ASSESSMENT OF ECOLOGICAL RISKS POSED BY ARSENIC CONTAMINATION IN YELLOWKNIFE, NWT

FINAL REPORT

Submitted to:
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EXECUTIVE SUMMARY

A screening level ecological risk assessment was conducted to evaluate the potential risks posed by selected avian and mammalian wildlife species by arsenic-contaminated soils in and near Yellowknife, NWT. The selected species were: goshawk, spruce grouse, snowshoe hare and lynx. Exposure pathways included ingestion of contaminated soil, ingestion of contaminated water, and ingestion of contaminated foods. The concentration of arsenic in soil averaged 133 μg/g (ppm), which is within the natural levels for the mineral-enriched geologic formation that underlies the area.

Considering direct soil ingestion as well as arsenic intake via ingestion of water and foods contaminated with arsenic, it was determined that goshawks and spruce grouse received estimated daily doses that were less than one quarter of the toxicological benchmark dose (BMD) for arsenic in those species. Estimated arsenic exposures in lynx and snowshoe hare exceeded their respective BMD values by factors of 2.4 and 2.6, respectively.

Whether or not these exposures present a true risk to Yellowknife mammalian wildlife could not be determined with the available information. Uncertainties existing in the risk assessment precluded certainty in the characterization of wildlife risks. In particular: a) lack of data on arsenic levels in prey animals may have resulted in an under- or over-estimation of arsenic intake via food consumption in goshawks and lynx; b) BMD values for selected wildlife species were extrapolated from bioassays conducted on other (laboratory) species; and c) the elevated natural levels of arsenic in the Yellowknife area likely results in local wildlife populations with resistance to arsenic toxicity, relative to laboratory animals. In other words, resident wildlife will likely have adapted to the high natural arsenic contamination in the area.

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1.0 INTRODUCTION

1.1 General

Under the authorization of the Yellowknife Arsenic Soils Remediation Committee (YASRC), Risklogic Scientific Services Inc. has undertaken a screening level assessment of the ecological risks posed by arsenic contamination in and near the NWT community of Yellowknife. Methods for ecological risk assessment are available from agencies such as the Canadian Council of Ministers of Environment (CCME, 1997a, 1996) and Environment Canada (EC, 1994). The ecological risk assessment presented herein focused on specific populations of selected wildlife species, and did not address broader community or ecosystem impacts, due largely to the nature and limitations of available data.

1.2 Background

Arsenic is a natural element that is ubiquitous in the environment. It is released from both natural and man-made sources and is most commonly found in association with deposits of lead, zinc, copper, gold and other sulphide ores.

Arsenic can occur in four oxidation states, as arsine (-3), arsenic metal (0), arsenite (+3), and arsenate (+5) (CCME, 1997). Arsenic bonds covalently with most nonmetals and metals and forms stable organic compounds in both its trivalent and pentavalent states (CCME, 1997). In the environment, arsenic occurs most commonly as sulphides and as complexes with iron, nickel, copper, and cobalt. Arsenic is found primarily in its pentavalent form in soil (EC, 1999). However, arsenic trioxide is the anthropogenic form of arsenic most commonly released to the environment as a result of ore roasting (EC/HC, 1993).

Gold was discovered in Yellowknife in 1936, with the Con Mine beginning operations in 1938 and the Giant Mine beginning production in 1948 (CPHA, 1977). Ore roasting was undertaken at the Con Mine from 1941 until 1970, with production and roasting being interrupted during the Second World War. At the Giant Mine, ore roasting was an integral part of the gold production process from initial production in 1948 until the roaster ceased operating in early October, 1999. The roasting process is specifically intended to liberate sulphur, along with arsenic and other constituents of the raw ore, to enable easier purification of the gold. Therefore, the emission of arsenic from the stack was a well known phenomenon, with regulatory attention, particularly arsenic emissions and deposition monitoring, becoming prominent in the early to mid 1970s.

A variety of environmental investigations and studies have been conducted since the 1970s in Yellowknife with respect to the arsenic contamination (see RSSI, 2002a). These studies were undertaken primarily to evaluate the potential human health risks posed by atmospheric emissions and deposition from the Giant Mine ore roaster stack. However, no risk assessment has yet been published on the ecological risks posed by arsenic contamination in the area.

The report presented herein is one of three reports prepared by RSSI on behalf of the Yellowknife Arsenic Soil Remediation Committee. Those other reports examine human health risks (RSSI, 2002a) and investigate the natural occurrence of arsenic in the Yellowknife area and propose remediation (clean up) objectives consistent with the levels of natural contamination (RSSI, 2002b). Those other reports can be consulted for additional information on previous studies conducted in the Yellowknife area and levels of arsenic measured in soil in different regions of the area.

2.0 PROBLEM FORMULATION

Problem formulation involves the review of available information in order to identify the contaminant(s) of concern, critical receptors, and the critical pathways for exposure, towards formulating a generalized model as the conceptual basis of the quantitative risk assessment. Problem formulation does not entail the detailed analysis of site data, nor the detailed characterization of the site and potential receptors. Rather, it entails a scoping and outline of the general issues to be addressed. The site-specific and receptor-specific data and other information used in this risk assessment are more thoroughly described in later sections of this report.

The conceptual model for this screening level ecological risk assessment is presented in Figure 2.1. The selected, representative species and potential pathways of exposure are described below.

2.1 Contaminant of Concern

The only contaminant of concern within the terms of reference of this project was arsenic. The overall objective of this screening level ecological risk assessment was to determine whether or not selected representative species might take in a sufficiently large daily dose of arsenic to present a potential risk of harm.

Arsenic levels in soil in and around Yellowknife are elevated relative to the Canadian Council of Ministers of Environment (CCME, 1997) ecologically-based soil quality guideline of 17 µg/g (for agricultural and residential/parkland soils). However, that ecological guideline was based on laboratory studies investigating the contact toxicity of arsenic with plants and soil invertebrates (EC, 1999). The applicability of that ecological soil quality guideline to the Yellowknife environment is questionable. In particular, the background (natural) levels of arsenic in soil in the area are known to exceed the guideline value by one to two orders of magnitude (see RSSI, 2002b) and populations of plants and invertebrates found in the region likely have adapted to these natural arsenic background levels.

