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ABSTRACT

Snowshoe hares (Lepus americanus) trapped near uranium tailings showed elevated levels of ^{226}Ra in their bones (250 $\pm$ 94 $mBq.g^{-1}$ dry weight), compared to 20-30 $mBq.g^{-1}$ d.w. at local control sites (3-15 km from tailings sites) and less than 3.7 $mBq.g^{-1}$ d.w. at a distant control site located 880 km away from uranium mining areas. Elevated levels of ^{210}Pb and ^{210}Po (95-245 $mBq.g^{-1}$ d.w.) were observed at all sites suggesting higher intake at non-tailings sites. No significant accumulation of U, Th and its isotopes were observed at any site. Higher levels of ^{228}Th compared to ^{232}Th are attributed to accumulation of ^{228}Ra similar to that of ^{226}Ra. Most stomach contents had levels of ^{226}Ra below detection limits (3.7 $mBq.g^{-1}$ d.w). Based on the levels of ^{226}Ra and ^{210}Po in bones, the maximum internal dose rate to the skeleton and the maximum lifetime dose of hares living near tailings were 3.9×10^{-5} $Gy.d^{-1}$ and 4.2×10^{-2} Gy, respectively. These levels are believed to be too low to produce osteosarcoma in hares or contribute significantly to environmental transport of radionuclides from tailings.

Key words: Radiation biology; Contamination uptake; Uranium tailings; Snowshow hares.

INTRODUCTION

Uranium mill tailings contain low levels of uranium and thorium decay series radionuclides which, after uranium milling operations, are left undepleted with the host rock residue together with other milling reagents. Upon weathering, these tailings undergo physical and chemical transformations thus mobilizing various radionuclides and chemical constituents. Via different environmental pathways, the tailings contaminants migrate to the surrounding environment affecting its natural quality in terms of soil, water, flora, fauna and eventually, man.

Environmental regulations require that during all phases of operations and after inactivity, the tailings are to be properly managed and the effluent treated to meet regulatory discharge guidelines. Usually inactive tailings areas are reclaimed and the surface stabilized to control erosion using revegetation or other chemical stabilization techniques. In order to develop long term management strategy and eventual abandonment scheme of such tailings, it is necessary to understand the dynamics of radionuclides migration from tailings to the environment and its effects on the area habitat.

There are about 180 million tonnes of uranium tailings in Canada. A majority of them, approximately 150 million tonnes placed over an area of 600 ha, are located in the uranium mining districts of Elliot Lake, Ontario, and the rest elsewhere in Ontario and Saskatchewan. The tailings in the Elliot Lake area are of different ages, a majority of them active and about 25 million tonnes inactive for the past twenty years or so. Most of the inactive sites have been successfully reclaimed by revegetation and some by chemical applications of lime for surface stability and erosion control. These tailings are to be returned to nature after rehabilitation and eventually will

be areas of wild life management. This varied set of deposits permits assessments to be made on the efficacy of the various waste stabilization treatments and their biohydrogeochemical evolution in relation to radionuclides and other contaminants migration. We are currently engaged in examination of radionuclides in the biota of the Elliot Lake region.

Previously, we have shown that the meadow vole (Microtus pennsylvanicus), a small microtine rodent established on revegetated uranium tailings habitat, and feeding on vegetation containing elevated levels of radionuclides, accumulates ^{226}Ra in its skeleton (Cloutier et al. 1985, 1986a, 1986b). Faecal pellets of snowshoe hare (Lepus americanus) collected near different uranium mine tailings have been shown to contain mean ^{226}Ra contents in their core ranging from 69 $\pm$ 10 to 648 $\pm$ 63 $mBq.g^{-1}$ dry weight (Clulow et al. 1986). It is known that alkaline earth series radionuclides, such as ^{90}Sr and ^{226}Ra being analogous to Ca, are bone seekers and lodge principally in the skeleton (Eisenbud 1973; Kestens et al. 1980; Leggett et al. 1982; Schlenker et al. 1982; Muth and Glöbel 1983) where they replace Ca and show a marked age dependence during the rapid bone development period. Other radionuclides such as ^{238}U, ^{234}U, ^{228}Th, ^{232}Th and daughter products of ^{226}Ra and ^{224}Ra are also known to retain and accumulate in the skeleton tissue of animals and humans (Schlenker 1988; Singh et al. 1987; Singh et al. 1988; La Touche et al. 1987). We believe that snowshoe hares are another bio-indicator of environmental perturbations related to uranium mining, milling and processing activities. In this study we report skeletal concentrations of ^{230}Th, ^{226}Ra, ^{210}Pb, ^{210}Po, ^{232}Th, ^{228}Th and total U and Th in showshoe hares trapped near uranium mill tailings and on local and distant control sites and calculate the average internal skeletal radiation dose. This study also permitted us to investigate the state of isotopic secular equilibrium in biological samples.

