ASSESSMENT OF RADIATION EFFECTS ON BIOTA IN PROXIMITY TO URANIUM MINING AND MILL SITES IN CANADA: FIELD OBSERVATIONS AND MODEL PREDICTIONS

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ÉVALUATION DES EFFETS DU RAYONNEMENT SUR LE BIOTE SITUÉ PRÈS DES MINES D'URANIUM ET DES USINES DE TRAITEMENT CONNEXES AU CANADA : OBSERVATIONS EN MILIEU ET PRÉDICTIONS DE MODÈLES

RÉSUMÉ

Nombre d'efforts ont été consacrés à l'élaboration d'études sur le terrain et en laboratoire, de modèles de voies environnementales, et d'évaluations du risque écologique afin d'identifier et d'évaluer les impacts possibles de l'exploitation des mines d'uranium sur les gens et sur l'environnement. En général, les résultats démontrent qu'il y a très peu d'impact différentiel, observé ou calculé, sauf pour le biote vivant au milieu de zones de gestion de résidus d'extraction minière. De plus, il n'y a aucun impact significatif sur le biote, tant au niveau de la population qu'au niveau de la collectivité, près des sites d'exploitation minière ou des usines de traitement d'uranium concasseur.

L'expérience acquise démontre l'avantage d'une exploitation judicieuse du caractère complémentaire des données et des modèles lors de l'élaboration d'un programme de surveillance d'impacts écologiques potentiels. Notamment, l'efficacité de la surveillance environnementale peut être améliorée par une boucle de réaction des prédictions de modèles dans le programme de suivi.

ABSTRACT

Considerable effort has been devoted to identifying and evaluating potential impacts from uranium mining on people and the environment. This includes field and laboratory experiments as well as pathways modelling and ecological risk assessment. Studies to date generally indicate that unless biota reside within a tailings waste management area, there is little incremental ecological impact (observed or calculated). Furthermore, there are no significant population-level or community-level impacts on biota in the vicinity of uranium mining and milling operations.

The practical experience gained from these studies shows that it is advantageous to exploit the complementary nature of data and models in designing monitoring plans for potential ecological impacts. In particular, the effectiveness of environmental monitoring can be enhanced by providing a feedback loop from the modelling results to the monitoring plan.

1.0 INTRODUCTION

Canada has a long history of uranium mining with activity concentrated in northern Ontario and northern Saskatchewan (see Figure 1.1). Over time, considerable effort has been devoted to identifying and evaluating potential impacts from uranium mining on people and the environment (e.g. TAEM and SENES 1993, 1994, 1995, 1996). The focus of this paper is to provide an overview of the field and laboratory studies and modelling assessments that have been carried out to assess the environmental impacts of uranium mining and waste management. Potential impacts on biota represent the combined effects of exposure to radiation (through both external and internal pathways), toxic metals and various physical stressors. This paper focusses specifically on the (potential) effects of radionuclides on aquatic and terrestrial biota in proximity to uranium mining, milling and tailings sites in Canada.

The objectives of this paper are:

- To provide a brief and current survey of what has been done in both northern Saskatchewan and the Elliot Lake region of Ontario towards assessing the ecological impact of uranium mining and milling; and

- To provide a snapshot of the state-of-the-art in ecological risk assessment and its use as a decision tool with respect to the environmental management of mining and milling operations.

The present paper surveys ecological studies carried out with respect to radiation and biota with a particular focus on the areas highlighted in Table 1.1.

Table 1.1 Sites of Ecological Risk Studies Related to Radionuclides

Site	Data	Modelling
Saskatchewan		
Langley Bay	+	–
McArthur River	+	+
Rabbit Lake	+	–
Key Lake	+	+
Cigar Lake	+	+
McClean Lake	+	+
Midwest	+	+
Elliot Lake		
Stanleigh	+	+
Quirke, Panel, Denison, etc.	+	+

[ ] subject present paper.

Figure 1.1 Canada Uranium Mining

2.0 McARTHUR RIVER

Cameco Corporation (Cameco) has proposed to develop its McArthur River Project in northern Saskatchewan by underground mining and to provide mined ore to the Key Lake milling facility as a replacement for Key Lake ore. A screening level assessment was performed as a sequence of calculations, beginning with extremely conservative assumptions about exposure and progressing towards more realistic exposure scenarios. Contaminants which passed the initial screening using the highest expected exposure concentrations, which suggested minimal risk, were excluded from further consideration. Contaminants which failed the initial screening were then re-examined using more realistic exposures (based on available data). Contaminants that were not removed in this stage cannot be easily dismissed and require further study (SENES 1994; SENES 1995a).

Results showed that all radionuclides examined appear to pose minimal risk to aquatic and terrestrial VECs. However, results of the initial screening-level assessment of potential ecological risks associated with the McArthur River Project suggested that a more detailed ecological analysis be undertaken with respect to non-radioactive contaminants (SENES et al. 1994). Specifically, the value of the screening index for northern pike (*Esox lucius*) and lake whitefish (*Coregonus clupeaformis*) exposed to expected copper and cadmium concentrations exceeded one. While index values greater than one do not necessarily correspond to imminent risk, such values support taking the next step in a sequential approach in assessing ecological risks. The next step was to incorporate more biological detail into the assessment by developing a population model for these pike and lake whitefish, and then using the model to further assess potential ecological risks posed by copper and cadmium for selected lakes in the Project.

A probabilistic demographic model of a northern pike population was developed (SENES 1995b) to estimate the impacts of copper toxicity on the survival and reproduction of different pike life stages (age classes), as well as for the total population. The model was developed to assess the population dynamics of pike for a typical background lake in northern Saskatchewan and for lakes representative of the Key Lake and McArthur River areas. (Although the model was applied to non-radioactive contaminants, the same methodology could be applied to radionuclides, had the screening assessment indicated potential concern.)

The results of the pike model are consistent with the screening-level assessments and suggest that there may be ecological risks to northern pike in the Key Lake and McArthur River areas from copper and cadmium. It must be emphasized again that the demographic model results are projections. The model results appear compelling, yet the actual impacts of these metals on pike population dynamics cannot be directly assessed from these comparative projections. Furthermore, it should be noted that the assessments were based on concentrations expected in the first receiving water body downstream of the point of discharge. These concentrations decrease rapidly from the point of discharge due to dilution and any effects would likely be limited to the mixing zones.

The results of the population modelling argue for additional assessment that addresses some of the shortcomings of the approach as previously outlined. For example, attempts could be made to include seasonal variability in water temperature that may affect fish activity (Bartell 1990). Seasonal variation in exposure concentrations might also be explored along with exposure estimates that more closely approximate expected average exposure concentrations. More detailed characterization of the total dissolved metal concentrations may indicate that exposure concentrations overestimate biologically available copper or cadmium. Ecosystem complexity not explicit in the population model might be better addressed using an aquatic ecosystem model to assess risk (e.g. Bartell 1990; Bartell et al 1992).