Despite the arsenic contamination being within natural levels for the area, the ecological risk assessment was conducted in order to quantify overall potential risks.

2.2 Critical Ecological Receptors

The risk assessment presented herein focuses on four representative terrestrial wildlife species common to the Yellowknife area. These are the spruce grouse and the goshawk, as representative avian species, and the snowshoe hare and common lynx, as representative mammalian species. Although other avian and mammalian species are known to exist in the area, an assessment of each and every species that has been observed in the region is beyond the scope of this screening assessment. The selected species were chosen for their representation of avian and mammalian fauna, and representation of herbivorous prey and carnivorous predatory species. No other wildlife species were identified that were considered to have special characteristics, or to inhabit a particular habitat or ecological niche, that would put them at greater risk than the four selected species. Therefore, it was considered likely that potential risks, or the lack thereof, posed to these four species would be indicative of potential risks posed to wildlife in general within the Yellowknife area.

Caribou were excluded from the risk assessment for two reasons. Firstly, caribou are seldom seen in the immediate vicinity of Yellowknife (B. Colpitts, Stanton Regional Health Board, personal communication). Secondly, their habit of constant migration (Calef, 1981) means that they would only remain in a contaminated area for a very short period of time each year, a duration that would have insignificant impact on their overall exposure to arsenic.

Fish and other aquatic species were also excluded from this assessment. The closure of the Giant Mine ore roaster in October 1999 eliminated industrial atmospheric fallout as a source of contamination to local surface waters. Fish monitoring studies (Jackson et al., 1996) indicate that local fish species are not contaminated with arsenic beyond levels typically seen in other locations in Canada (Braune et al., 1999a,b; EC/HC, 1993). Although mine tailings deposits still contaminate local freshwater sediments (Jackson et al., 1996), the assessment of sediment contamination was outside the terms of reference of this project.

Potential impacts on local flora (plants) and terrestrial invertebrates were also excluded from this risk assessment, due to a lack of data, the fact that the local ecology in areas beyond mine property shows no overt signs of impact (such as inhibition of vegetation growth, forest die-back, etc.), and the relatively high natural levels of arsenic in area soils would preclude a determination of whether or not the natural or anthropogenic arsenic was the cause of any postulated impacts.

2.3 Pathways of Potential Exposure for Ecological Receptors

Exposure to arsenic in terrestrial wildlife species may occur by direct ingestion of soil. Animals are known to directly ingest soil as a result of feeding, grooming and, in the case of certain avian species, ingestion of grit (Sheppard, 1995; McMurter, 1993). Indirect exposure will also result via ingestion of contaminated foods (vegetation consumed by herbivores; prey consumed by predators) into which arsenic has accumulated from the soil.

Following closure of the Giant Mine ore roaster, most surface waters (other than those with direct mine discharges) are no longer impacted. However, some local lakes within the Yellowknife area may still have elevated dissolved arsenic concentrations due to the presence of contaminated sediments and tailings. Therefore, exposure of wildlife via surface water ingestion was also included as a pathway of concern.

With the closure of the roaster facility, air quality monitoring indicates that arsenic is no longer an atmospheric contaminant (Koski, 2001a,b). As a result, inhalation of particle-borne arsenic was excluded as a pathway of concern for wildlife species. Also excluded as a pathway of concern was dermal absorption. The presence of fur or feathers on receptor species will inhibit or prevent the contact of soil with the skin. However, soil deposited onto feathers or fur will be ingested during grooming activities, and this intake is considered as a component of soil ingestion.

3.0 ECOLOGICAL RECEPTOR CHARACTERIZATION

Characteristics of the wildlife species required to evaluate the potential exposures and risks posed by arsenic, are summarized in Table 3.1 and are described in detail below.

3.1 Goshawk (Accipiter gentilis)

Many goshawks are year-round residents of the NWT (although varying proportions of their populations also migrate south) (GNWT-RWED, undated-a). The goshawk is North America's largest woodland hawk species. As with most birds of prey, this species is sexually dimorphic with an adult female weighing about 1.02 kg, and adult males weighing slightly less about 0.88 kg (Clarke, 1994). The goshawk diet consists of grouse, ptarmigan, snowshoe hares, other small mammals (squirrels, voles, shrews) as well as some of the larger songbirds and shorebirds (Clarke, 1994).

No data were located on the rate of food ingestion by the goshawk, specifically. However, based on body weight and metabolic rate considerations, food intake for birds can be estimated from the following equation (Sample et al., 1997):

$$\text{Daily food intake (kg dry weight)} = 0.0582 \times W^{0.651}$$

where, W = body weight (kg)

Based on this allometric equation, average adult food intake requirements for adult goshawk (average body weight = 0.95 kg) are estimated to be approximately 56 g of food (dry weight) per day. The food for goshawks is composed exclusively of meat. Meat, such as that of the spruce grouse, is approximately 68% water (Sample et al., 1997). Therefore the dry food intake of goshawks corresponds to a fresh weight food intake rate of 175 g of meat (fresh weight)/day.

Birds are known to inadvertently ingest soil during feeding and may intentionally ingest grit to aid digestion (Beyer et al., 1994; Sheppard, 1995). Inadvertent soil intake is lower among bird species that do not probe into soil, mud or sediment for food (Sheppard, 1995; Beyer et al., 1994). Based on analyses of faeces, browsing birds ingest soil corresponding to<2% to 10% of dry food ingested (Sheppard, 1995; Beyer et al., 1994). Based on a dry weight food intake rate of 56 g per day, goshawks may therefore ingest between 1 g and 6 g of soil daily. For purposes of this screening level ecological risk assessment, and lacking other more precise data, it was assumed that adult goshawks ingest 6 g of soil per day.

Although winter conditions and snow cover would likely reduce or preclude contact by goshawks with soil (for ingestion) during winter months, it was conservatively assumed that soil would be available for ingestion throughout the entire year.

No data were located on the rate of water intake by the goshawk. However, based on body weight and metabolic rate considerations, water intake for birds can be estimated from the following equation (Sample et al., 1997):

$$\text{Daily water intake (L)} = 0.059 \times W^{0.67}$$

where, W = body weight (kg)

Based on this allometric equation, average adult water intake requirements for adult goshawks (average body weight = 0.95 kg) are estimated to be approximately 0.06 L of water per day.