METHODS

Study sites included three inactive uranium mine waste deposits near Elliot Lake, Ontario (46° 23' N, 82° 39' W): the Stanrock tailings, chemically stabilized by addition of lime slurry to the surface, and the Nordic and Lacnor tailings, which have been revegetated (Figure 1). Collection sites were downwind from the wastes and as close as possible to their boundaries. Prevailing winds in excess of 18.5 km/h are from the North and West as measured at the Elliot Lake Airport (Dave et al. 1985). Local control sites were located approximately: a) 3 km East of the Nordic tailings (C-1) (Figure 1); b) 15 km NW of the closest uranium tailings in Elliot Lake (C-2) (Figure 1); and c) a distant 880 km E of Elliot Lake at Disraeli, Quebec (45° 54' N, 71° 22' W), C-3.

Hares were snared during winter season (January to March) where tracks were observed on the snow surface. No attempts were made to snare animals in the absence of snow cover as runways were poorly defined due to the rocky terrain. Because of cold winter temperatures (-10 to -30°C) all samples were frozen when collected. Hares were taken to the laboratory, thawed, measured (Anderson 1965), skinned and dissected. Stomach contents were removed and oven dried at 70°C for one week. Bones of one hind leg (femur and tibiofibula) and both upper arms (humeri) were removed and cleaned by scraping. The proximal ends of both humeri were scoured in commercial bleach solution (Gross and Gross 1966) to reveal more clearly the presence or absence of epiphyseal cartilages used in age determinations (Hale 1949, Dodds 1965). Bones of the hind leg were oven dried at 70°C for one week, cut into 2-3 cm lengths, cooled in a desiccator for one hour and weighed. Additional drying time in the oven did not produce an appreciable weight loss. Moisture loss of the bones when dried to constant weight was 27.3 $\pm$ 0.63% (n=9). Each dried

sample was split into three equal representative parts. One sub-sample was used for ^{226}Ra analysis at the Elliot Lake Laboratory, and the other two were shipped to commercial radioanalytical laboratories, Monenco Analytical Laboratory, Calgary, Alberta, for ^{230}Th, ^{232}Th, ^{228}Th, ^{210}Pb and ^{210}Po analyses, and to Atomic Energy of Canada Ltd., Ottawa, Ontario, for total U and Th determinations, respectively.

For ^{226}Ra analysis, the bone samples were first spiked with a measured amount of ^{133}Ba tracer solution, oven dried at 100°C, and then slowly ashed in a muffle furnace at a maximum temperature of 550°C, for at least 8 hours. Ashes were then crushed, sufficient of the oxidant NH_4NO_3 (30%) added to wet them, and oven dried at 70°C overnight. Next day, samples were re-ashed at 550°C for over 12 h, then dissolved in 10% HCl and diluted to 1 L. An aliquot of the final solution was analyzed for ^{226}Ra content. Tests of the digestion and radio-analytical technique, using pairs of beef shank bone samples, with or without known amounts of ^{226}Ra and ^{133}Ba added, indicated a ^{226}Ra recovery rate of 98 $\pm$ 10%. Stomach contents were digested following the procedure described for faecal pellets in Clulow et al. (1986).

Ra-226 concentrations were measured, following precipitation as Ra-Ba sulphate from solution samples, by α-spectroscopy in which the 4.78 MeV α-decay peak of ^{226}Ra was counted. Analytic and measuring methods are described in detail elsewhere (Lim and Dave 1981). The ^{133}Ba tracer solution was added to the starting solid sample to measure the overall recovery for ^{226}Ra analysis. For ^{228}Th, ^{230}Th, ^{232}Th, ^{210}Pb, and ^{210}Po isotopes, the bone samples were wet ashed using a 1:1 mixture of 16 N HNO_3 and 70% $HClO_4$. Tracer solutions of ^{234}Th and ^{207}Bi, respectively were used for Th and Pb recovery determinations. Th isotopes were measured by α-spectrometry, and ^{210}Pb was measured indirectly by measuring the β-activity of its daughter ^{210}Bi after an ingrowth period of at least 30 days to achieve an equilibrium of greater than

98%. Details of these methods are provided in Chiu and Dean (1986).