The ecological risk assessment results were compared with biological field data for aquatic biota (Cameco 1995a). The comparisons of the ecological risk assessment versus biological data indicate that in many instances correlations were low. Evaluations of ecological risk predictions were mainly based on screening index values, as model of population-level risks were limited to northern pike. Screening index values are not direct estimates of the probability of ecological risk (or impact). Higher index values imply greater risks. The risk probabilities are generally conservative in approach as scenarios worse than actual are assumed (often worst case scenarios). The frequently poor correlations between screening index values and actual biological changes may be due to the fact that worst case scenarios did not occur.

3.0 RADON AND LONG-LIVED PROGENY

One of the radionuclides released by uranium mining activity is radon-222. It is an inert gas with a half-life of 3.82 days and produced from the radioactive decay of radium-226, as part of the uranium-238 decay series. The combination of its form (an inert gas) and its half-life allows it to readily escape from the uranium ore during the various stages of mining and milling. If measured on an activity basis, radon-222 comprises most of the radionuclides released by a uranium mine. For example, it is estimated that the Key Lake Operation emits about 1.4×10^{8} Bq/s of radon and about 3.2×10^{5} Bq/s of U-238 (Cameco 1996a). However, monitoring around uranium mines and theoretical calculations of the atmospheric dispersion of radon show that the concentration of radon drops to natural background levels within a few kilometres of the mines (Cameco 1995a). This is because of the relatively large amounts of radon that are naturally present in the environment.

One environmental pathway that has been the focus of ongoing assessments is that of air-to-lichen-to-caribou. This is because of the very wide dispersal of radon and its decay products into the environment and the importance of lichen as a food source to caribou (and the importance of caribou as a food source for some aboriginal communities). Of particular interest is lead-210 (Pb-210), one of the decay products of radon-222, which can be deposited onto and incorporated into the plant. In addition, Pb-210 decays into polonium-210 (Po-210), which has a relatively large dose conversion factor. Monitoring and modelling has shown that the potential dose to caribou and humans from this exposure pathway is a small fraction of the natural background dose (Cameco 1995a).

Lichens are a slow-growing, long-lived species and they can accumulate large quantities of radionuclides without themselves being affected (Snyder and Platt 1983). They derive nutrients primarily from aerially dispersed dust, water and snow. However, substrates to which they are attached may also be important. They have been commonly used as pollution indicators, especially for heavy metals and radionuclides. Lichens have been sampled around both the Rabbit Lake and Key Lake Operations since the early 1980's and there have been no temporal or spatial trends detected in the concentration of Pb-210 in lichen around the operations. An overview of the data for each site is discussed below.

To examine spatial trends of radionuclide concentrations in lichen around the Rabbit Lake Operation, several studies have been conducted along east-west and north-south transects centred around the Rabbit Lake pit (Cameco 1992). In particular, detailed sampling was done in 1980 and 1986. The lichen species *Cladina mitis* was the species collected in 1980 and 1986. Subsequently, a mixture of *Cladina mitis* and *Cladina stellaris* has been collected. These species belong to a group referred to as the "caribou lichens" and are common in the area (Vitt et al 1988, GMCL 1980). Figure 3.1 shows no trend in the Pb-210 concentration as a function of distance from the Rabbit Lake pit. Based on the results of the 1980 and 1986 sampling programs, routine sample locations were established and samples are taken every three years. Figure 3.2 shows that there has been no increase in the Pb-210 concentration in lichen around the Rabbit Lake Operation over the period 1980 to 1995.

Lichen sampling has been conducted around the Key Lake Operation at several routine sample locations since 1982, before the start of uranium ore mining. Two sample locations, Kapesin Lake and Douglas, have been consistently sampled and are about four to five kilometres south of the site. In 1982 the lichen species *Cladina stellaris* was collected and subsequent sampling has been a mixture of *Cladina stellaris* and *Cladina mitis*. The data for these locations is plotted in Figure 3.3. Similar to the data from the Rabbit Lake site, there has been no trend in either the Pb-210 or Po-210 concentrations in lichen around the Key Lake Operation.