3.2 Spruce grouse (Dendragapus canadensis)

Spruce grouse are a major prey species for the goshawk (Clarke, 1994). Adult spruce grouse weigh approximately 0.5 kg (Gov. Nfld&Lab, no date). Their diet is varied, consisting principally of berries and other succulent vegetation in the spring, summer and fall, but rely almost exclusively on spruce needles for sustenance during winter (Ellison, 1994).

No data were located on the rate of food ingestion by the spruce grouse, specifically. However, based on body weight and metabolic rate considerations, food intake for birds can be estimated from the following equation (Sample et al., 1997):

$$\text{Daily food intake (kg dry weight)} = 0.0582 \times W^{0.651}$$

where, W = body weight (kg)

Based on this allometric equation, average food intake requirements for adult spruce grouse (average body weight = 0.5 kg) are estimated to be approximately 40 g of food (dry weight) per day.

Birds are known to inadvertently ingest soil during feeding and may intentionally ingest grit to aid digestion (Beyer et al., 1994; Sheppard, 1995). Inadvertent soil intake is lower among bird species that are browsers, and that do not probe into soil, mud or sediment for food (Sheppard, 1995; Beyer et al., 1994). Spruce grouse are browsers. Based on analyses of faeces, browsing birds ingest soil corresponding to<2% to 10% of dry food ingested (Sheppard, 1995; Beyer et al., 1994). Based on a dry weight food intake rate of 40 g per day, spruce grouse may ingest between 1 g and 4 g of soil daily. For purposes of this screening level ecological risk assessment, and lacking other more precise data, it was assumed that adult spruce grouse ingest 4 g of soil per day.

Although winter conditions and snow cover would likely reduce or preclude contact by grouse with soil (for ingestion) during winter months, it was conservatively assumed that soil would be available for ingestion throughout the entire year.

No data were located on the rate of water intake by the spruce grouse. However, based on body weight and metabolic rate considerations, water intake for birds can be estimated from the following equation (Sample et al., 1997):

$$\text{Daily water intake (L)} = 0.059 \times W^{0.67}$$

where, W = body weight (kg)

Based on this allometric equation, average adult water intake requirements for adult spruce grouse (average body weight = 0.5 kg) are estimated to be approximately 0.04 L of water per day.

*3.3 Snowshoe hare (*Lepus americanus*)*

Adult snowshoe hares are somewhat larger than cottontail rabbits (*Sylvilagus* spp.), averaging 1.4 kg to 1.8 kg in weight (Earnest, 1994). They feed on a wide variety of plant material - grasses, buds, twigs, and leaves in the summer and spruce twigs and needles, bark, and buds of hardwood such as aspen and willow in the winter (Earnest, 1994). Snowshoe hares are one of the more important food items of northern furbearers, particularly lynx (GNWT-RWED, undated-b).

No data were located on the rate of food ingestion by the snowshoe hare, specifically. However, based on body weight and metabolic rate considerations, food intake for herbivorous mammals can be estimated from the following equation (Sample et al., 1997):

$$\text{Daily food intake (kg dry weight)} = 0.0875 \times W^{0.727}$$

where, W = body weight (kg)

Based on this allometric equation, the average food intake requirement for the adult snowshoe hare of 1.6 kg average weight is estimated to be approximately 125 g of food (dry weight) per day.

Soil intake by herbivorous rodents, including rabbits and hares, ranges between <2% and 7.9% of daily dietary intake (as dry weight) (McMurter, 1993; Beyer et al., 1994). Based on a daily food intake rate of 125 g dry weight, and assuming that soil intake in the snowshoe hare is 7.9% of dry matter intake, then the estimated daily soil ingestion rate for this species is 10 g/d (dry weight).

Although winter conditions and snow cover would likely reduce or preclude contact by hare with soil (for ingestion) during winter months, it was conservatively assumed that soil would be available for ingestion throughout the entire year.

No data were located on the rate of water intake by the snowshoe hare. However, based on body weight and metabolic rate considerations, water intake for mammals can be estimated from the following equation (Sample et al., 1997):

$$\text{Daily water intake (L)} = 0.099 \times W^{0.90}$$

where, W = body weight (kg)

Based on this allometric equation, average adult water intake requirements for snowshoe hare (average body weight = 1.6 kg) are estimated to be approximately 0.15 L of water per day.

3.4 Lynx (Lynx Canadensis)

Within the Northwest Territories, lynx (*Lynx canadensis)* are found below the tree line and are most numerous in the southwest and in the Mackenzie Delta (Berrie et al., 1994; GNWT-RWED, undated-b). Although other prey are eaten, lynx depend heavily on snowshoe hares to thrive. As a result, the lynx populations fluctuate with population cycles of the snowshoe hare. Lynx are medium-sized animals, with the adults weighing an average of 10 kg for males and 8.5 kg for females (GNWT-WRED, undated-b).

No data were located on the rate of food ingestion by the lynx, specifically. However, based on body weight and metabolic rate considerations, food intake for placental mammals can be estimated from the following equation (Sample et al., 1997):

$$\text{Daily food intake (kg dry weight)} = 0.0687 \times W^{0.822}$$

where, W = body weight (kg)

Based on this allometric equation, average food intake requirements for adult lynx (average adult body weight = 9.25 kg) are estimated to be approximately 430 g of food (dry weight) per day. Given that rabbit meat is approximately 68% water (Sample et al., 1997), this corresponds to a fresh weight food intake rate of 1,340 g of rabbit (fresh weight)/day.

