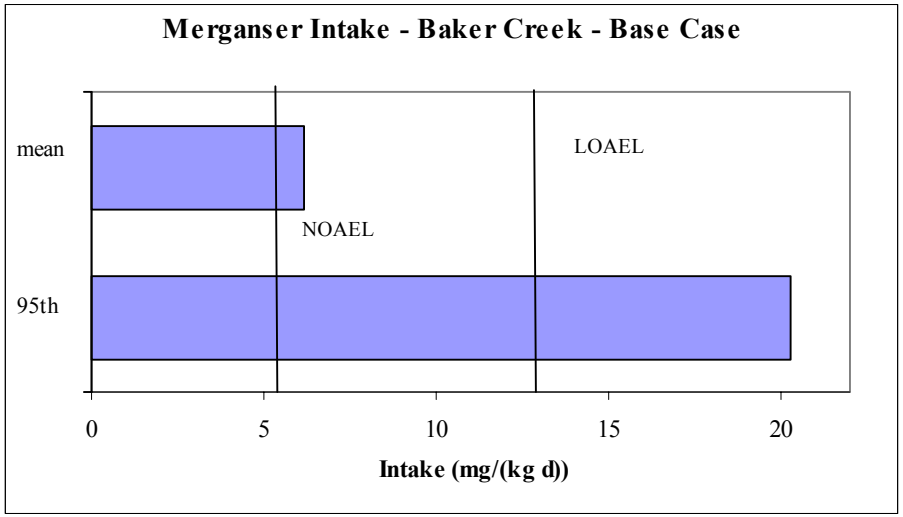
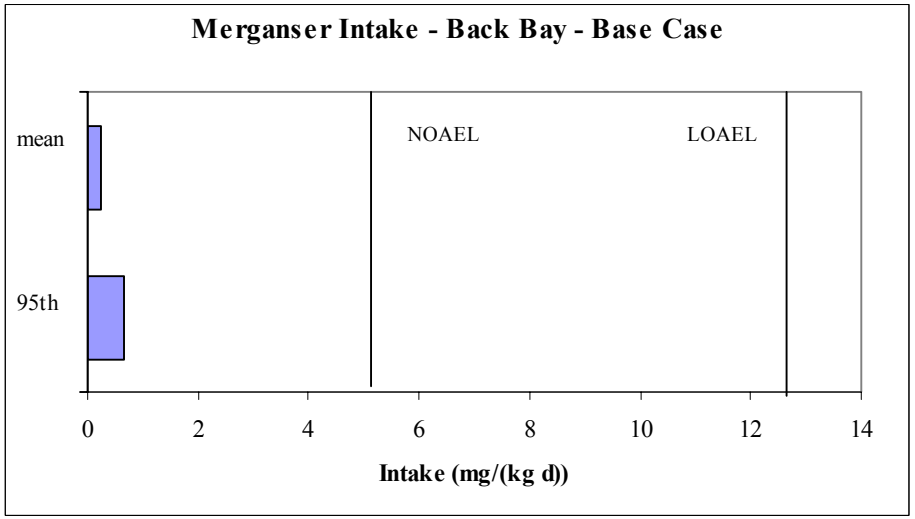


FIGURE F.2-4c
COMPARISON OF ARSENIC INTAKES FOR DUCKS IN 2010 TO ARSENIC TOXICITY REFERENCE VALUES





F.2.3 Overall Ecological Significance

The ecological risk assessment for the No Remediation scenario has demonstrated that aquatic receptors in Baker Creek may be adversely affected as well as terrestrial receptors such as hare and bear that rely on Baker Creek and the Giant Mine site. There are not expected to be any affects on ecological species in Back Bay or Yellowknife Bay.

F.3 HUMAN HEALTH ASSESSMENT

The human health risk assessment for the No Remediation (Base Case) scenario was conducted in the same manner as outlined in the main report. Specifically, the risk assessment evaluated potential adverse effects of arsenic on members of the public in the vicinity of the Giant Mine site. Representative receptors were identified at four locations, as described in Section 3.2 of the main report.

The 2010 estimated mean daily total arsenic intakes (including the contribution from toxic organic and inorganic forms of arsenic) by the eight adult and child receptors living in the study area are presented on Figures F.3-1 to F.3-4. The bar graphs presented on the figure show the contributions of market foods, terrestrial pathways and aquatic pathways to the total estimated daily intakes. The terrestrial pathways bar segments include the contributions from air inhalation, garden produce and berry consumption, caribou, hare and grouse consumption and soil ingestion. The aquatic pathways bar segments include water consumption, fish ingestion and moose and duck consumption. Moose and duck were included in the aquatic pathways as they obtain most of their arsenic intake from consumption of water, aquatic biota and sediments. Also shown on the figures are the typical ranges of daily intakes of inorganic arsenic by adults and children taken from an Environment Canada (1993) study and the Health Canada Toxicity Reference Value (TRV) for arsenic of 0.002 mg/(kg d).

Detailed summaries of the contributions of each pathway to the total daily intake estimates are provided in Table F.3-1 for the No Remediation (Base Case) scenario. The pathways contributions are expressed as daily intake rates in mg/(kg d) (top half of each table) and as a percent of the mean daily intakes (bottom half of tables). The summary statistics presented on the tables include the 5^{th} percentile, median (50^{th} percentile), mean and 95^{th} percentile values.

Receptors 1a and 1c – Giant Mine Townsite Residents

The estimated mean total daily arsenic intakes for Receptors 1a and 1c with a diet similar to that of a typical Canadian, are presented on Figure F.3-1. These receptors were assumed to obtain their drinking water from the municipal supply, and soil, garden produce and berries from the Giant Mine Townsite. It was assumed that ducks and fish were obtained from North

Yellowknife Bay, while large game and small game were assumed to be taken from the Baker Creek watershed to maximize the exposure estimate.

The main pathways of arsenic intake by Receptors 1a and 1c in the No Remediation scenario are associated with consumption of market foods (42 to 44%) and fish (37 to 45%), as demonstrated in Figure F.3-1 and Table F.3.1. In comparison to the arsenic intake range reported for Canadians, the estimated intakes of total arsenic for the No Remediation scenario for Receptor 1 fall above the typical expected range for both the adult and child.

Receptors 2a and 2c – Latham Island Residents

Figure F.3-2 presents the estimated mean total daily arsenic intakes for Receptors 2a and 2c consuming an average fish diet, based on the results of a survey of Dogrib communities that indicated fish consumption of 84 g/d for a typical adult. The dietary intakes by Receptors 2a (adult) and 2c (child) were based on the Receveur *et al.* (1996) survey, which features a high reliance on country foods and low intake of market foods.