Figure 3.1 Pb-210 Concentration in Lichens vs Distance: Rabbit Lake

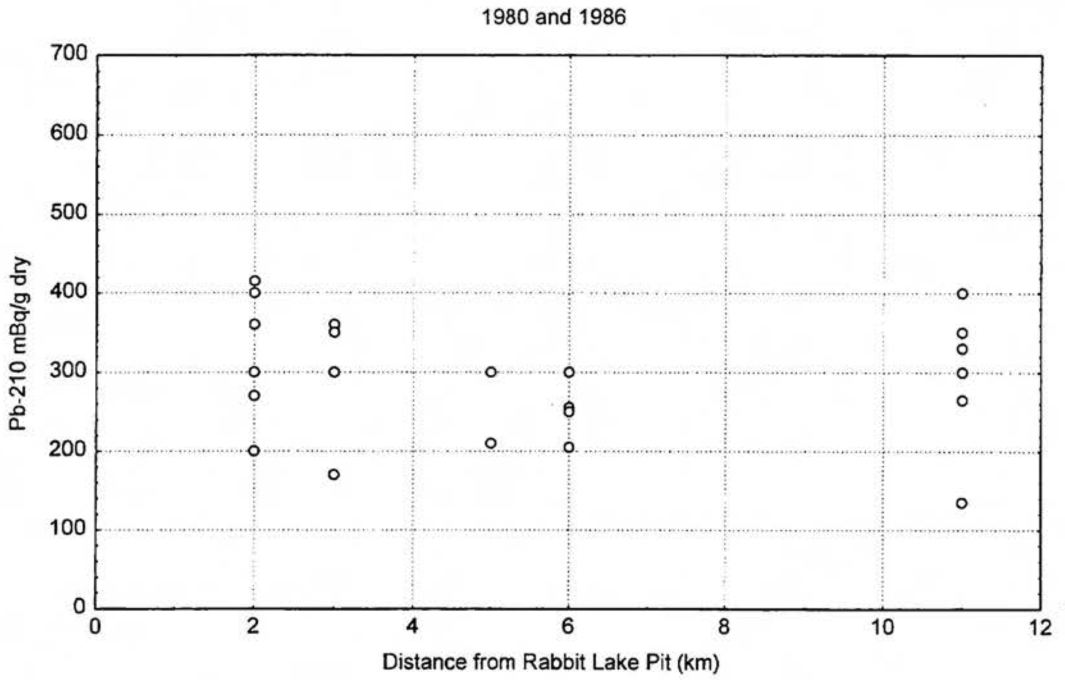


Figure 3.2 Pb-210 Concentration in Lichens vs Time: Rabbit Lake

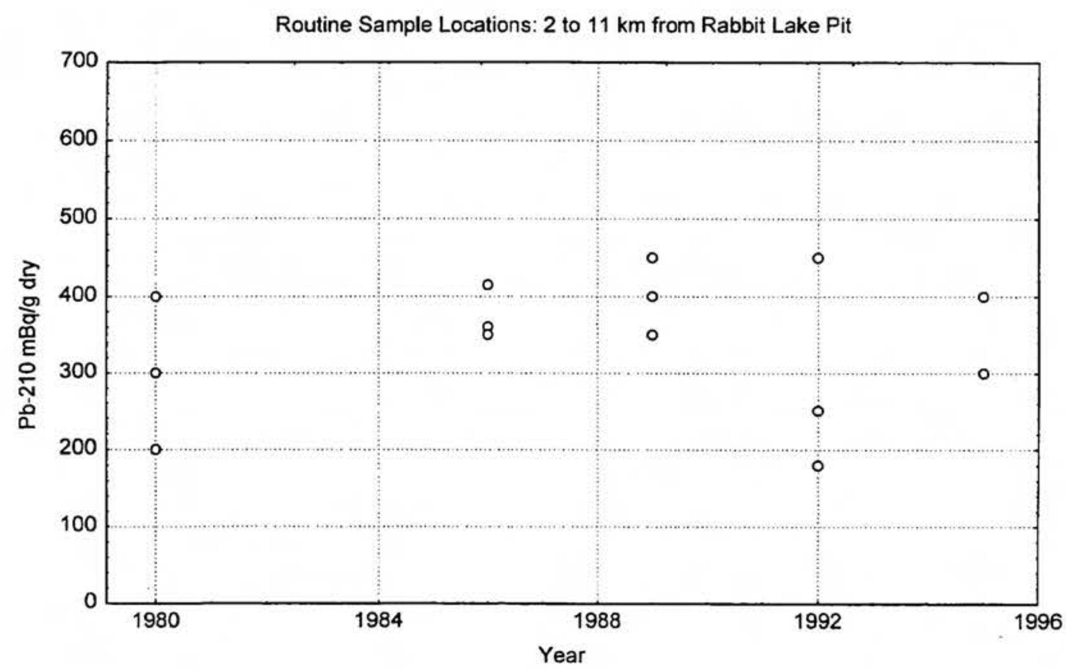
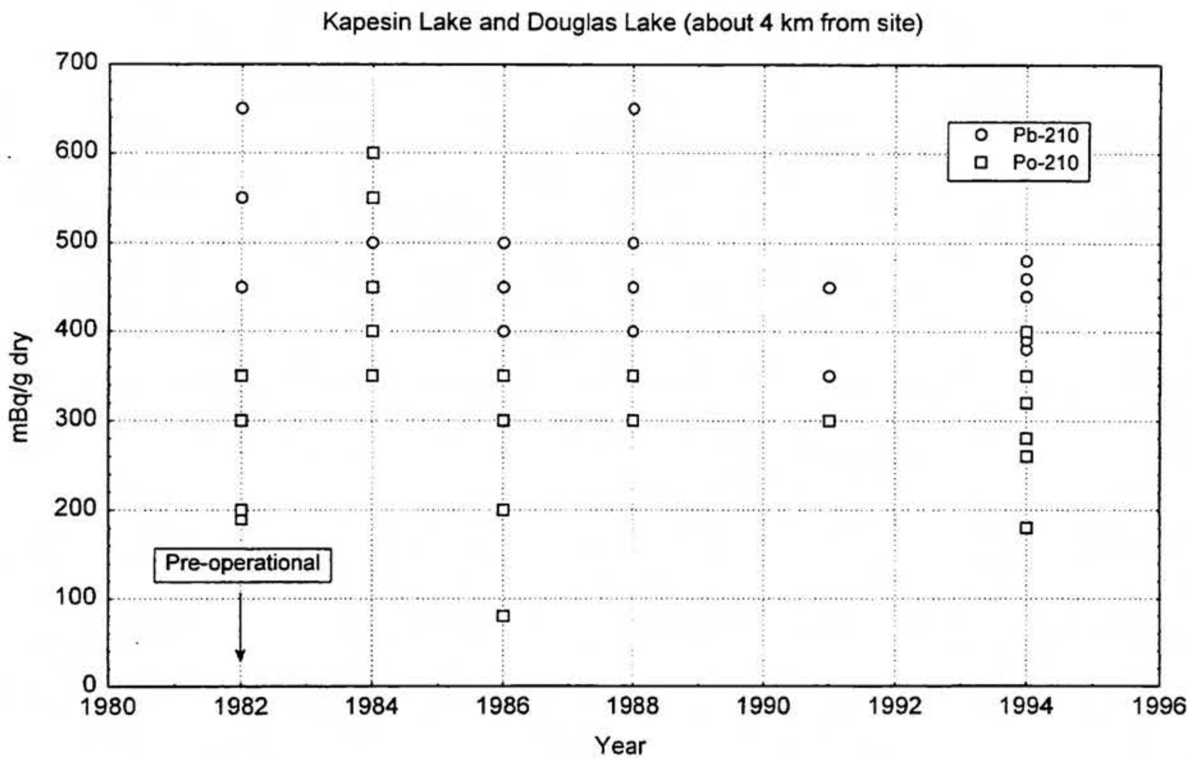


Figure 3.3 Pb-210 and Po-210 Concentrations in Lichens vs Time: Key Lake Facility