Animals are known to ingest soil during grooming, feeding and other activities (Sheppard, 1995). However, there are no data specifically on the rate of soil ingestion in the lynx. Data on soil ingestion by wild animals are limited and generally relate to herbivorous or omnivorous animals which ingest soil deposited onto the plants being consumed (Sheppard, 1995). Soil ingestion by predatory species is more likely due to grooming and mouthing of unwashed objects. Data for two predatory mammals (raccoon, red fox) indicate that soil ingestion, as determined from studies of faeces, corresponds to 3% to 9% of food intake on a dry mass basis (Beyer et al. 1994). The average dry food intake by the lynx is 430 g (of rabbit, dry weight) per day. At a soil intake rate of 3% to 9% of dry food intake, then the lynx would be expected to ingest 13 to 38 g of soil per day. This estimate is slightly higher than that reported for a domestic dog, that ranged from 10 to 20 g/day (Calabrese and Stanek, 1995). Lacking any other data, it was assumed that the lynx would inadvertently ingest 38 g of soil per day.

Although winter conditions and snow cover would likely reduce or preclude contact by lynx with soil (for ingestion) during winter months, it was conservatively assumed that soil would be available for ingestion throughout the entire year.

No data were located on the specific rate of water intake by lynx. However, based on body weight and metabolic rate considerations, water intake for mammals can be estimated from the following equation (Sample et al., 1997):

Daily water intake (L) = $0.099 \times W^{0.90}$

where, W = body weight (kg)

Based on this allometric equation, average adult water intake requirements for lynx (average body weight = 9.25 kg) are estimated to be approximately 0.73 L of water per day.

4.0 PATHWAY ANALYSIS FOR SELECTED ECOLOGICAL RECEPTORS

The equations used to estimate the daily intake of arsenic by selected ecological receptors are presented in Table 4.1.

The concentration of arsenic in soil was characterized as the average concentration derived from all relevant soil sampling surveys conducted since 1987. These data are summarized in Table 4.2. Spruce grouse and snowshoe hare are routinely observed within and near the city limits. Therefore, soil arsenic concentration data collected from urban areas as well as unoccupied and wild lands were considered equally relevant to the risk assessment. However, soil samples containing arsenic concentrations in excess of 1,000 µg/g were excluded, as most of these samples were located on the Giant and Con mine sites and these industrial locations will be the subject of specific, independent risk assessments and/or remediation.

Data relating to arsenic concentrations in local surface water following closure of the Giant Mine ore roasting facility were not available. However, two recent studies of water samples collected while the ore roaster was still operational were available (Mace, 1998; Jackson, 1998). These studies collected samples from waters with wide variations in their degree of industrial arsenic impact (effluent receiving waters, tailings ponds, proximity to contaminated sediments, etc.). As a result, individual measurements of arsenic concentration ranged from < 0.3 µg/L to 11,236 µg/L. Given the limited spatial distribution of sampling efforts and uncertainties regarding the frequency of wildlife visits to the various surface water areas for purposes of water intake, a simple weighted average water concentration was derived from the data presented in both reports. More data is always better. This approach gives greater statistical importance to data collected as part of the larger survey that sampled water from more locations, giving weight to the larger body of data. The resulting grand average was employed for estimating wildlife arsenic intake via water ingestion. These pre-closure water quality data are summarized in Table 4.2.

Other information required to estimate arsenic exposure in wildlife species are presented in Table 4.3.

4.1 Soil-to-Plant Accumulation of Arsenic

In order to estimate the dose of arsenic to herbivorous wildlife (hares, spruce grouse) the concentration of arsenic in plants that serve as food for these species must be estimated. Levels of arsenic in plants are generally greater in those grown on contaminated soil, compared to plants grown in uncontaminated areas (EC, 1999). However, arsenic does not bioaccumulate in plants and plant-borne concentrations are always less that the soil concentration. In general, roots contain higher levels than stems, leaves or fruit (O'Neill, 1995). Limited data were available regarding the concentrations of arsenic in local Yellowknife vegetation that could be used to estimate the likely concentration in plants might serve as foods for spruce grouse and hare. That data, collected by Koch et al. (2001), indicated an overall average concentration of arsenic in backyard garden produce of 0.070 µg/g fresh weight (0.7 µg/g on a dry weight, assuming an average 90% water content of produce (USEPA, 1996)) based on a total of 55 produce samples collected from 10 backyard gardens located in and around Yellowknife. With an average soil arsenic concentration in those gardens of 68.4 µg/g dry weight, the data suggested a site-specific soil-to-plant (dry weight) accumulation factor of 0.010.

Published reports of arsenic accumulation rates from soil to plants are summarized by the U.S.EPA (1996). On a produce dry weight basis, the overall geometric mean soil-to-plant accumulation factor was 0.011, a value more or less equivalent to that determined from the Yellowknife data of Koch et al. (2001). Therefore, it was assumed that the average concentration of arsenic in plants was 0.01 times the average soil-borne concentration.

Although Koch et al. (2000) report concentrations of arsenic in grasses and shrubs from the Yellowknife area, no data on the concentration of arsenic in the underlying soil were reported, thus precluding these data from determining a soil to plant uptake factor.

4.2 Accumulation of Arsenic in Prey of Goshawk and Lynx

In order to evaluate the overall intake of arsenic by the lynx and the goshawk, it was necessary to estimate the likely steady-state tissue (meat) concentration of arsenic in their prey. Lacking direct recent data on the arsenic concentration in tissues of locally-hunted hare or grouse, arsenic tissue concentrations were necessarily derived from ingestion-to-tissue biotransfer coefficients derived from available published data.

The hare will be exposed to arsenic through inadvertent ingestion of soil, through consumption of contaminated vegetation and through ingestion of surface water (see section 3.1.3, above). To estimate the concentration of arsenic in the meat of hares, biotransfer factors from oral intake to meat (BFT_{Meat}) were applied. Biotransfer factors for arsenic from oral intake to the meat of snowshoe hares (the primary food of the lynx), or other rabbit species, have not been published. Lacking data specific to the prey species of interest, biotransfer factors for other herbivorous species may be adopted and applied in a screening risk assessment (Sample et al., 1997).

The only specifically-published biotransfer factor for arsenic from oral intake to meat for herbivorous animals relate to those published for cattle. Stevens (1992) examined the bioaccumulation of arsenic in bovine tissues following oral exposure. Very little tissue specificity was observed for arsenic. The steady-state daily oral dose-to-tissue biotransfer factor (the ratio of the concentration of arsenic in tissue, in units of mg/g, to the daily oral dose of arsenic, in units of mg/d) was calculated to be 1.3×10^{-6} d/g, 2.5×10^{-6} d/g, and 2.1×10^{-6} d/g, for muscle, liver, and kidney, respectively.