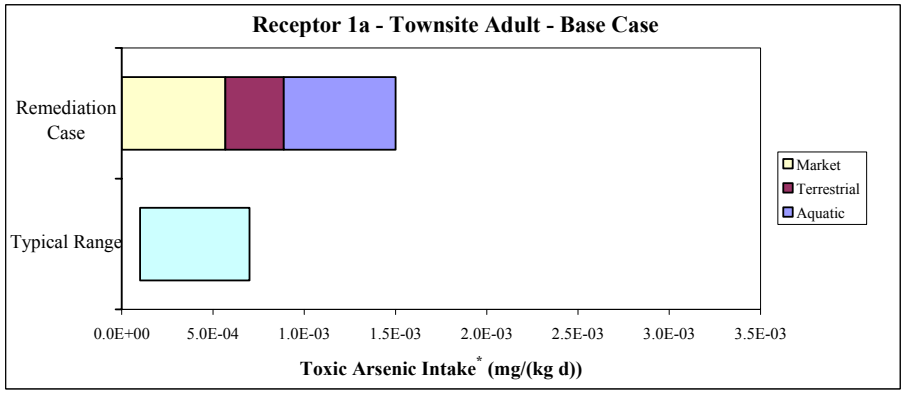
The results plotted on Figure F.3-2 indicate that the estimated total arsenic intakes for adult and children are higher than the reported range of typical exposures for Canadians. For Receptors 2a and 2c, the most important pathway of exposure is consumption of fish which contributes to over 80% of the total intake.

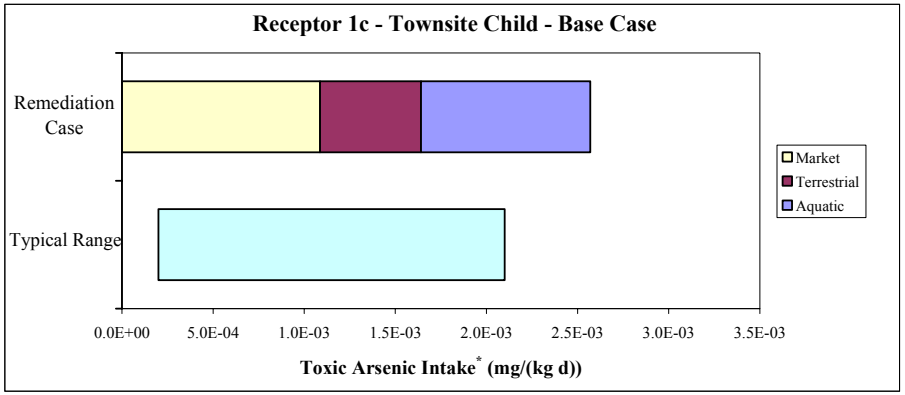
Receptors 3a and 3c – City of Yellowknife Residents

The estimated mean daily total arsenic intakes for Receptors 3a and 3c, who live in Yellowknife and have a diet similar to that of the typical Canadian, are presented in Figure F.3-3. As evidenced in the figure, the contribution of arsenic from market foods dominates the total arsenic intake and terrestrial pathways are the next significant contributor for the No Remediation scenario. The aquatic pathways are a relatively minor contributor. The estimated intakes of total arsenic are within the typical range of arsenic exposures for Canadian adults and children.

Next to market foods, the principal pathways of arsenic intake by these adult and child receptors are ingestion of fish (24 to 29%) as summarized in Table F.3-1. Receptors 3a and 3c are assumed to obtain their fish and ducks from North Yellowknife Bay.

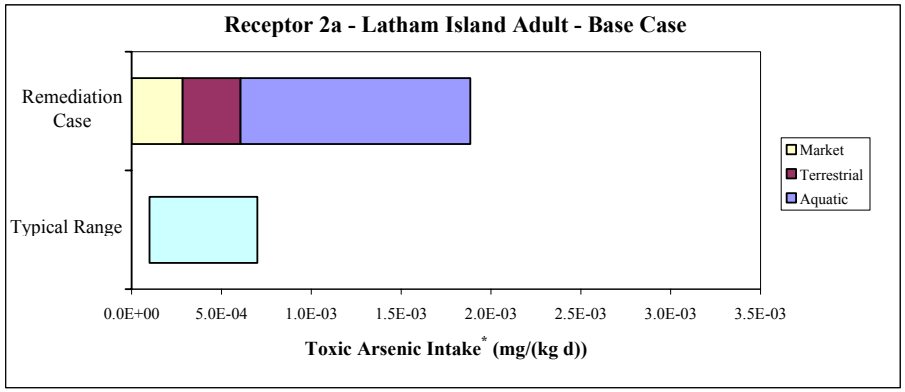
FIGURE F.3-1
ESTIMATED MEAN DAILY INTAKE OF TOXIC ARSENIC BY RECEPTOR 1 (GIANT MINE TOWNSITE) ADULT AND CHILD IN 2010

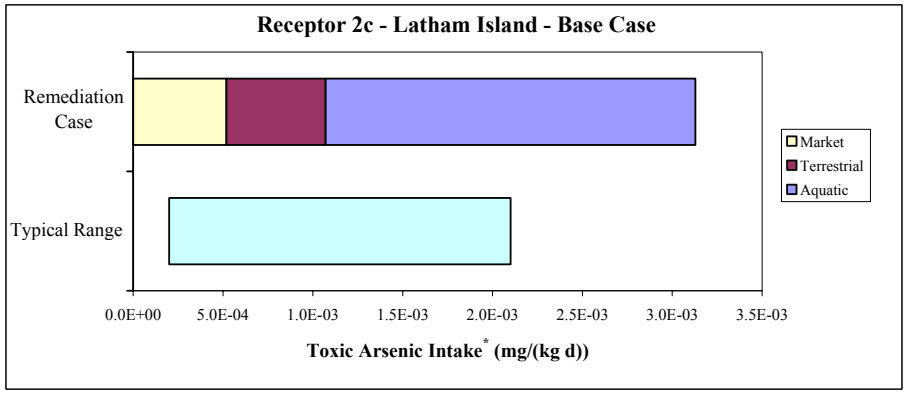




* Includes inorganic + toxic organic arsenic. Note all modelled market foods, 78% of fish and all other aquatic and terrestrial intakes were assumed to be in a toxic form. Typical range estimates are from Environment Canada (1993) and are an estimate of the inorganic arsenic only.

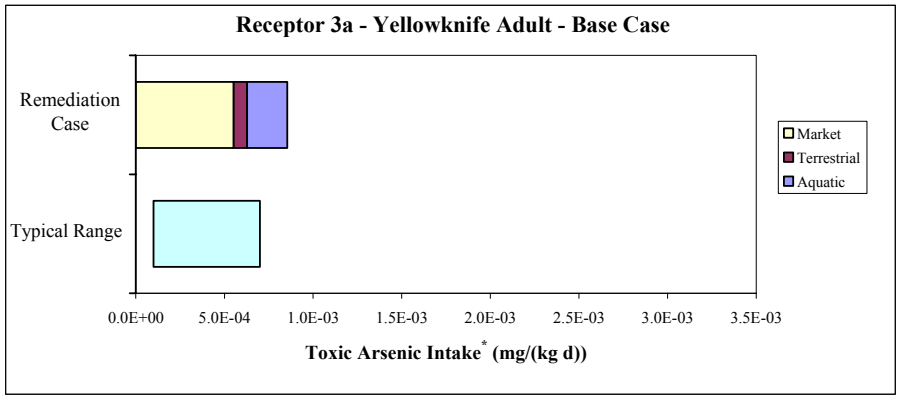
FIGURE F.3-2
ESTIMATED MEAN DAILY INTAKE OF TOTAL ARSENIC BY RECEPTOR 2 (LATHAM ISLAND) ADULT AND CHILD IN 2010