Total U and Th were determined by the neutron activation, delayed neutron and γ-counting techniques using a SLOW POKE reactor at the commercial radioanalytical division of Atomic Energy of Canada Ltd., Ottawa, Canada.

Statistical analyses of data were carried out using the SPSS software package (Nie et al. 1985). Differences among bone concentrations at various sites were tested for significance at the 5% level by one way analysis of variance (ANOVA), where significance was indicated in the ANOVA, a Student-Newman-Kauls range test was used to separate sample means different at the 5% level.

RESULTS AND DISCUSSION

Because of the limited sample size for a given location, the hares were not classified according to sex. Accurate age determination was also difficult. As snowshoe hares are born only up to the end of August (Meslow and Keith 1968) and all our animals were trapped in January or later, they were clearly at least four months old. To distinguish juvenile from adult hares, several methods were considered. Although Keith et al. (1968) identified eye lens weight as the surest indicator of juvenile males after the month of December, and juvenile females after mid-March, this method was unuseable because of the breakdown and loss of lens tissue due to the freezing of the sample which occurred (unavoidably) in the field (Friend 1967). Hale's (1949) method for aging cottontail rabbits, based on regression of the epiphyseal cartilage of the humerus, has been adapted for showshoe hare aging by Dodds (1965). All our samples showed a complete absence of the cartilage. Using Dodds' criteria this would indicate an age of 235 days (approximately 8 months) or more. Since Dodds cautions that ossification may vary

individually, and among populations in different areas, and that this may cause error in aging wild animals, it is possible that some of our samples may have been younger than eight months. We conclude that bone development was complete in our specimens and that they were certainly at least four months, and probably at least eight months old.

Radionuclide levels of ^{230}Th, ^{226}Ra, ^{210}Pb, ^{232}Th, ^{228}Th and total U and Th concentrations in bones of animals from various sites are shown in Table 1. Though detectable levels of ^{230}Th, ^{226}Ra, ^{210}Pb, ^{210}Po, ^{232}Th and ^{228}Th were observed at all sites (except ^{226}Ra at distant control C-3), only ^{226}Ra levels were significantly higher (about ten times) at Stanrock and Lacnor tailings sites ($P < 0.05$) than those of local control sites. As only two specimens were collected from the area of the Nordic tailings site, their data were not included in the statistical analyses, however, these two animals had levels approaching those of the local controls. These values are to be compared to the average ^{226}Ra concentration in dried vegetation (211 ± 22 $mBq.g^{-1}$) and tailings substrate ($11,900 \pm 370$ $mBq.g^{-1}$) (Cloutier et al. 1986b). In meadow voles, established on tailings, the bone values were respectively, 4079 ± 444, 4699 ± 851, 1036 ± 296 and 592 ± 37 $mBq.g^{-1}$ dry weight for winter, spring, summer and autumn.

Lower levels of ^{226}Ra in faecal samples from the Nordic compared to Stanrock and Lacnor tailings, reported previously (Clulow et al. 1986), and bone levels, noted here, may reflect lower levels in the diet of the Nordic animals compared to those of the other tailings sites. At both Stanrock and Lacnor sites, perhaps due to the abundant forest vegetation which attracts the animals and provides them with protective cover, hares approach and penetrate tailings areas. Signs of hare feeding activity were visible on the snow surface of the Lacnor tailings in winter. On the other hand, the Nordic site lacks forest cover and is surrounded by open country for 200 m; it seems

reasonable to suppose that animals from this site browse at some distance from the waste deposit and that vegetation in their diet contains less ^{226}Ra than at the other two sites.

Based on the ^{226}Ra levels in hare faeces reported earlier (Clulow et al. 1986) we concluded that airborne contaminants at Control C-1 were too low to enter into the diet of the animals in appreciable amounts, although the control site is within the downwind range (6.6 km) where contamination by airborne particulates has been recorded in the U.S.A. (Skinner 1982). Data from the present study support our previous conclusions: ^{226}Ra levels in bones of hares collected at Control C-1, located 3 km downwind of the Nordic tailings were similar to those of Control C-2, located 15 km upwind from the closest uranium tailings.