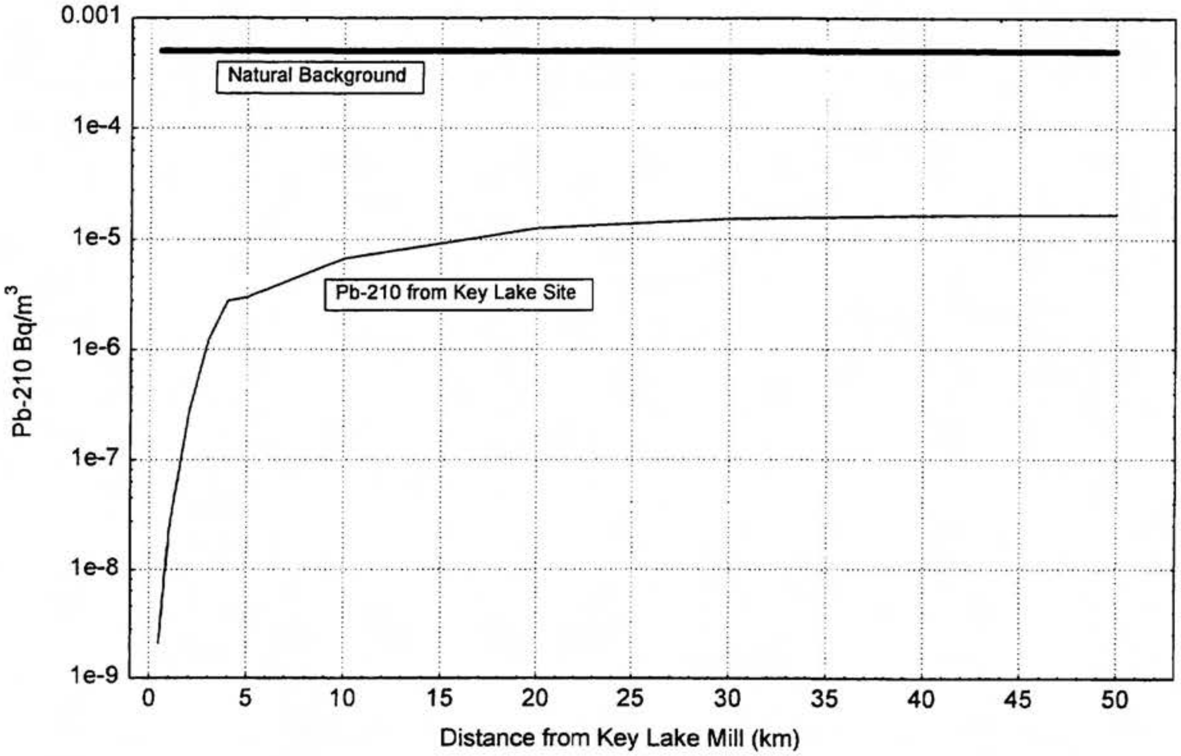


Monitoring data around the Key Lake and Rabbit Lake Operations shows that there have been no spatial or temporal trends in the concentration of Pb-210 in lichen. In addition, the concentrations of Pb-210 in the lichen from sampling programs conducted around both sites are within the range reported by Sheard et al (1988) for a variety of locations in northern Saskatchewan and the Northwest Territories.

The fact that the uranium mines do not have a measurable impact on Pb-210 concentrations in lichen is supported by modelling results of the atmospheric dispersion of radon. Modelling of emissions from the Key Lake Operation shows that the concentration of radon reaches background levels within about 10 km from the site (Cameco 1995a). These modelling results, which have been validated by field data were used to predict Pb-210 concentrations around the site.

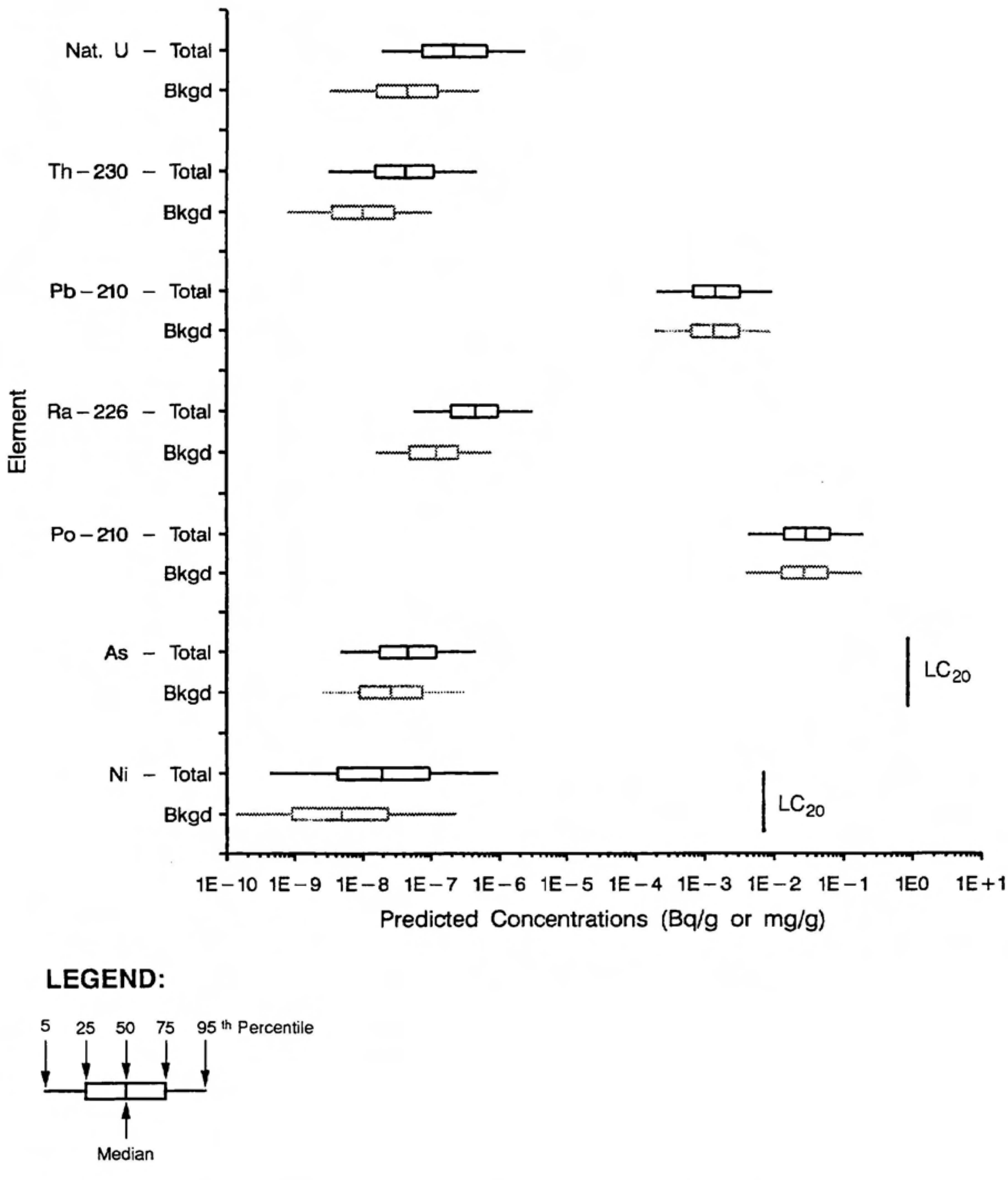
Radon released from the site will initially have very little Pb-210 associated with it, but will increase as a function of time as the radon decays. However, atmospheric dispersion reduces the concentration of radon, and hence Pb-210, as it is transported away from the site. The effects of these competing mechanisms were quantified by calculating the Pb-210 concentration from the modelled radon results. Weather data from the Key Lake site and from the Environment Canada station at Cree Lake show that the average wind speed for the area is about 3 m/s (i.e. 11 km/h) (Cameco 1995b, 1996b). Based on this wind speed, the Pb-210 concentrations were calculated as a function of distance using RadDecay 3.03 (Grove Engineering Inc. 1990); see Figure 3.4. The plot clearly shows that Pb-210 from the Key Lake mine site remains a small fraction (about 3 per cent) of the natural background concentration.

Figure 3.4 Pb-210 Concentration vs Distance from Key Lake Facility



This result shows that the contribution of Pb-210 in lichens from mine-site radon emissions will be insignificant with respect to natural background and not detectable. Monitoring of lichens close to the uranium mines has confirmed this prediction. The atmospheric emissions of radon from uranium mines do not cause a significant dose to caribou (see Figure 3.5) or humans eating the caribou (SENES and TAEM 1995c).

Figure 3.5 Predicted Concentrations in Wolf (Winter)



NOTE:

1. Bkgd. = Radionuclide / metal concentration in wolf from consumption of caribou containing natural background levels in winter.
2. Total = Radionuclide / metal concentration in wolf from consumption of caribou containing natural background levels plus facility emissions in winter.