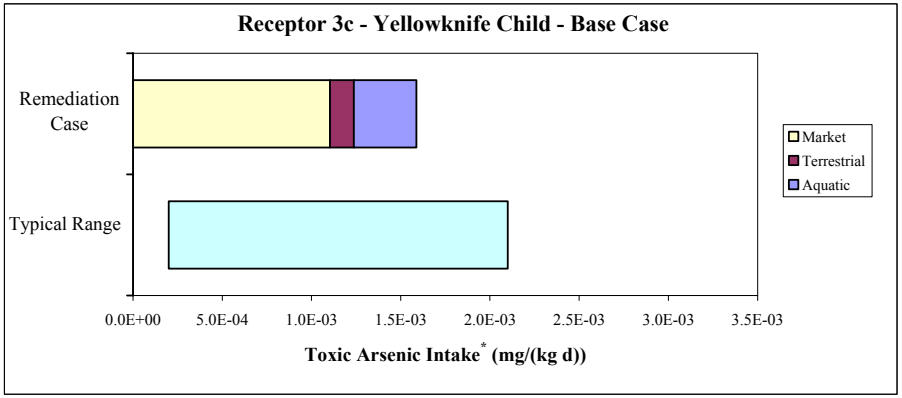




* Includes inorganic + toxic organic arsenic. Note all modelled market foods, 78% of fish and all other aquatic and terrestrial intakes were assumed to be in a toxic form. Typical range estimates are from Environment Canada (1993) and are an estimate of the inorganic arsenic only.

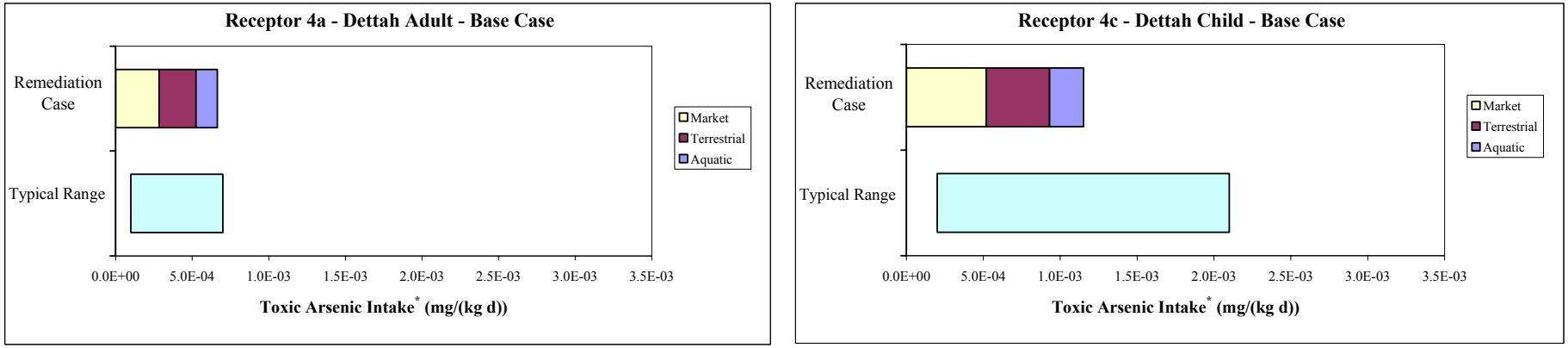
FIGURE F.3-3
ESTIMATED MEAN DAILY INTAKE OF TOTAL ARSENIC BY A RECEPTOR 3 (CITY OF YELLOWKNIFE) ADULT AND CHILD IN 2010





* Includes inorganic + toxic organic arsenic. Note all modelled market foods, 78% of fish and all other aquatic and terrestrial intakes were assumed to be in a toxic form. Typical range estimates are from Environment Canada (1993) and are an estimate of the inorganic arsenic only.

FIGURE F.3-4
ESTIMATED MEAN DAILY INTAKE OF TOTAL ARSENIC BY A RECEPTOR 4 (DETTAH COMMUNITY) ADULT AND CHILD IN 2010



* Includes inorganic + toxic organic arsenic. Note all modelled market foods, 78% of fish and all other aquatic and terrestrial intakes were assumed to be in a toxic form. Typical range estimates are from Environment Canada (1993) and are an estimate of the inorganic arsenic only.

TABLE F.3-1
ESTIMATED DAILY TOTAL ARSENIC INTAKES BY PATHWAY FOR EACH RECEPTOR FOR NO REMEDIATION (BASE CASE) SCENARIO IN 2010

Receptor	Total (mg/kg/d)				Mean Intake (mg/kg/d)												
					Water	Inhalation	Vegetation	Fish	Meat						Soil	Medicinal	Market
	5th	Mean	Median	95th					Moose	Caribou	Hare	Grouse	Mallard	Total		Tea	Foods
1a.Townsite - adult	5.6E-04	1.6E-03	1.3E-03	**3.5E-03**	6.3E-06	8.8E-07	1.5E-04	6.0E-04	6.1E-06	5.8E-05	3.2E-07	2.0E-05	1.2E-04	2.0E-04	5.7E-05	NA	5.6E-04
1c.Townsite - child	1.1E-03	**2.8E-03**	**2.4E-03**	**5.9E-03**	6.9E-06	1.8E-06	2.3E-04	8.9E-04	9.5E-06	9.1E-05	5.0E-07	2.9E-05	1.9E-04	3.2E-04	2.7E-04	NA	1.1E-03
2a.N'Dilo - adult	6.3E-04	**2.1E-03**	1.6E-03	**5.4E-03**	6.1E-06	9.1E-07	2.7E-05	1.3E-03	2.4E-05	3.7E-04	8.4E-07	5.8E-05	2.0E-05	4.7E-04	7.9E-06	1.1E-05	2.8E-04
2c.N'Dilo - child	1.1E-03	**3.5E-03**	**2.5E-03**	**9.0E-03**	6.9E-06	1.7E-06	4.2E-05	**2.1E-03**	4.0E-05	5.9E-04	1.4E-06	9.1E-05	3.3E-05	7.5E-04	3.9E-05	NA	5.2E-04
3a.Yellowknife - adult	3.2E-04	9.0E-04	7.7E-04	2.0E-03	5.9E-06	9.0E-07	2.4E-05	2.3E-04	6.2E-06	5.8E-05	3.0E-07	2.0E-05	2.0E-06	8.6E-05	3.4E-06	NA	5.5E-04
3c.Yellowknife - child	7.1E-04	1.6E-03	1.5E-03	**3.1E-03**	6.9E-06	1.7E-06	3.7E-05	3.5E-04	1.0E-05	9.2E-05	4.7E-07	3.0E-05	3.4E-06	1.4E-04	1.6E-05	NA	1.1E-03
4a.Dettah - adult	3.7E-04	8.6E-04	6.5E-04	**2.1E-03**	6.1E-06	9.1E-07	1.8E-05	1.3E-04	7.2E-06	3.5E-04	8.1E-07	5.7E-05	1.3E-06	4.1E-04	1.7E-06	2.2E-06	2.8E-04
4c.Dettah - child	6.7E-04	1.4E-03	1.1E-03	**3.4E-03**	6.9E-06	1.7E-06	2.7E-05	2.2E-04	1.2E-05	5.6E-04	1.4E-06	8.9E-05	2.2E-06	6.6E-04	8.4E-06	NA	5.2E-04