Ra-226 levels in skeletons of hares collected in an area far removed from Elliot Lake (Control C-3) were all below detection limits (<3.7 mBq/g^{-1} dry weight), and therefore at least five times lower than background levels in the Elliot Lake area (Control C-1 and C-2) (Table 1). Statistical comparison of data from all three control sites was not possible because the Control C-3 levels were not quantified. The higher levels of the Elliot Lake control samples probably reflect a generally higher background level of the radionuclide related to the rich uranium deposits in the region (Dave et al. 1985).

The distribution of ^{226}Ra levels in stomach contents at the time of capture is shown in Table 2. These values were quite different from the faecal levels reported previously (Clulow et al. 1986). With the exception of one sample from Stanrock and one from Lacnor, all samples analyzed contained ^{226}Ra levels below the detection limit of 3.7 $mBq.g^{-1}$ dry weight and considerably below levels in faeces. Although considered to have a sedentary nature, Lepus americanus has a home range area which may extend to several

hectares (Boutin 1979) and animals are known to make temporary but long-range movements from these centres of activity on occasion (Boutin et al. 1985). These observations carry the implication that feeding and defaecation of the hares may occur in widely spaced locations. Consequently, faeces and stomach contents of animals collected on or near tailings do not necessarily contain forage taken in the vicinity of the high ^{226}Ra levels. Further, the levels of faeces may reflect higher body excretion rates affected by dietary variations in $^{226}Ra/Ca$ ratio and total body burdens. As the measured levels are so heterogeneous, it is not possible to calculate a transfer coefficient from food to animal, based on the stomach content data we obtained.

The results also show that for all sites (including tailings and controls), no total uranium and thorium (chemical form, predominant isotopes ^{238}U and ^{232}Th) were detected. Th-230 and ^{228}Th were measured in the ranges of <10-14 and 20-65 $mBq.g^{-1}$, respectively. These values were not significantly different ($P<0.05$) for tailings and control sites, suggesting much lower intake and retention of U and Th. The higher measured values of ^{228}Th compared to its series parent ^{232}Th is primarily because of higher intake and retention of ^{228}Ra. which decays to ^{228}Th and accounts for most of the ^{228}Th and its uniform distribution in the skeleton (Singh 1988). They also observed that in dogs and humans, the skeleton contained the majority of Th isotopes (80-90%), the rest being distributed in soft tissues: lungs, lymph nodes, liver, kidney, spleen, gonads and thyroid. Higher values of ^{226}Ra compared to its parent ^{230}Th in the skeleton tissues of hares (this study) further support the preferential accumulation of radium than thorium in bones.

Results for ^{210}Pb and ^{210}Po are most interesting due to the fact that for the two tailings sites: Stanrock and Lacnor, there is apparent equilibrium between ^{226}Ra and its daughter products ^{210}Pb and ^{210}Po (within

statistical uncertainty). For other sites the ratio of $^{210}Pb/^{226}Ra$ is much greater than 1.0, varying between 5-35. The apparent equilibrium between ^{210}Pb and ^{210}Po is misleading because although the data showed equilibrium at the time of measurement, it may not necessarily be true at the time of sampling because a considerable time elapsed beween sampling and analysis (approximately 180 days), compared to the half-life of ^{210}Po (140 days). Though the availability of ^{210}Pb is higher or equivalent to that of ^{226}Ra at the tailings sites, higher skeleton burdens of ^{210}Pb at Nordic and control sites (especially the distant control) suggest different biological sources of intake and retention. Wren (1988) observed higher levels of chemical Pb, in mink and otters, close to industrial sites related to atmospheric loading of Pb from emissions and from antropogenic sources. Singh (1986) suggested additional intake sources of ^{210}Pb by inhalation and ingestion of Rn and its daughter products and their transfer from lung fluid to blood to bone. Present data support the radon pathway through both the groundwater and atmospheric routes. The results at the two tailings sites (Lacnor and Stanrock) may also suggest a biological limiting retention process irrespective of intake levels where higher ^{210}Pb levels are available. The processes for radium and lead are also independent.