4.0 THE ELLIOT LAKE REGION; STANLEIGH MINE

The discovery of uranium near Elliot lake in 1953, in conjunction with the then high demand for uranium, resulted in the development of 12 mines and 11 mills by seven owners between 1955 and 1959. The demand for uranium then declined and by the mid-1960's only Denison Mines Limited and Rio Algom Limited's Nordic Mine remained in operation. In the mid-1970's, major shifts in the global energy markets stimulated the demand for uranium as a source of energy. However, by 1990, the world price of uranium declined and Rio Algom and Denison mines were unable to operate economically. Rio Algom's Panel and Quirke mines ceased operation in 1990, Denison Mine Limited closed the Denison mine in 1992 and the Stanleigh mine ceased operations 30 June 1996.

The decommissioning plan for the Stanleigh waste management area (WMA) is based on a water-cover concept with raised dams. A detailed pathways analysis was prepared for the assessment of potential impacts from the flooded tailings (SENES 1996). The pathways model was calibrated using site-specific data and used to estimate water and sediment concentrations downstream of the WMA (see Figure 4.1) (SENES 1996). Sample water quality and sediment quality results are shown in Figures 4.2 and 4.3, respectively. It should be noted that the water quality is gradually improving in the Serpent River watershed and is gradually approaching background levels.

A screening level ecological risk assessment was carried out as part of the decommissioning study (SENES 1996). A representative list of Valued Ecosystem Components (VECs) was selected (Tables 4.1 and 4.2) for this assessment based on several factors. The list represents species that may be relatively sensitive to the effects of ionizing radiation (e.g. mammals and birds), are present in the watershed, have a relatively high potential for exposure (e.g. occur near the mine site), are of importance in the human food chain (e.g. fish, duck and moose) and represent various trophic levels (e.g. plants, invertebrates, fishes, small prey species, fur bearers, omnivores and predators). Specific biota included in the study are for example mallard, grouse, deer, moose, lake trout, whitefish as well as hare, bald eagle, black bear, wolf, merganser, scaup, muskrat, aquatic plants and invertebrates, and benthic invertebrates. The range of contaminant impacts on these species is reasonably expected to encompass the impacts felt by other species that are not specifically considered in the model.

The ecological risk assessment considered the effects of exposure to both radioactive and non-radioactive contaminants present in the WMA discharge. The effect of contaminants on the VECs has been estimated taking into account ingestion of contaminated food and water, exposure to gamma radiation from waterborne radiation (e.g. while immersed in water) and sediment-associated radiation. A summary of predicted mean dose rates to aquatic and terrestrial biota is provided in Table 4.3.

Based on Table 4.3, the dose rate estimates for the biota considered range from about 0.1 to 21.3 μGy/h, with the exception of those for wolf and eagle which were much lower. Detailed results (SENES 1996) indicated that essentially all of these estimates relate to biota exposure to sediments (the contribution from water was negligible). It is then not surprising that the highest estimates correspond to the biota that are most closely associated with sediments, e.g. snails Table 4.4.

The result of the screening level assessment of radiation doses to selected aquatic and terrestrial biota indicate that there are predicted to be no significant ecological consequences (in terms of the biota health) associated with the aquatic impacts from the decommissioned Stanleigh WMA. Furthermore, given the lower dose rate estimate for fish in McCabe Lake than for fish in WMA, ecological consequence (in terms of the biota health) associated with potential aquatic impacts from the decommissioned Stanleigh WMA on its immediate receiving downstream water body (McCabe lake) are also predicted to be not significant.