Receptor	Total (mg/kg/d)				Breakdown by Pathway (%)												
					Water	Inhalation	Vegetation	Fish	Meat						Soil	Medicinal	Market
	5th	Mean	Median	95th					Moose	Caribou	Hare	Grouse	Mallard	Total		Tea	Foods
1a.Townsite - adult	5.6E-04	1.6E-03	1.3E-03	**3.5E-03**	0.5%	<0.1%	11%	45%	0.5%	4%	<0.1%	1.5%	9.0%	15%	4.3%	NA	42%
1c.Townsite - child	1.1E-03	**2.8E-03**	**2.4E-03**	**5.9E-03**	0.3%	<0.1%	10%	37%	0.4%	4%	<0.1%	1.2%	7.8%	13%	11.5%	NA	45%
2a.N'Dilo - adult	6.3E-04	**2.1E-03**	1.6E-03	**5.4E-03**	0.4%	<0.1%	2%	82%	1.5%	24%	<0.1%	3.7%	1.3%	30%	0.5%	0.7%	18%
2c.N'Dilo - child	1.1E-03	**3.5E-03**	**2.5E-03**	**9.0E-03**	0.3%	<0.1%	2%	82%	1.6%	23%	<0.1%	3.6%	1.3%	29%	1.5%	NA	21%
3a.Yellowknife - adult	3.2E-04	9.0E-04	7.7E-04	2.0E-03	0.8%	0.1%	3%	29%	0.8%	7%	<0.1%	2.6%	0.3%	11%	0.4%	NA	71%
3c.Yellowknife - child	7.1E-04	1.6E-03	1.5E-03	**3.1E-03**	0.5%	0.1%	3%	24%	0.7%	6%	<0.1%	2.1%	0.2%	9%	1.1%	NA	74%
4a.Dettah - adult	3.7E-04	8.6E-04	6.5E-04	**2.1E-03**	0.9%	0.1%	3%	20%	1.1%	53%	0.1%	8.7%	0.2%	63%	0.3%	0.3%	43%
4c.Dettah - child	6.7E-04	1.4E-03	1.1E-03	**3.4E-03**	0.6%	0.2%	2%	20%	1.1%	50%	0.1%	8.0%	0.2%	60%	0.8%	NA	48%

Note: Total arsenic includes toxic organic and inorganic arsenic. Values that are shaded and in bold exceed the Health Canada TRV.

Receptors 4a and 4c – Dettah Community Residents

The estimated mean total daily arsenic intakes for Receptors 4a and 4c with an average fish diet are presented on Figure F.3-4. These Receptors (4a and 4c) were assumed to obtain all their drinking water from the municipal Yellowknife water supply and their, fish and ducks from South Yellowknife Bay. Garden produce and berries were assumed to be obtained from the area of the Dettah community. Large game and small game were assumed to be taken from the Baker Creek watershed to maximize the exposure estimate.

The main pathway of arsenic intake by Receptor 4a and 4c is associated with consumption of market foods and the terrestrial pathways; the aquatic pathways are the smallest contributor, as demonstrated on Figure F.3-4. Details of the contributions of all pathways are summarized in Table F.3-1 for the No Remediation scenario. A review of these tables indicates that when only inputs from the Giant Mine site and Yellowknife area are considered, caribou (over 50%) and fish (20%) are the main contributors to the total intakes by Receptor 4a and 4c.

In comparison to the range of arsenic intakes reported for Canadians eating a typical diet, the estimated intake of total arsenic for the child (Receptor 4c) falls within the range for the No Remediation scenario. For the adult receptor, the estimated total arsenic intake falls slightly above the typical range (see Figure F.3-4).

F.3.1 Non-Carcinogenic Effects of Inorganic Arsenic Exposure

The assessment of the daily intake of total (inorganic and toxic organic) arsenic by the four different hypothetical adult receptors showed that diets relating to Receptors 1a and 2a result in the highest exposures to total arsenic. Child receptors 1c and 2c received the highest exposures amongst the child receptors. The intakes for Townsite child (Receptor 1a) and the Latham Island adult and child (2a and 2c) were above the Health Canada Toxicity Reference Value (TRV) for arsenic of 0.002 mg/(kg d). This indicates that there is the potential for adverse health affects for these receptors. All other receptors examined had estimated arsenic intakes below the TRV.

F.3.2 Carcinogenic Risk of Inorganic Arsenic Exposure

Inorganic arsenic is known to have carcinogenic effects. Carcinogenic risk is expressed as the incremental incidence of developing cancer for a lifetime of exposure.

For this assessment, a composite person was used to capture the exposure over a lifetime (70 years of exposure) spanning a person's childhood and adult years. Table F.3-2 shows the lifetime risk levels for cancer calculated for the No Remediation (Base Case) scenario. The risk estimates summarized in the table are inclusive of all major oral exposure pathways including ingestion of water, food and contaminated soil. From this assessment, it can be seen that the

highest risk estimates are calculated for Receptor 2, at 2.8 in 1000 for the No Remediation scenario.

TABLE F.3-2
ESTIMATED MEAN LIFETIME CARCINOGENIC RISK (BASE CASE)

Receptor[a]	Receptor Name	Base Case
1	Giant Mine Townsite	2.1×10^{-3}
2	Latham Island	2.8×10^{-3}
3	City of Yellowknife Resident	1.2×10^{-3}
4	Dettah Community Resident	1.1×10^{-3}

Note: a - Composite individual encompassing 11 years as a child and 59 years as an adult.

The risk from arsenic exposure for Receptor 2 is higher than the risk associated with typical arsenic exposure, as are the cancer risks for the other three receptors.

F.3.3 Overall Significance

The risk assessment of human exposure to arsenic levels in the aquatic, atmospheric and terrestrial environments in the Yellowknife study area showed that exposure levels are potentially high for some individuals. The most exposed individuals are Receptors 1 and 2 living at the Giant Mine Townsite and on Latham Island respectively. The major contributors to exposure for these receptors are fish and market foods. The estimated total arsenic intakes for these individuals were higher than the range reported for typical Canadians (Environment Canada, 1993).

Estimated arsenic intake levels were above the toxicity reference value for non-cancer health effects for the child at the Townsite (Receptor 1c) and the adult and child on Latham Island (Receptors 2a,c). It is noted however, that some of arsenic consumed by the adult and child receptors carried through the assessment may be present in non-toxic forms. By contrast, the TRV is based on the toxic inorganic form. As there is a large uncertainty in the amount of toxic organic arsenic in fish, the assumption that a large portion of the organic arsenic was in the toxic form may lead to an overestimate of exposure.

F.4 REFERENCES

Canadian Council of Ministers of the Environment (CCME) 2002. *Canadian Environmental Quality Guidelines*.

Canadian Council of Ministers of the Environment (CCME) 1999. *Canadian Environmental Quality Guidelines.*

Environment Canada 1993. *Canadian Environmental Protection Act: Priority Substances List Assessment Report - Arsenic*.

Federal-Provincial-Territorial Committee on Drinking Water 2004. *Arsenic in Drinking Water. Document issued for Public Comment, May 17, 2004.*

Liber, K. and S.L. White Sobey 2000. *Toxicity of Uranium, Molybdenum, Nickel, and Arsenic to Hyalella azteca and Chironomus Tentans in Water-only and Spiked-sediment Toxicity Tests.* Toxicology Centre, University of Saskatchewan.