The biotransfer factor for arsenic from steady-state oral intake to whole body (carcass) of meadow voles may be estimated from data presented by Pascoe et al. (1994). Using arsenic concentration data in voles, soil, water and vegetation as described by Pascoe et al. (1994) along with body weight, food intake, water intake and soil intake rates for voles presented by those authors, a biotransfer factor of 3.3×10^{-4} d/g was derived. This assumed that below ground plant parts (roots) composed 10% of voles' dietary intake, and above ground parts composed 90% (based on review of dietary information presented in USEPA, 1993), and also assumed that forbes and grasses comprised equal proportions of the diet .

Given that the biotransfer factor derived for voles from data of Pascoe et al. (1994) is more conservative than the value published for cattle by Stevens (1992), the value for voles was applied herein to estimate the tissue concentration of arsenic in hares.

Data to permit the derivation of arsenic intake-to-tissue biotransfer factors specific to avian species could not be located. Therefore, the same biotransfer factor of 3.3×10^{-4} d/g was used to estimate steady state tissue concentrations in spruce grouse for purposes of estimating arsenic intake by goshawks due to predation on spruce grouse.

5.0 TOXICOLOGICAL BENCHMARK DOSES FOR MAMMALIAN AND AVIAN RECEPTORS

Data on the toxicity of arsenic to the wildlife species of interest in an ecological risk assessment are rarely available. Such is the case here. Toxicological studies of arsenic in the mammalian wildlife species being considered could not be located. Therefore, benchmark doses for these species were extrapolated from published toxicological studies on other mammalian and avian species, according to the methods of Sample et al. (1996). The BMDs employed herein are presented in Table 3.1.

With the exception of snowshoe hare, all benchmark doses were based on no-observed-adverse-effect-levels (NOAELs) in the most sensitive mammalian or avian species tested, rather than on other higher doses known to be associated with toxicological effects. For snowshoe hare, a NOAEL dose determined from a toxicological study in rabbits was considered to be more relevant for risk assessment purposes than an extrapolated NOAEL value from another species.

The benchmark doses employed herein are anticipated to be free of harmful effects. NOAEL-based benchmarks are used in Screening Assessments because they are conservative and represent maximum concentrations that are believed to be non-hazardous. Therefore, exceeding a NOAEL-based benchmark does not suggest that adverse effects are likely; rather it only indicates that contamination is sufficiently high to warrant further investigation (Sample et al., 1996).

5.1 Mammals

Where appropriate toxicological studies specific to the wildlife species of interest are lacking, Sample et al. (1996) provide methods of estimating appropriate benchmark doses for the species of interest from studies done on other mammalian species. Numerous physiological functions in mammals, including responses to toxic chemicals, are a function of body size. Based on known body size scaling factors, Sample et al. (1996) determined that the NOAEL dose in the wildlife species of interest can be derived from the NOAEL in a known test species by the following relationship:

$$NOAEL_W = NOAEL_T \times [(BW_T / BW_W)^{1/4}]$$

where, $NOAEL_W$ = estimated NOAEL dose in the wildlife species of interest;

$NOAEL_T$ = known NOAEL dose in the test species;

BW_T = body weight of test species;

BW_W = body weight of wildlife species of interest.

The lowest reported chronic no-observed-adverse-effect-level (NOAEL) dose for a mammalian species other than humans (see summary of mammalian toxicity studies in ATSDR, 2000) is 0.6 mg/kg-d (Schroeder et al., 1968). That oral dose, delivered in drinking water, produced no effects on respiratory, cardiovascular, hepatic, renal, or dermal organ systems, nor caused any detectable change in body weight of exposed Long-Evans rats.

5.1.1 Lynx

Based on the allometric equation of Sample et al. (1996), and the rat NOAEL of 600 µg/kg-d from the study of Schroeder et al. (1968; see also ATSDR, 2000), the average body weights of rats (0.35 kg; Sample et al., 1996) and lynx (9.25 kg), a chronic NOAEL benchmark dose for lynx can be derived as 265 µg/kg-d.

5.1.2 Snowshoe hare

Based on the allometric equation of Sample et al. (1996), and the rat NOAEL of 600 μg/kg-d from the study of Schroeder et al. (1968; see also ATSDR, 2000), the average body weights of rats (0.35 kg; Sample et al., 1996) and snowshoe hare (1.6 kg), a chronic NOAEL benchmark dose for hares can be derived as 410 μg/kg-d. However, short-term studies of rabbits, specifically investigating the developmental effects caused by arsenic exposure during gestation, have reported a NOAEL for developmental effects of 370 μg/kg-d (Nemec et al., 1998; see also ATSDR, 2000). Given that the developmental effect NOAEL from studies specifically of rabbits is lower than the estimated chronic BMD for snowshoe hare, the NOAEL for developmental effects of 370 μg/kg-d was selected as the BMD for characterization of risks posed to snowshoe hare by arsenic contamination in and around Yellowknife, NWT.

5.2 Birds

Data and methods to establish benchmark doses for metals in avian species have also been prescribed by Sample et al. (1996). From a review of available toxicological investigations of As in bird species, the study by the USFWS (1964) in which sodium arsenite was administered orally to mallard ducks in the diet was identified by Sample et al. (1996) as the most appropriate basis for deriving BMDs for other avian species for which no direct toxicological studies have been carried out. The study of the USFWS (1964) investigated mortality. The chronic NOAEL dose in mallard ducks was determined to be 5135 μg/kg-d (Sample et al., 1996).

The effects of contaminants in avian species appear to be relatively constant across bird species of varying body weights (Sample et al., 1996). Mineau et al. (1996) determined that the average body size scaling factor for LC_{50} data for 37 pesticides in bird species was 1.15, and scaling factors for the majority of the chemicals evaluated (29 of 37) were not significantly different from 1. Therefore, it is recommended that the NOAEL doses for test avian species be applied to other (wildlife) bird species with no body size scaling adjustment (Sample et al., 1996). Based on the NOAEL in mallard ducks of 5135 μg/kg-d, the estimated NOAEL for goshawks and for spruce grouse is also 5135 μg/kg-d.