Figure 4.1 Serpent River Basin

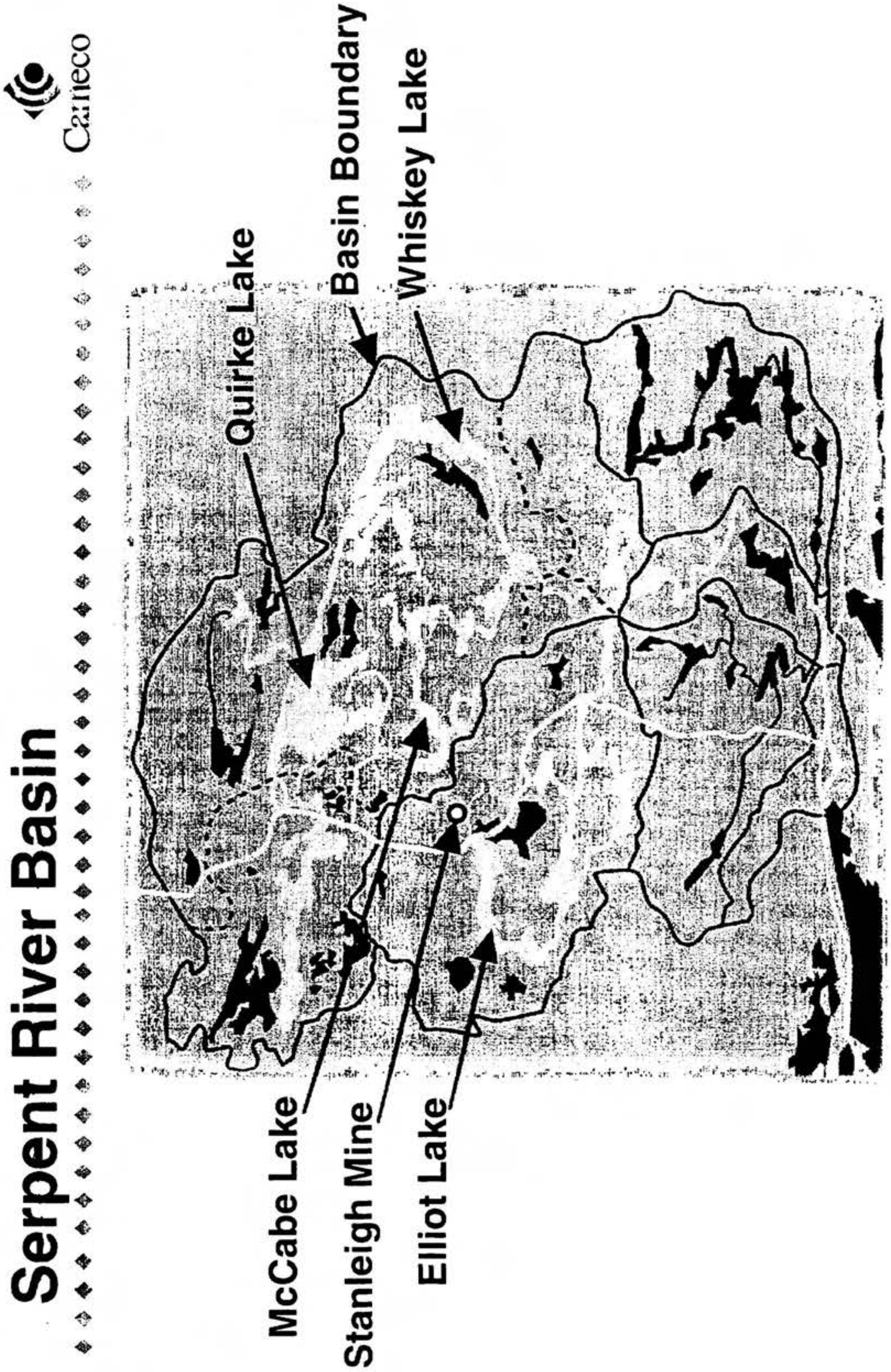


Figure 4.2 Predicted Ra-226 Concentration in McCabe Lake Water

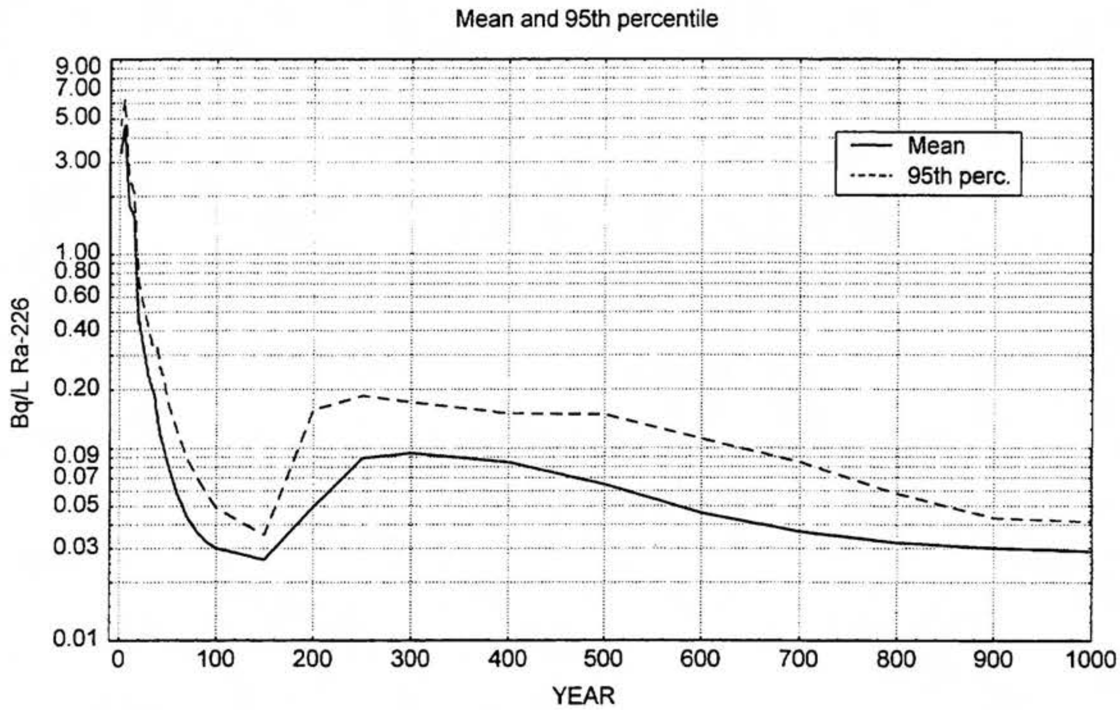
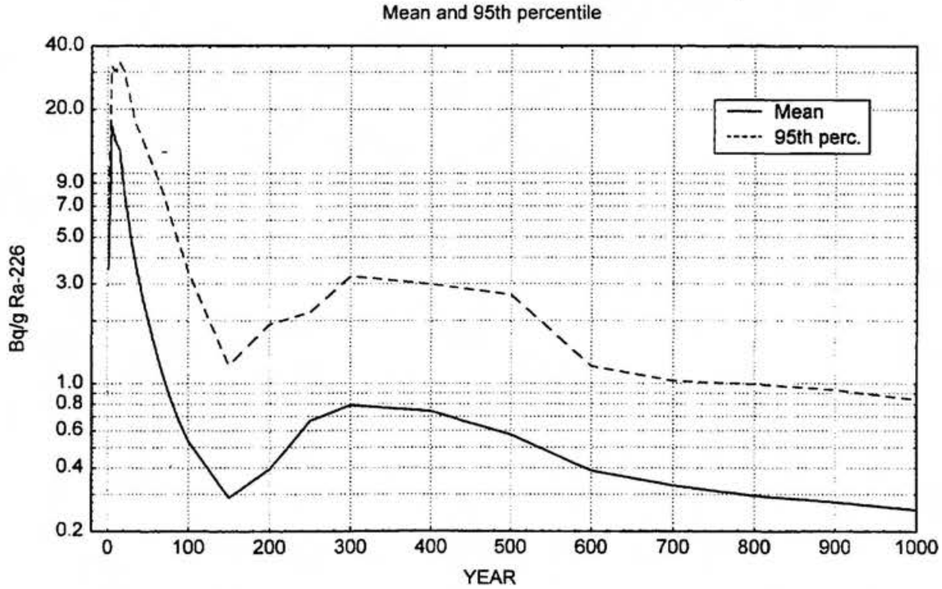


Figure 4.3 Predicted Ra-226 Concentration in McCabe Lake Sediments



Screening indices were also estimated for aquatic and terrestrial biota due to exposure to heavy metals, including uranium. Both mean values and 95th percentile were calculated (SENES 1996). The results for the effect of uranium on aquatic biota are illustrated in Table 4.5. The aquatic VECs included in this analysis are: water milfoil, primary producers, benthic invertebrates, zooplankton, northern pike, lake whitefish and white sucker.

Table 4.1 Location of Valued Terrestrial Ecosystem Components Considered for Pathways Modelling

Terrestrial Biota	Home Location	Water and/or Food Source	Residency Factor[1]	Occupancy Factor[2]	
				Water	Sediment
Bald Eagle	WMA	SRWN/WMA[4]	0.25	0	0
	McCabe	McCabe	0.25	0	0
Bear	WMA	SRWN/WMA[4]	033	0.05	0
	McCabe	McCabe	0.33	0.05	0
Beaver	WMA	WMA	1	0.25	0
	McCabe	McCabe	1	0.25	0
Deer[3]	Serpent River Watershed N	McCabe-May-Hough-Pecor	1	0	0
Grouse[3]	McCabe	McCabe	1	0	0
Hare	McCabe	McCabe	1	0	0
Mallard[3]	WMA	WMA	0.25	0.8	0
	McCabe	McCabe	0.25	0.8	0
Merganser	WMA	WMA	0.5	0.8	0
	McCabe	McCabe	0.5	0.8	0
Moose[3]	Serpent River Watershed N	McCabe-May-Hough-Pecor	1	0.1	0
Muskrat	WMA	WMA	1	0.5	0.5
	McCabe	McCabe	1	0.5	0.5
Scaup	WMA	WMA	0.05	0.8	0
	McCabe	McCabe	0.05	0.8	0
Wolf'	Serpent River Watershed N.	McCabe-May-Hough-Pecor	0.25	0	0

Notes:

1. Mean values. Residency factor <1 for animals with large home range relative to home-location areas and for migratory animals.
2. Occupancy factors represent fractions of local residency factor where animal is assumed immersed in water or sediment.
3. Animals consumed by critical groups.
4. Water source (Serpent River Watershed North)/Fish source (Waste Management Area).