Receveur, O., M. Boulay, C. Mills, W. Carpenter and H. Kuhnlein 1996. *Variance in Food Use in Dene/Metis Communities.* Centre for Indigenous Peoples' Nutrition and Environment (CINE). October.

Thompson, P.A., J.A. Kurias and S.S. Mihok. 2002. *Using Benthic Invertebrates and Sediment Chemistry Monitoring Data for Estimating Lowest Effect/Severe Effect Levels at Uranium Mines In Canada*. Canadian Nuclear Safety Commission - Working Benchmarks. 2002 in preparation.

APPENDIX G

SAMPLE CALCULATION

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APPENDIX G SAMPLE CALCULATION

This appendix presents a sample calculation for the intake of arsenic by the adult and child Giant Mine Townsite receptors (Receptors 1a,c). To complete this calculation the results of the water and sediment modeling were used in the pathways calculations. The calculation procedure for the animals that are part of the diet (i.e. moose, hare, grouse, mallard) are demonstrated along with the estimation of intake by people. It is important to note that the example calculations are conducted in a deterministic fashion using the mean water and sediment concentrations in Baker Creek and Back Bay, and mean values for transfer factors.

In the full-scale assessment presented in the main body of this report, the pathways model was run in a probabilistic fashion integrated with the water quality model. Many of the input parameters in the model, such as sediment ingestion by the mallard, were assigned a distribution function to ensure that the entire range of possible values are captured within the calculation. The results from typically 1000 simulations were analyzed to determine the 5th percentile, mean and 95th percentile values of the model outputs.

For comparison, the mean estimated arsenic intakes by the adult and child receptors from the probabilistic simulation are compared to the estimated intakes determined from the deterministic spreadsheet calculations (see last page of spreadsheet printout). As seen in the calculations, the model mean output values agree well with the deterministic calculations.

G.1 REFERENCES

Baes, C.F. R.D. Sharp, A.L. Sjoreen, and R.W. Shar 1984. *A Review and Analysis of Parameters for Assessing Transport of Environmentally Released Radionuclides Through Agriculture.* ORNL-5786.

Belovsky, G.E., P.A. Jordan, and D.B. Botkin 1973. *Summer Browsing by Moose in Relation to Preference, and Animal Density: A New Quantitative Approach.* Proceedings of the 9th North American Moose Conference and Workshop, Quebec, City, March.

Beyer, W. N., E. Connor, and S. Gerould 1994. *Survey of Soil Ingestion by Wildlife*. Journal of Wildlife Management, 58:375-382.

Canadian Standards Association (CSA) 1987. *Guidelines for Calculating Derived Release Rate Limits for Radioactive Material in Airborne and Liquid Effluents for Normal Operation of Nuclear Facilities.* CAN ICSA-N288.1-M87.

Canadian Wildlife Service (CWS) 1997. *Hinterland Who's Who: Moose On-Line Report.* Environment Canada.

International Atomic Energy Agency (IAEA) 1994. *Handbook of Parameter Values for the Prediction of Radionuclide Transfer in Temperature Environments.* International Atomic Energy Agency, Vienna.

Kirk, R.W. (ed.), 1977. *Current Veterinary Therapy.* W.B. Sauders Co., Toronto, pp 1418.

Schmidt, J.L. and D.L. Gilbert 1978. *Big Game of North America, Ecology and Management.* Stackpole Books, Harrisburg, PA 17105.

SENES Consultants Limited 1987. *Uranium Tailings Assessment Program (UTAP.3): Component Model Documentation*. Research Report Prepared for the National Uranium Tailings Program, CANMET, EMR, under Supply and Services Canada Contract No. 0SQ85-00182.

Thomas, D.C. and S.J. Barry 1991. *Microhistological Analyses of Caribou Diet: Fecal Versus Rumen Samples and Other Variables.* Pages 516-529 in Proc. Fourth North Amer. Caribou Workshop. C.E. Buttler and S.P. Mahoney (eds). Newf.-Lab. Wild1. Div., St. John's, Newfoundland.

United States Environmental Protection Agency (U.S. EPA) 1998. *Human Health Risk Assessment Protocol for Hazardous Waste Combustion Facilities.* Peer Review Draft. EPA 530-D-98-001.

United States Environmental Protection Agency (U.S. EPA) 1993. *Wildlife Exposure Factors Handbook. Volume I and II.* Office of Research and Development, Washington DC.

United States National Council on Radiation Protection and Measurements (U.S. NCRP) 1996. *Screening Models for Release of Radionuclides to Atmosphere, Surface Water and Ground.* NCRP Report No. 123.

Wales, R.A., L.P. Milligan and E.H. McEwan 1975. *Urea Recycling in Caribou, Cattle and Sheep.* In First Int. Reinder and Caribou Sympo. J.R. Luick, P.C. Lent, D.R. Klein and R.G. White (eds), Univ. Alaska, Fairbanks, 297-307.

Concentrations				
Baker Creek water concentration	mg/L	watc_BC	1.18E-01	based on modelling (See Section 6.1, Table 6.1-1))
Baker Creek sediment concentration	mg/kg	sedc_BC	2210	based on modelling (See Section 6.1, Table 6.1-2)
Back Bay water concentration	mg/L	watc_BB	3.03E-03	based on modelling (See Section 6.1, Table 6.1-1))
Back Bay sediment concentration	mg/kg	sedc_BB	759	based on modelling (See Section 6.1, Table 6.1-3))
Municipal water concentration	mg/L	dwatc	3.0E-04	arsenic concentration in Yellowknife River
soil concentration - Townsite	mg/kg (dw)	soiltown	350	remediated case
soil concentration - Giant Mine site	mg/kg (dw)	soilmine	350	remediated case
air concentration	mg/m3	airc	4.00E-06	Average measured outdoor concentration. (See Appendix A, Table A.9-1)
above-ground vegetation concentration	mg/kg (ww)	agveg	5.50E-01	Average measured concentration. (See Appendix A. Table A.2-4).
below-ground vegetation concentration	mg/kg (ww)		2.40E-01	Average measured concentration. (See Appendix A, Table A.2-4).
berry concentration	mg/kg (ww)	berryc	5.20E-01	Average of measured on Giant Mine site. (See Appendix A, Table A.3-3)
bioaccessibility factor		biof	0.17	See (Appendix C)
soil concentration- Townsite (with bioavailability)	mg/kg (dw)	soiltownba	59.50	=soiltown*biof. Community soil concentration with bioavailablity - to assess human health
soil concentration- Giant Mine (with bioavailability)	mg/kg (dw)	soilmineba	59.50	=soilmine*biof. Minsite soil concentration with bioavailability - to assess terrestrial intake
Baker Creek sediment concentration (with bioavailability)	mg/kg (ww)	sedc_BCa	376	=sedc_BC*biof
Back Bay sediment concentration (with bioavailability)	mg/kg (ww)	sedc_BBa	129	=sedc_BB*biof
fish				
fish transfer factor	m3/g (ww)	fishTF	8.83E-05	site-specific transfer factor (average value). (See Appendix A, Table A5-3)
fish concentration - Baker Creek	mg/kg (ww)	fishc_BC	1.04E+01	=watc_BC*fishTF*1e6
fish concentration - Back Bay	mg/kg (ww)	fishc_BB	2.68E-01	=watc_BB*fishTF*1e6
forage				
vegetation transfer factor (summer forage & browse)	mg/kg (ww) per mg/kg (dw)	vegTF	0.0026	site-specific transfer factor. (See Appendix A, Table A.2-5)
deposition velocity	m/s	Vd	2.00E-02	SENES 1987
fraction of deposition retained	-	Frv	2.00E-01	SENES 1987