6.0 RESULTS, ECOLOGICAL RISK CHARACTERIZATION AND DISCUSSION

Estimated daily doses received by each of the four selected species, and subsequent Hazard Quotients (with respect to the benchmark doses in Table 3.1), are presented in Table 6.1. The estimated exposures in both avian species were below the estimated benchmark doses for those species. However, estimated exposures in mammalian species were between 2.4 (lynx) and 2.6 (hare) times greater that their estimated BMDs.

It should be noted that the benchmark doses (BMDs) employed herein for risk characterization were based on no-observed-adverse-effect levels. Generally speaking, doses known or observed to have some deleterious effect in test species, termed lowest-observed-adverse-effect levels (LOAEL), may be anywhere from 2 to 10 times greater than NOAEL doses. The true threshold for chemically-induced toxic effects in animals lies between the NOAEL and LOAEL. Therefore, the degree of actual harm to lynx or hares, if any, presented by exposures exceeding the NOAEL-based BMDs, is not known at this time.

7.0 UNCERTAINTY IN THE ECOLOGICAL RISK ASSESSMENT

Screening level ecological risk assessments present a variety of uncertainties and limitations. The lack of adequate data pertaining specifically to arsenic concentrations in the vegetation consumed by hares or spruce grouse, and arsenic levels in hare and spruce grouse tissues that serve as food for lynx and goshawks, are major limitations within this study.

The accuracy and validity of exposures estimated herein for lynx and goshawks due to the consumption of contaminated prey are highly uncertain. No data were available on the concentration of arsenic in the tissues of hare collected from the Yellowknife area. The use of oral intake to tissue biotransfer factors for the meadow vole, to estimate arsenic concentrations in the meat of snowshoe hare, may significantly under- or over-estimate the actual hare tissue concentration, thus leading to errors in prediction of actual risks posed to lynx.

Likewise, the application of a biotransfer factor derived from a mammalian species (meadow vole) to estimate tissue concentrations in spruce grouse may have significantly under- or over-estimated the concentration of arsenic in the tissues of that avian species. No data could be located on the concentration of arsenic in tissues of spruce grouse collected from the Yellowknife area to confirm predictions. The average concentration of arsenic in the muscle tissue of 9 willow ptarmigan (a bird species of similar weight and dietary habits as the grouse) that were collected from the Yellowknife area in 1975 and 1976 (Gemmill, 1977), when the Giant Mine ore roaster was still operational, was 0.45 μg/g fresh weight, a value approximately 2 times greater than the 0.2 μg/g estimated herein for grouse.

The toxicological benchmark doses (BMDs) are also uncertain. Owing to a lack of BMD values published specifically for the species of interest herein, BMDs were necessarily estimated from NOAELs of other species. However, inter-species trends in toxicity do not always follow precisely the allometric relationships recommended by Sample et al. (1996) to estimate those BMD values.

Also with respect to relevant BMD values, the animal populations extant in the Yellowknife area have evolved, and will likely have adapted, to the relatively high natural arsenic levels that exist due to geologic sources. Various physiological responses, such as metallothionein induction, have evolved to aid in the detoxification of metals. Metallothionein production and cellular levels are increased by arsenic exposure in mammals (Albores et al., 1992). The toxicity of arsenic is reduced in rodents expressing metallothionein production compared to animals void of metallothionein (Liu et al., 2000), and there is decreasing arsenic toxicity with increasing metallothionein levels (Park et al., 2001). Also, elevated chronic arsenic exposure appears to result in an irreversible (phenotypic) increase in arsenic tolerance unrelated to metallothionein levels or induction (Romach et al., 2000).

Therefore, it is likely that mammalian populations in this area, given their elevated chronic arsenic exposure, will have greater resistance to arsenic toxicity that would rodents exposed in a laboratory bioassay. Such laboratory rodents will have been maintained in an environment virtually free of arsenic prior to bioassay exposure and would, therefore, likely be more susceptible to toxic effects than Yellowknife-area wild populations. Such local resistance to arsenic toxicity could only be determined through appropriate studies employing animals collected from the Yellowknife area.

The risk assessment presented herein conservatively assumed that all (100%) of the arsenic consumed from soil, water or food would be absorbed by the mammals and birds considered. It is likely, however, that less than 100% of that arsenic is bioavailable. For example, the Risk Assessment Information System (RAIS) maintained by the Oak Ridge National Laboratory in the United States (ORNL, undated) suggests a standard assumption for oral bioavailability of only 41%, less than half that assumed herein. Therefore, the estimates of total dose for each wildlife species considered may have over-estimated exposure by a large margin if that arsenic was less than completely bioavailable.

It was also conservatively assumed that winter conditions and snow cover would not reduce or preclude contact by animals with soil or surface water (for ingestion) during winter months. However, activity patterns, including possible hibernation, and changes in activity and diet following deposition of snow, would likely reduce the amount or rate of arsenic intake during winter months.

Finally, alternate methods do exist for estimating average or typical concentrations of arsenic in soil, surface water, plants and prey animals. Use of those alternate methods, although no more or less valid than the methods employed herein, would alter the determination of estimated daily intakes of arsenic by the species considered.

8.0 SUMMARY, CONCLUSIONS AND RECOMMENDATIONS

Based on the foregoing assessment, it was determined that goshawks and spruce grouse received a daily dose of arsenic that is less than one quarter of the estimated toxicological benchmark dose for those species. It is also anticipated that other avian species are also within safe limits of exposure, given that the BMD for avian species is considered to be constant across species.

Estimated arsenic exposures in lynx and snowshoe hare exceeded their respective BMD values by factors of 2.4 and 2.6, respectively. However, whether or not these exposures present a true risk to Yellowknife mammalian wildlife can not be determined at this time. First, chronic toxicological studies of oral arsenic exposure have not been conducted specifically for the animal taxa of interest (felids, rabbits/hares). Second, BMD values do not represent a distinction between health and disease, but are based on doses known to have no effects in the laboratory species studied. The BMD values are based on no-observed-adverse-effect-levels (NOAEL) that are some unknown degree lower than the actual threshold for toxic effects. Third, the NOAEL used to estimate the BMDs for lynx was taken from laboratory bioassay study of laboratory rats, and the inter-species variability in arsenic toxicity is not well known. Finally, owing to the elevated arsenic exposure that will result from the natural geologic arsenic that occurs in the Yellowknife area, wildlife in that area will likely have a significant resistance to arsenic toxicity, relative to laboratory animals reared and maintained (prior to bioassay) under near arsenic-free conditions.