Table 4.2 Location of Valued Aquatic Ecosystem Components Considered for Pathways Modelling

Aquatic Biota	Home Location	Water and/or Food Source	Residency Factor	Occupancy Factor	
				Water	Sediment
Benthic Invertebrates	McCabe	McCabe	1	0	1
Lake trout	WMA	WMA	1	1	0
	McCabe	McCabe	1	1	0
Pond Weed & Primary Producers	McCabe	McCabe	1	1	0
Snail	WMA	WMA	1	1	0.5
	McCabe	McCabe	1	1	0.5
Whitefish	WMA	WMA	1	1	0.4
	McCabe	McCabe	1	1	0.4

Table 4.3 Summary of Predicted Mean Dose Rates to Aquatic and Terrestrial Biota

Category of Biota	Mean Dose Rate (µGy/h)				
	Internal	% of Total	External	% of Total	Total
WMA - Fish[1]	1.94×10^{-2}	0.2	1.06×10^{1}	99.8	1.06×10^{1}
WMA - Snail[1]	7.20×10^{-2}	0.3	2.12×10^{1}	99.7	2.13×10^{1}
WMA - Moose[2]	6.63×10^{-4}	0.1	5.88×10^{-1}	99.9	5.88×10^{-1}
WMA - Bear[2]	1.86×10^{-5}	0.0	9.69×10^{-2}	100.0	9.70×10^{-2}
WMA - Wolf[2]	4.96×10^{-6}	100.0	0.00	0.0	negligible
WMA - Eagle[2]	1.04×10^{-4}	100.0	0.00	0.0	negligible
WMA - Beaver[1]	6.55×10^{-6}	0.0	1.33×10^{1}	100.0	1.33×10^{1}
WMA - Muskrat[1]	2.26×10^{-5}	0.0	1.59×10^{1}	100.0	1.59×10^{1}
WMA - Mallard[1]	1.02×10^{-3}	0.0	2.12	100.0	2.12
WMA - Scaup[1]	1.35×10^{-4}	0.1	1.06×10^{-1}	99.9	1.06×10^{-1}
WMA -Merganser[1]	2.93×10^{-6}	0.0	5.31×10^{-1}	100.0	5.31×10^{-1}
McCabe Lake - Fish[1]	2.09×10^{-2}	0.8	2.76	99.2	2.78

Notes:

1. For comparison, the above mean dose estimates for aquatic biota are considerably below (by about a factor 20 or more) the suggested guideline of 400 µGy/h of the International Atomic Energy Agency (IAEA 1992) and the United States Council on Radiation Protection and Measurements (U.S. NCRP 1991) for ensuring the prediction of aquatic biota. (400 µGy/h approx. = 10 mGy/d.)

2. For comparison, the above mean dose estimates for beaver and muskrat are about a factor 2.5 below the suggested guideline of 40 µGy/h of the IAEA (1992), a level below which there is no convincing evidence from the scientific literature which associates harm to animals and plants with chronic exposure. For other terrestrial biota (moose, bear, wolf and eagle), the estimated dose rates are considerably below (by about a factor 70 or more) the suggested IAEA (1992) guidelines of 40 µGy/h. (40 µGy/h approx. = 1 mGy/d.)

Table 4.4 Relationship Between the Mean Dose Rate Estimates and the Sediments

Category of Biota	Mean Dose Rate (μGy/h)	Assumptions Include
WMA Snail	21	Completely buried in fresh sediments (with radioactivity levels equivalent to those of the tailings solids).
WMA Beaver and Muskrat	13-16	When in its habitat, assumed surrounded by dry shoreline sediments (with radioactivity levels equivalent to those of the tailings solids). When in water, assumed exposed to fresh shoreline sediments (with radioactivity levels equivalent to those of the tailings solids).
WMA Fish	11	Always exposed to fresh sediments (with radioactivity levels equivalent to those of the tailings solids).
McCabe Lake Fish	3	Always exposed to fresh sediments from McCabe Lake with lower radioactivity levels than those in tailings solids.
WMA Ducks	0.1-2	When in water, exposed to fresh sediments (with radioactivity levels equivalent to those of the tailings solids).
WMA Moose and Bear	0.1-0.6	When in water, exposed to fresh sediments (with radioactivity levels equivalent to those of the tailings solids).
WMA Wolf and Eagle	<0.0001	No exposure to sediments.

The detailed results (SENES 1996) show that the screening indices of all the metals, except for copper, are much smaller than one, indicating that the release of these metals from the Stanleigh WMA should not result in any measurable ecological effect. The only screening index that exceeds one is the copper screening index for primary producers such as chlorophyta (SENES 1996). However, the copper concentrations in water at McCabe Lake are not significantly higher than background, and the calculated screening indices indicate that even background copper concentrations in the area will result in a screening index that exceeds one.

Table 4.6 summarizes the screening index calculations for aquatic invertebrates based on sediment concentrations in McCabe Lake (in Year 10). These calculations show that the screening indices for copper (95th percentile) and nickel (95th percentile) are somewhat larger than one. Background sediment concentrations are also presented in Table 4.6. The background concentrations for copper are similar to the McCabe Lake sediment concentrations and result in similar screening index values. Nickel concentrations in McCabe Lake sediment could potentially affect aquatic invertebrates. It should be noted that screening indices are not direct estimates of ecological risk, rather they indicate priorities for further investigation.

Table 4.5 Screening Index Calculations For Uranium In McCabe Lake

Valued Ecosystem Component	Test Species	Toxicity Levels (mg/L)		Concentration (mg/L)		Screening Index Values	
		LC/ EC50	LC/ EC20	Mean	95th Perc.	Mean	95th Perc.
Uranium:							
Water Milfoil	Myriophyllum	n/a	n/a	0.0029	0.0043		
Primary Producers	Chlorophyta	n/a	n.a	0.0029	0.0043		
Benthic Invertebrates	Chironomus sp.	n/a	n.a	0.0029	0.0043		
Zooplankton	Daphnia sp.	n/a	n/a	0.0029	0.0043		
Northern Pike	Rainbow Trout	8	3.2	0.0029	0.0043	0.001	0.001
Lake Whitefish	Bluegill Sunfish	16.7	6.68	0.0029	0.0043	<0.001	0.001
White Sucker	Bluefill Sunfish	16.7	6.68	0.0029	0.0043	<0.001	0.001

Literature data and recent field observations in the area indicate that all streams and lakes affected by the Stanleigh WMA discharge support viable benthic macroinvertebrate communities (Beak 1996). However, the benthic community in McCabe Lake is different than communities in a reference lake (Dunlop Lake). This difference may be partially associated with concentrations of nickel, copper (based on the screening model results) and perhaps iron and aluminum (empirical correlations; Beak 1996) in the sediment. These results are useful to the design of future monitoring programs.