fraction of deposition on edible portions	-	Ev	9.00E-01	SENES 1987
weathering loss decay constant	1/s	lambda	5.73E-07	SENES 1987
duration of plant exposure to atmospheric deposition	s	tv	2.59E+06	SENES 1987
yield density	g (ww)/m2	Yv	7.50E+02	SENES 1987
forage concentration	mg/kg (ww)	foragec	0.93	=if(lambd*tv<75,vegTF*soilmine+(Vd*Frv*Ev*(1-exp(-lambda*tv))/(Yv*lambda)*airc*1000,vegTF*soilmine+(Vd*Frv*Ev)/(Yv*lambda)*airc*1000)
browse				
vegetation transfer factor (summer forage & browse)	mg/kg (ww) per mg/kg (dw)	vegTF	0.0026	site-specific transfer factor. (See Appendix A, Table A.2-5)
deposition velocity	m/s	Vd	2.00E-02	SENES 1987
fraction of deposition retained	-	Frv	2.00E-01	SENES 1987
fraction of deposition on edible portions	-	Ev	9.00E-01	SENES 1987
weathering loss decay constant	1/s	lambda	5.73E-07	SENES 1987
duration of plant exposure to atmospheric deposition	s	tv	7.80E+06	SENES 1987
yield density	g (ww)/m2	Yv	7.50E+02	SENES 1987
browse concentration	mg/kg (ww)	browsec	0.93	=if(lambd*tv<75,vegTF*soilmine+(Vd*Frv*Ev*(1-exp(-lambda*tv))/(Yv*lambda)*airc*1000,vegTF*soilmine+(Vd*Frv*Ev)/(Yv*lambda)*airc*1000)
lichen				
deposition velocity	m/s	Vd	2.00E-02	SENES 1987
fraction of deposition retained	-	Frv	9.00E-01	SENES 1987
fraction of deposition on edible portions	-	Ev	9.00E-01	SENES 1987
weathering loss decay constant	1/s	lambda	7.30E-09	SENES 1987
duration of plant exposure to atmospheric deposition	s	tv	1.00E+10	SENES 1987
yield density	g (ww)/m2	Yv	5.00E+02	SENES 1987
lichen concentration	mg/kg (ww)	lichen	1.78E+01	=if(lambd*tv<75,Vd*Frv*Ev*(1-exp(-lambda*tv))/(Yv*lambda)*airc*1000,Vd*Frv*Ev/(Yv*lambda)*airc*1000)

benthic				
TF water-to-benthic	L/kg ww	Tfben	6.30E+01	site-specific transfer factor. (See Appendix A, Section A.7)
benthic concentration	mg/kg (ww)	benc_BC	7.43E+00	=watc_BC*Tfben
benthic concentration	mg/kg (ww)	benc_BB	1.91E-01	=watc_BB*Tfben
aquatic vegetable				
TF water-to-aquatic veg	L/kg ww	Tfveg	7.49E+01	site-specific transfer factor (average). (See Appendix A, Table A.4-2)
aquatic veg concentration	mg/kg (ww)	aquavegc_BC	8.84E+00	=watc_BC*Tfveg
aquatic veg concentration	mg/kg (ww)	aquavegc_BB	2.27E-01	=watc_BB*Tfveg
caribou				
water intake	L/d	Qwatc	9.50E+00	based on Kirk (1977) and Wales et al. (1975)
total food intake (FW)	g (FW)/d	Qffwc	8.00E+03	calculated from U.S. EPA (1993)
fraction that is forage	-	ffc	1.00E-02	based on Thomas and Barry (1991)
fraction that is browse	-	fbwc	5.00E-02	based on Thomas and Barry (1991)
fraction that is soil	-	fsc	3.00E-02	based on Beyer et al. (1994)
fraction that is lichen	-	flic	9.10E-01	based on Thomas and Barry (1991)
soil intake	g (DW)/d	Qsdwc	2.40E+02	=fsc*Qffwc
body weight	kg	BWc	1.05E+02	Schmidt 1978
fraction of time in area	-	flocc	1.00E-01	Assumed
intake from water	mg/d	Iwater	1.12E-01	=watc_BC*Qwatc_a*flocc_a
intake from browse	mg/d	Ibrowse	3.73E-02	=browsec*Qffwc*fbwc/1000*flocc
intake from forage	mg/d	Iforage	7.40E-03	=foragec*Qffwc*ffc/1000*flocc
intake from soil	mg/d	Isoil	1.43E+00	=soilmineba*Qsdwc/1000*flocc
intake from lichen	mg/d	Ilichen	1.29E+01	=lichenc*Qffwc*flic/1000*flocc
total intake	mg/d	Itotal	14.51	=Iwater+Ibrowse+Iforage+Isoil+Ilichen
			13.7	*model output (mean value)*
TF feed-to-caribou	d/kg (ww)	Tfcaribou	3.04E-03	Based on IAEA (1994), U.S. NCRP (1996), Baes (1984), U.S. EPA (1998) and CSA (1987)
caribou concentration	mg/kg (ww)	caribouc	4.42E-02	=Itotal*Tfcaribou
			0.042	*model output (mean value)*
Moose				
water intake	L/d	Qwatmo	31	calculated from U.S. EPA (1993)
total food intake (FW)	g (FW)/d	Qffwmo	23000	Canadian Wildlife Service (1997)
fraction that is browse	-	fbwmo	0.9	Belovsky et al. (1973)