Based on the foregoing, and in particular due to the anticipated arsenic resistance in local wildlife, it is not anticipated that the estimated exposures in lynx and hare are toxicologically significant. However, further investigation would be required to confirm this conclusion.

In order to further refine the estimates of arsenic exposures and risks for ecological receptors in the Yellowknife area, the following recommendations could be considered:

1. survey arsenic levels in vegetation that serves as food for hares and spruce grouse (and/or other herbivorous species);

2. survey arsenic levels in the tissues of hare and spruce grouse (and/or other prey species) that serve as food for lynx and goshawks (and/or other predatory mammals and birds);

3. survey population density, age structure, fecundity, and other population dynamics of wildlife populations in the Yellowknife area to determine if those populations are healthy relative to similar populations elsewhere in Canada where arsenic levels are not naturally elevated.

9.0 CLOSURE

This report has been prepared for the exclusive use of the Yellowknife Arsenic Soils Remediation Committee (YASRC). The conclusions in this report are based on the results of a screening level ecological risk assessment, performed using limited site-specific information. As is routine for such risk assessments, it is considered to be conservative in nature, owing to the use and application of assumptions designed to ensure that exposures are not under-estimated. It is limited to arsenic, and to the species and exposure pathways specifically referenced herein.

It should be noted that risk assessments of the type presented herein quantify potential risks based on average or typical receptor characteristics, with toxicity benchmark doses extrapolated from other species studied under laboratory conditions. Therefore, the absence of effects or risks in all individuals of an exposed animal population can not be guaranteed under all circumstances.

The services performed in the preparation of this report were conducted in a manner consistent with the level of skill and care ordinarily exercised by risk assessors practicing under similar conditions.

We trust that the enclosed is satisfactory for your present requirements. If you should have any questions or comments, please contact the undersigned.

Respectfully submitted,

RISKLOGIC SCIENTIFIC SERVICES INC.

G. Mark Richardson, Ph.D.

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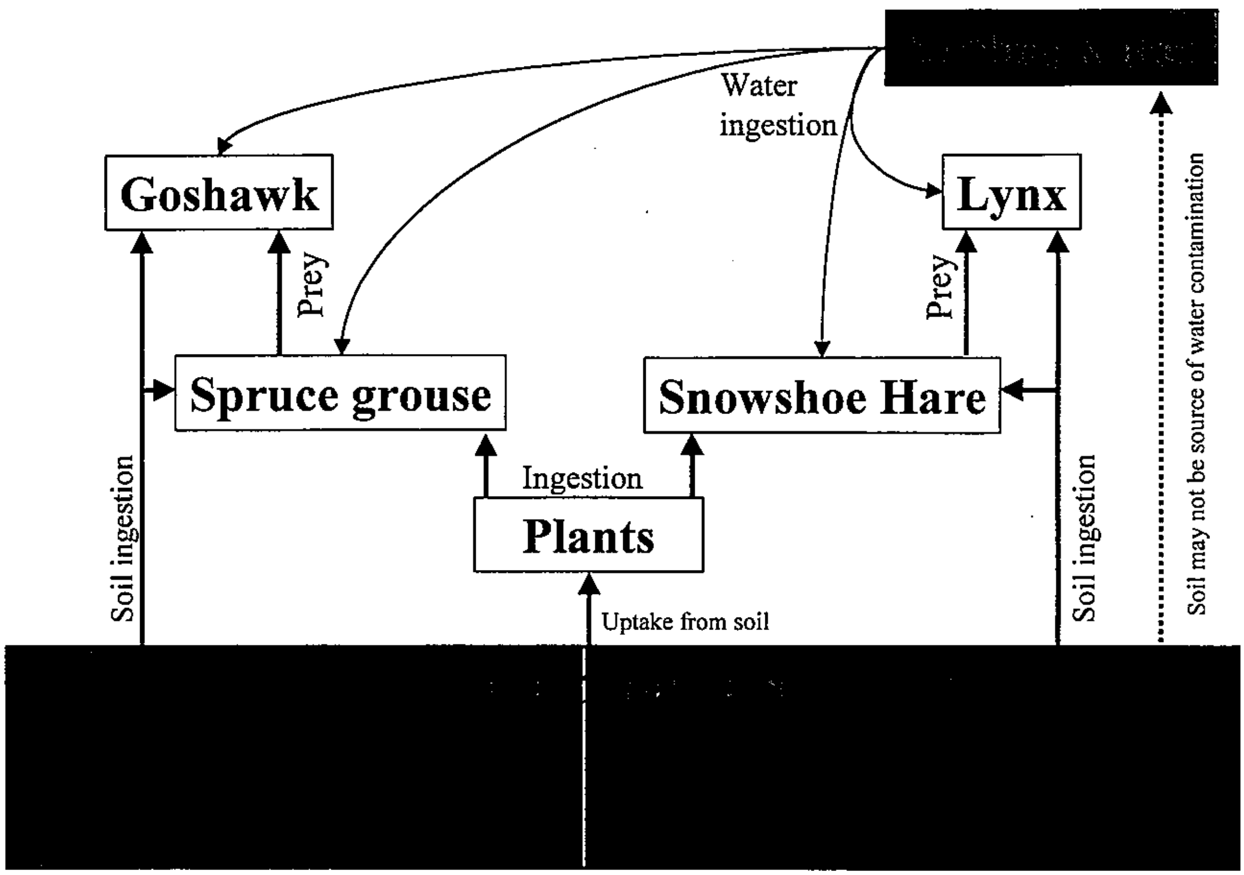

Figure 2.1
Conceptual Model for Assessment of Ecological Risks of Arsenic in Yellowknife Area Soils