Table 4.6 Screening Index Calculations for Aquatic Invertebrates Based on Year 10 (I.E. 2005) Sediment Concentrations

Contaminant	Sediment Quality Criteria (:g/g)*	Sediment Concentration (:g/g)				Screening Index Value			
		Background		McCabe Lake		Background		McCabe Lake	
		Mean	95th Perc.	Mean	95th Perc.	Mean	95th Perc.	Mean	95th Perc.
Cadmium	10	1.2	3.4	1.4	3.9	0.12	0.34	0.14	0.39
Copper	110	47.1	159	47.9	150	0.43	1.45	0.44	1.36
Lead	250	42.7	149	56.6	145	0.17	0.60	0.23	0.58
Nickel	75	17.6	59	60.1	164	0.23	0.79	0.80	2.19
Zinc	820	119	322	234	722	0.15	0.39	0.29	0.88

* Ontario Severe Effect Level (MOEE 1993).

5.0 THE USE OF ECOLOGICAL RISK ASSESSMENT AS A DECISION TOOL WITH RESPECT TO THE MONITORING OF URANIUM MINING AND MILLING OPERATIONS

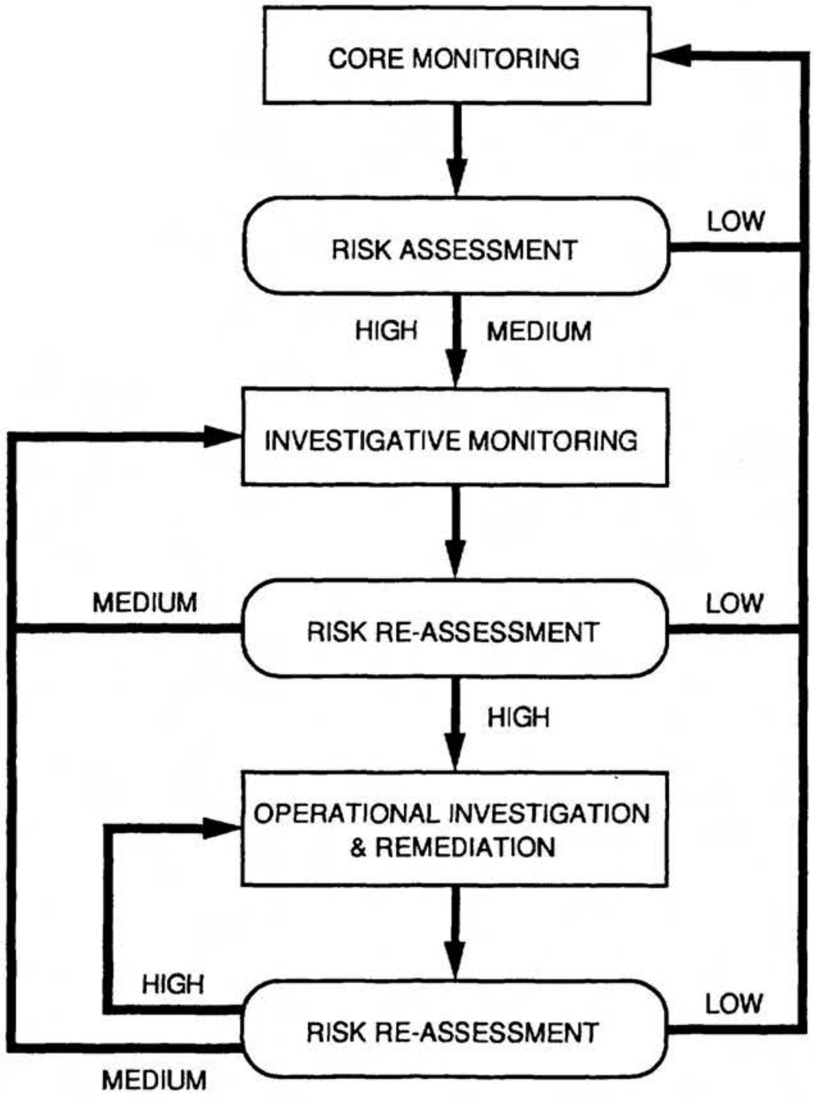
The concept of an integrated risk-based environmental management program is being investigated by mining companies in northern Saskatchewan. Such a program, augments baseline investigations and specifies extensive monitoring requirements of the aquatic, atmospheric and terrestrial environments. The monitoring program concept being investigated entails routine visual inspection of all facilities (e.g. pipelines), frequent monitoring of physical attributes of environmental components (e.g. lake levels) and routine collection of samples for analyses of various selected parameters (e.g. water and sediment sampling for chemical and radionuclide analyses). The data will provide the mining companies with routine and timely information for evaluating the performance of plant facilities (e.g. water treatment plants) and assessing impacts on environmental components (e.g. benthic invertebrates or fish). Likewise, the routine reporting requirements ensure that the regulators and the public will be kept apprised of environmental impacts and corrective action taken to minimize unforseen impacts.

The concept behind the integrated risk-based management approach to environmental monitoring links the response of valued ecosystem components to contaminant concentrations in the environment (i.e. is based on dose-response relationships). The risk-based approach is seen to offer several significant advantages over the traditional approach to interpretation of environmental monitoring data. In particular, it:

- incorporates dose-response information in the assessment of environmental data;
- allows direct assessment of impacts on valued ecosystem components;
- incorporates statistical power in the data assessment and comparison to natural variability;
- provides benchmarks on which to base decisions on follow-up investigative work;
- identifies potential concerns so that proactive steps can be taken at an early date;
- provides feedback for modifying or intensifying components of the environmental monitoring program; and
- provides direction to research and development activities.

A simplified diagram of risk-based environmental monitoring plan is shown in Figure 5.1.

Figure 5.1 Simplified Flow Diagram of Risk-based Environmental Monitoring Response Plan



6.0 CONCLUSIONS

The main conclusions from the studies surveyed in this paper are that:

(i) Unless the biota reside inside the tailings waste management area, there is little incremental ecological impact (observed or calculated). Furthermore, population-level models for biota in the vicinity of uranium mining and milling show no significant population impact due to radioactivity.

(ii) The benefits of environmental monitoring can be enhanced by implementing an integrated risk-based management strategy for environmental monitoring and by providing a feedback loop from the modelling results to the monitoring program.

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8.0 DISCUSSION

Question No.1: ***Did you perform any statistical power analysis? For example, in comparing Langley Bay to controls, did you in fact have an 80% (i.e., reasonable) chance of detecting a radiological impact, if one had occurred? My point is that the fact you report no impact is no proof you didn't have one.***

Mr. Takala was not able to provide a figure on what the power of the study was. The statistics he had quoted were at the 95% level of significance, but he acknowledged that this did not answer the question, and said he would have to go back to the original studies to be able to provide an answer later.

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