fraction of sediment ingestion	-	fsedmo	0.01	based on Beyer et al. (1994)
fraction that is aquatic veg	-	favmo	0.09	Belovsky et al. (1973)
sediment intake	g (DW)/d	Qsdwmo	230	=Qffwmo*fsedmo
body weight	kg	Bwmo	600	Canadian Wildlife Service (1997)
fraction of time in area	-	flocmo	0.25	assumed
intake from water	mg/d	Iwatermo	9.15E-01	=watc_BC*Qwatmo*flocmo
intake from browse	mg/d	Ibrowsemo	4.83E+00	=browsec*Qffwmo*fbwmo/1000*flocmo
intake from aquatic vegetable	mg/d	Iaqvegmo	4.57E+00	=aquavegc_BC*Qffwmo*favmo/1000*flocmo
intake from sediment	mg/d	Isedmo	2.16E+01	=sedc_BCa*Qsdwmo/1000*flocmo
total intake	mg/d	Itotalmo	3.19E+01	=Iwatermo+Ibrowsemo+Iaqvegmo+Isedmo
			111.0	*model output (mean value)*
TF feed-to-moose	d/kg (ww)	Tfmoose	3.04E-03	IAEA 1994, Baes et al. 1984, U.S. EPA 1998, CSA 1987
moose concentration	mg/kg (ww)	moosec	9.71E-02	=Itotalmo*Tfmoose
			0.095	*model output (mean value)*
Mallard				
water intake	L/d	Qwatma	6.20E-02	U.S. EPA (1993)
total food intake (FW)	g (FW)/d	Qffwma	2.50E+02	U.S. EPA (1993)
fraction that is aquatic vegetation	-	favma	2.50E-01	U.S. EPA (1993)
fraction that is sediment	-	fsedma	1.00E-02	based on Beyer et al. (1994)
fraction that is benthic invertebrates		fbima	7.40E-01	U.S. EPA (1993)
sediment intake	g (DW)/d	Qsdwma	2.50E+00	=Qffwma*fsedma
body weight	kg	Bwma	1.13	U.S. EPA (1993)
fraction of time in area	-	flocma	5.00E-01	U.S. EPA (1993)
Baker Creek				
intake from water	mg/d	Iwaterma	3.66E-03	=watc_BC*Qwatma*flocma
intake from aquatic vegetable	mg/d	Iaqvegma	2.76E-01	=aquavegc_BC*Qffwma*favma/1000*flocma
intake from benthic	mg/d	Ibencma	6.88E-01	=benc_BC*Qffwma*fbima/1000*flocma
intake from sediment	mg/d	Isedma	4.70E-01	=sedc_BCa*Qsdwma/1000*flocma
total intake	mg/d	Itotalma	1.44E+00	=Iwaterma+Iaqvegma+Ibencma+Isedma
			0.82	*model output (mean value)*
TF feed-to-mallard	d/kg (ww)	Tfmallard	3.65E+00	IAEA 1994, Baes et al. 1984, U.S. EPA 1998, CSA 1987
mallard concentration	mg/kg (ww)	mallardc_BC	5.25E+00	=Itotalma*Tfmallard
			2.04	*model output (mean value)*
Back Bay				

intake from water	mg/d	Iwaterma	9.39E-05	=watc_BB*Qwatma*flocma
intake from aquatic vegetable	mg/d	Iaqvegma	7.09E-03	=aquavegc_BB*Qffwma*favma/1000*flocma
intake from benthic	mg/d	Ibencma	1.77E-02	=benc_BB*Qffwma*fbima/1000*flocma
intake from sediment	mg/d	Isedma	1.61E-01	=sedc_BBa*Qsdwma/1000*flocma
total intake	mg/d	Itotalma	1.86E-01	=Iwaterma+Iaqvegma+Ibencma+Isedma
TF feed-to-mallard	d/kg (ww)	Tfmallard	3.65E+00	IAEA 1994, Baes et al. 1984, U.S. EPA 1998, CSA 1987
mallard concentration	mg/kg (ww)	mallardc_BB	6.80E-01	=Itotalma*Tfmallard
Hare				
water intake	L/d	Qwath	0.14	calculated from U.S. EPA (1993)
total food intake (FW)	g (FW)/d	Qffwh	300	Pease et al. 1979
fraction that is browse	-	fbrh	0.6	U.S EPA 1993
fraction of food that is summer forage	-	fsfh	0.38	U.S. EPA 1993
fraction that is soil		fsh	0.02	based on Beyer et al. (1994)
soil intake	g (DW)/d	Qsh	6	=Qffwh*fsh
body weight	kg	BWh		
fraction of time in area	-	floch	1	assumed
intake from water	mg/d	Iwaterh	1.65E-02	=watc_BC*Qwath*floch
intake from browse	mg/d	Ibrowseh	1.68E-01	=brousec*Qffwh*fbrh/1000*floch
intake from soil	mg/d	Isoilh	3.57E-01	=soilmineba*Qsh/1000*floch
intake from summer forage	mg/d	Iforage	1.06E-01	=foragec*Qffwh*fsfh/1000*floch
total intake	mg/d	Itotalh	6.47E-01	=Iwaterg+Ibrowseh+Isoilh+Iforageh
			0.67	*model output (mean value)*
TF feed-to-hare	d/kg (ww)	Tfhare	3.04E-03	IAEA 1994, Baes et al. 1984, U.S. EPA 1998, CSA 1987
hare concentration	mg/kg (ww)	harec	1.97E-03	=Itotalh*Tfhare
			1.9E-03	*model output (mean value)*
Grouse				
water intake	L/d	Qwatgr	0.036	calculated from U.S. EPA (1993)
total food intake (FW)	g (FW)/d	Qffwgr	120	U.S. EPA 1993
fraction that is berries	-	fbegr	0.15	U.S. EPA 1993
fraction of food that is summer forage	-	fsfgr	0.83	U.S. EPA 1993
fraction that is soil		fsgr	0.02	based on Beyer et al. (1994)
soil intake	g (DW)/d	Qsgr	2.4	=Qffwgr*fsgr
body weight	kg	BWgr		
fraction of time in area	-	flocgr	1	assumed

intake from water	mg/d	Iwatergr	4.25E-03	=watc_BC*Qwatgr*flocgr
intake from berries	mg/d	Iberrygr	9.36E-03	=berryc*Qffwgr*fbegr/1000*flocgr
intake from soil	mg/d	Isoilgr	1.43E-01	=soilmineba*Qsgr/1000*flocgr
intake from summer forage	mg/d	Iforagegr	9.22E-02	=foragec*Qffwgr*fsfgr/1000*flocgr
total intake	mg/d	Itotalgr	2.49E-01	=Iwatergr+Iberrygr+Isoilgr+Iforagegr
			0.26	*model output (mean value)*
TF feed-to-grouse	d/kg (ww)	Tfgrouse	3.65E+00	IAEA 1994, Baes et al. 1984, U.S. EPA 1998, CSA 1987
grouse concentration	mg/kg (ww)	grousec	9.08E-01	=Itotalgr*Tfgrouse
			6.2E-01	*model output (mean value)*