TABLE 3.1

SUMMARY OF ASSUMED CHARACTERISTICS AND FACTORS FOR ECOLOGICAL RECEPTORS

Receptor Characteristic	Symbol	Units	Spruce grouse	Goshawk	Snowshoe hare	Lynx	Reference
Body Weight	BW	kg	0.5	0.95	1.6	9.25	see text, Section 3
Soil Ingestion Rate	IR_S	g/d	4	6	10	38	see text, Section 3
Water Ingestion Rate	IR_W	L/d	0.04	0.06	0.15	0.73	Calculated, according to Sample et al. (1997)
Vegetation (Food) Ingestion Rate	$IR_{F\text{-}veg}$	g (dry wt)/d	40	n/a	125	n/a [1]	Calculated, according to Sample et al. (1997)
Meat (Food) Ingestion Rate	$IR_{F\text{-}meat}$	g (fresh wt)/d	n/a [1]	175	n/a	1340	Calculated, according to Sample et al. (1997)
Relative Absorption Factor via the Gastro-intestinal tract	RAF_{GI}	unitless	1	1	1	1	Assumed
Arsenic Benchmark Dose	BMD	µg/kg-d	5135	5135	370	265	Calculated, according to Sample et al. (1996); see Section 5

1 n/a = not applicable; pathway or source does not exist for the particular receptor.

TABLE 4.1

EQUATIONS USED TO ESTIMATE DOSES

INADVERTENT INGESTION OF SOIL

The predicted intake of arsenic via soil ingestion is calculated as:

$$Dose(\mu g / kg - d) = [C_S \times IR_S \times RAF_{GI} \times D_1] / BW$$

Where:

C_S	=	concentration of element in soil (µg/g)
IR_S	=	receptor soil ingestion rate (g/d)
RAF_{GI}	=	relative absorption (bioavailability) factor for gastrointestinal exposure (unitless)
D_1	=	duration of exposure (yr/yr)
BW	=	body weight (kg)

INGESTION OF CONTAMINATED FOOD

The predicted intake of arsenic via ingestion of food is calculated as:

$$Dose(\mu g / kg - d) = C_F \times IR_F \times RAF_{GI} / BW$$

Where:

C_F	=	concentration of element in animals' food
	=	$C_{F\text{-veg}}$ for plants (µg/g dry weight)
	=	$C_{F\text{-meat}}$ for animal tissue (µg/g fresh weight)
IR_F	=	food ingestion rate
	=	$IR_{F\text{-veg}}$ for herbivores (g dry weight/d)
	=	$IR_{F\text{-meat}}$ for carnivores (g fresh weight/d)
RAF_{GI}	=	relative absorption (bioavailability) factor for ingestion exposure (unitless)
BW	=	body weight (kg)

INGESTION OF CONTAMINATED WATER

The predicted intake of arsenic via ingestion of water is calculated as:

$$Dose(\mu g / kg - d) = C_W \times IR_W \times RAF_{GI} / BW$$

Where:

C_W	=	concentration of element in animals' drinking water (µg/L)
IR_W	=	water ingestion rate (L/d)
RAF_{GI}	=	relative absorption (bioavailability) factor for ingestion exposure (unitless)
BW	=	body weight (kg)

TABLE 4.2
SUMMARY DATA ON ARSENIC IN YELLOWKNIFE AREA SOILS AND SURFACE WATERS

MEDIUM	UNITS	MEAN	RANGE	SAMPLE SIZE (N)	DATE SAMPLED	DATA SOURCE
Yellowknife Soil [1]	µg/g	122	3.5 - 1570	401	1987 - 2001	See RSSI (2001a), Section 4.2.1
Trailer Court Soil [1]	µg/g	404	4.0 - 4950	70	1987 - 2001	See RSSI (2001a), Section 4.2.2
Wilderness Soil [1,2]	µg/g	117	4.0 - 900	928	1987 - 2001	See RSSI (2001a), Section 4.2.3
Weighted mean from all soil data	µg/g	133	3.5 - 4950	1399	1987 - 2001	
Surface water (mean)	µg/L	1638.7	1.3-11236	14 (sites)	1997	Mace, 1998
		31.3	<0.3 - 315	85 [3]	1992-1994	Jackson, 1998

1. Surface area to which data relate are described in detail by RSSI (2001a).

2. Excludes a small number of samples with obvious industrial impact (1000 ≤ [As] ≤ 10000, n = 32)

3. From 15 to 18 different sampling dates at 5 locations

TABLE 4.3

SUMMARY OF SITE-SPECIFIC VARIABLES AND ASSUMPTIONS USED FOR THE RISK ASSESSMENT

Variable	Symbol	Units	Value	Reference
Average Soil Concentration	C_S	µg/g	133	see Table 4.2
Average Surface Water Concentration	C_W	µg/L	259	Weighted average from data in Table 4.2
Average Concentration in Plants that serve as food	$C_{F\text{-veg}}$	µg/g (dry weight)	1.33	Calculated [1]
Concentration in Prey Species Tissue that serves as food	$C_{F\text{-meat}}$	µg/g (fresh weight)	0.20 (spruce grouse) 0.51 (hare)	Calculated [2]
Duration of Exposure	D_1	yr/yr	1	Assumed

1 See text, Section 4.1; $C_{F\text{-veg}} = C_S \times 0.01$ (0.01 = accumulation factor for arsenic from soil to vegetation (dry weight))

2 See text, Section 4.2; $C_{F\text{-meat}}$ = estimated total daily oral dose for hare or spruce grouse (µg/kg-d) x biotransfer factor from oral dose to meat (3.3×10^{-4} d/g) x body weight of hare or spruce grouse (kg)

Table 6.1

ESTIMATED DOSES AND HAZARD QUOTIENTS FOR SELECTED ECOLOGICAL RECEPTORS, YELLOWKNIFE, NWT

Pathway	units	Spruce grouse	Goshawk	Snowshoe hare	Lynx
Soil Ingestion	µg/kg-d	1064	840	831	546
Food consumption	µg/kg-d	106	36	104	73
Water ingestion	µg/kg-d	21	16	24	20
TOTAL INTAKE	µg/kg-d	1191	892	959	639
HQ [1]	unitless	0.23	0.17	2.6	2.4

1 HQ = TOTAL INTAKE/BMD; BMD values for each species listed in Table 3.1