SAMPLE CALCULATION - Townsite Receptor

Adult Characteristics				**Adult**
Water ingestion rate	L/d	wira	1.5	(Section 3.4, Table 3.4-9)
Caribou ingestion rate	g/d	cira	54.3	(Section 3.4, Table 3.4-9)
Moose ingestion rate	g/d	mira	1.2	(Section 3.4, Table 3.4-9)
Hare ingestion rate	g/d	hira	1.5	(Section 3.4, Table 3.4-9)
Fish ingestion rate	g/d	fira	34.3	(Section 3.4, Table 3.4-9)
Grouse ingestion rate	g/d	gira	0.3	(Section 3.4, Table 3.4-9)
Mallard ingestion rate	g/d	maira	0.6	(Section 3.4, Table 3.4-9)
Berries ingestion rate	g/d	bira	8.7	(Section 3.4, Table 3.4-9)
Above-ground veg ingestion rate	g/d	agvegira	6.8	(Section 3.4, Table 3.4-9)
Below-ground veg ingestion rate	g/d	bgvegira	9.4	(Section 3.4, Table 3.4-9)
Body weight	kg	BWa	70.7	(Section 3.4, Table 3.4-9)
Time at site	-	loca	1	Assumed equal to 12 months/yr
Soil ingestion rate	g/d	sira	0.02	(Section 3.4, Table 3.4-9)
Air Inhalation Rate	m^3/d	aira	15.8	(Section 3.4, Table 3.4-9)
above-ground veg - fraction remaining after prep	-	fpagv	0.83	SENES 1987
below-ground veg - fraction remaining after prep	-	fpbgv	0.75	SENES 1987
Intake for Adult (Townsite)				
dose from water	mg/(kg d)	Dwater_a	6.36E-06	=dwatc*wira/BWa*loca
dose from caribou	mg/(kg d)	Dcaribou_a	3.39E-05	=caribouc*cira/BWa*loca/1000
dose from moose	mg/(kg d)	Dmoose_a	1.68E-06	=moosec*mira/BWa*loca/1000
dose from hare	mg/(kg d)	Dhare_a	4.19E-08	=harec*hira*loca/1000/BWa
dose from fish	mg/(kg d)	Dfish_a	1.30E-04	=fishc_BB*fira/BWa*loca/1000
dose from grouse	mg/(kg d)	Dgrouse_a	4.42E-06	=grousec_a*gira*loca/1000/BWa
dose from mallard	mg/(kg d)	Dmallard_a	2.71E-05	=avg(mallardc_BB,mallardc_BC)*maira/BWa*loca/1000
dose from berries	mg/(kg d)	Dberry_a	6.41E-05	=berryc*bira/BWa*loca/1000
dose from above-ground veg	mg/(kg d)	Dagveg_a	4.42E-05	=agvegc*agvegira/BWa*loca/1000*fpagv
dose from below-ground veg	mg/(kg d)	Dbgveg_a	2.39E-05	=bgvegc*agvegira/BWa*loca/1000*fpbgv
dose from soil	mg/(kg d)	Dsoil_a	1.48E-05	=soiltownba*sira*loca/1000/BWa
dose from market foods	mg/(kg d)	Dmarket_a	5.64E-04	=model output (as described in Appendix d)
total dose from ingestion	mg/(kg d)	Dingestion_a	9.14E-04	=Dwater_a+Dcaribou_a+Dmoose_a+Dhare_a+Dfish_a+Dgrouse_a+Dmallard_a+Dbeaver_a+Dberry_a+Dsoil_a+Dmarket_a
dose from inhalation	mg/(kg d)	Dair_a	8.94E-07	=aira*airc*loca/Bwa

H Sample Calculation – Townsite Receptor

Child Characteristics				**Child**
Water ingestion rate	L/d	wirc	0.8	(Section 3.4, Table 3.4-9)
Caribou ingestion rate	g/d	circ	41.7	(Section 3.4, Table 3.4-9)
Moose ingestion rate	g/d	mirc	0.9	(Section 3.4, Table 3.4-9)
Hare ingestion rate	g/d	hirc	1.2	(Section 3.4, Table 3.4-9)
Fish ingestion rate	g/d	firc	26.4	(Section 3.4, Table 3.4-9)
Grouse ingestion rate	g/d	girc	0.3	(Section 3.4, Table 3.4-9)
Mallard ingestion rate	g/d	mairc	0.5	(Section 3.4, Table 3.4-9)
Berries ingestion rate	g/d	birc	6.4	(Section 3.4, Table 3.4-9)
Above-ground veg ingestion rate	g/d	agvegirc	4.9	(Section 3.4, Table 3.4-9)
Below-ground veg ingestion rate	g/d	bgvegirc	8.1	(Section 3.4, Table 3.4-9)
Body weight	kg	BWc	32.9	(Section 3.4)
Time at site	-	locc	1	Assumed equal to 12 months/yr
Soil ingestion rate	g/d	sirc	0.04	(Section 3.4)
Air Inhalation Rate	m^3/d	airch	14.5	(Section 3.4)
above-ground veg - fraction remaining after prep	-	fpagv	0.83	SENES 1987
below-ground veg - fraction remaining after prep	-	fpbgv	0.75	SENES 1987
Intake for Child (Townsite)				
dose from water	mg/(kg d)	Dwater	7.29E-06	=dwatc*wirc/BWc*locc
dose from caribou	mg/(kg d)	Dcaribou	5.60E-05	=caribouc*circ/BWc*locc/1000
dose from moose	mg/(kg d)	Dmoose	2.77E-06	=moosec*mirc/BWc*locc/1000
dose from hare	mg/(kg d)	Dhare	6.91E-08	=hirc*harec*locc/1000/BWc
dose from fish	mg/(kg d)	Dfish	2.15E-04	=fishc_BB*firc/BWc*locc/1000
dose from grouse	mg/(kg d)	Dgrouse	7.29E-06	=grousec*girc*locc/1000/BWc
dose from mallard	mg/(kg d)	Dmallard	4.47E-05	=avg(mallardc_BC,mallardc_BB)*mairc/BWc*locc/1000
dose from berries	mg/(kg d)	Dberry	1.01E-04	=berryc*birc/BWc*locc/1000
dose from above-ground veg	mg/(kg d)	Dagveg	6.80E-05	=agvegc*agvegirc/BWc*locc/1000*fpagv
dose from below-ground veg	mg/(kg d)	Dbgveg	4.41E-05	=bgvegc*agvegirc/BWc*locc/1000*fpbgv
dose from soil	mg/(kg d)	Dsoil	7.23E-05	=soiltownba*sirc*locc/1000/BWc
dose from market foods	mg/(kg d)	Dmarket	1.09E-03	=model output (as described in Appendix d)
total dose from ingestion	mg/(kg d)	Dingestion	1.71E-03	=Dwater+Dcaribou+Dmoose+Dhare+Dfish+Dgrouse+Dmallard+Dbeaver++Dberry+Dsoil
dose from inhalation	mg/(kg d)	Dair	1.76E-06	=airch*airc*locc/BWc

Comparison of Sample Calculation and Model Output

Townsite Adult	Total Intake Mean	Mean Intake (mg/kg/d)												
		Water	Inhalation	Vegetation	Fish	Meat						Soil	Medicinal Tea	Market Foods
						Moose	Caribou	Hare	Grouse	Mallard	Total			
*Model Output**	*8.8E-04*	*6.2E-06*	*9.0E-07*	*1.4E-04*	*1.0E-04*	*1.7E-06*	*3.2E-05*	*4.6E-08*	*3.4E-06*	*1.9E-05*	*5.6E-05*	*1.5E-05*	*NA*	*5.6E-04*
Sample Calc	9.1E-04	6.4E-06	8.9E-07	1.3E-04	1.3E-04	1.7E-06	3.4E-05	4.2E-08	4.4E-06	2.7E-05	6.7E-05	1.5E-05	NA	*5.6E-04*

Townsite child	Total Intake Mean	Mean Intake (mg/kg/d)												
		Water	Inhalation	Vegetation	Fish	Meat						Soil	Medicinal Tea	Market Foods
						Moose	Caribou	Hare	Grouse	Mallard	Total			
*Model Output**	*1.6E-03*	*7.0E-06*	*1.7E-06*	*2.3E-04*	*1.5E-04*	*2.6E-06*	*4.7E-05*	*7.1E-08*	*5.1E-06*	*2.8E-05*	*8.3E-05*	*7.1E-05*	*NA*	*1.1E-03*
Sample Calc	1.7E-03	7.3E-06	1.8E-06	2.1E-04	2.1E-04	2.8E-06	5.6E-05	6.9E-08	7.3E-06	4.5E-05	1.1E-04	7.2E-05	NA	*1.1E-03*

* Mean value from 1000 trials.