The results also clearly showed a complete loss of radioisotopic secular equilibrium between the source (uranium tailings), geochemical pathway of radionuclide release (oxidation and leaching of tailings) and biological uptake through vegetation and herbivores. Lim (1989) also suggested disequilibrium in the biological pathway based on his geochemical measurements on tailings solid and porewater.

Hares established near uranium tailings are exposed to such external radiation sources as radon gas and its decay products, by inhalation, and internal ones such as ingested radionuclides, which may accumulate in the

body. Although calculation of the dose from external sources is beyond the scope of our study, our knowledge of ^{226}Ra and its daughter levels in the skeleton permits a calculation of the radiation dose from Ra decay series α-particles, which have major biological effects. Beta-ray and γ-ray contributions have been omitted from the dose calculation (after Van Dilla et al. 1958; Harley and Pasternack 1976; Momeni et al. 1976).

According to Momeni et al. (1976), the time-integrated dose rate, i.e., the cumulative dose, based on the energy from α-particles, plus recoil energy produced by each ^{226}Ra disintegration as 4.86 MeV, and ^{222}Rn and its daughters as 20.36 MeV (not including ^{210}Po), are:

for ^{226}Ra:

$$D_{Ra} = 1.38 \times 10^{-8} \times 4.86 \int_{to}^{t} C(t)dt \qquad \text{Eq 1}$$

and for ^{222}Rn and its daughter products:

$$D_{Rn\ daugh.} = 1.38 \times 10^{-8} \times 20.36 \int_{to}^{t} C(t)f(t)dt \qquad \text{Eq 2}$$

and

$$D_{^{210}Po} = 1.38 \times 10^{-8} \times 5.406 \int_{to}^{t} C(t)dt \qquad \text{Eq 3}$$

Where $C(t)$ and $C_{^{210}Po}(t)$ are the average skeletal radium and ^{210}Po concentrations in fresh bone ($mBq.g^{-1}$) at time t (days) and f(t) is the radon-to-radium ratio in the skeleton. The unit of dose is the Gray (Gy).

The radon:radium fraction f(t) in the skeleton is assumed to be the same as the total body radon retention. Harley and Pasternack (1976) suggested α-dose computations from ^{226}Ra based on a radon:radium ratio of 1:3. In a previous paper (Cloutier et al. 1986a) the total dose from ^{226}Ra and its daughter products to the bone was thus obtained by combining Equations 1, 2 and 3, using a radon:radium ratio of 1:3, and by expressing C_f in terms of measured ^{226}Ra based on dry weight C_d ($mBq.g^{-1}$ dry weight), and moisture content fraction 'm' to give:

$$D_{226_{Ra}} = 1.6 \times 10^{-7} \times C_d \times (1-m) \times t_{max} \quad \text{Eq 4}$$

and $$D_{210_{Po}} = 7.46 \times 10^{-8} \times C_{d\ 210_{Po}} \times (1-m) \times t_{max} \quad \text{Eq 5}$$

where, t_{max} is the age of the animal on the day of death and $C_{d\ Po\text{-}210}$ is the measured ^{210}Po concentration, $mBq.g^{-1}$ dry weight.

The moisture content 'm' was found to be 27.3% (see Methods), and the average dose rate (Gy/day) is:

$$\frac{dD}{dt}\ 226_{Ra} = 1.60 \times 10^{-7} \times C_d \times (1-m) \quad \text{Eq 6}$$

$$\frac{dD}{dt}\ 210_{Po} = 7.46 \times 10^{-8} \times C_{d\ 210_{Po}} \times (1-m) \quad \text{Eq 7}$$

Equations 4 to 7 are based on the assumption that the skeletal levels of ^{226}Ra and its daughter products were constant throughout the life of the hares. This assumption leads, most certainly, to an overestimation of the dose calculation as the level of ^{226}Ra was likely to be less before ossification was complete, and ^{210}Po may not be in equilibrium with ^{210}Pb at the time of sampling.

Using Equations 4 and 7, the measured skeletal ^{226}Ra and ^{210}Po concentrations and moisture content, the average dose rate dD/dt for hares near uranium tailings and control sites have been calculated and are shown in Table 3. Because the bones of the specimens all showed signs of reaching maximum development, and because the Ra:Ca ratio in the diet is taken to remain constant throughout life, it is assumed that the levels of ^{226}Ra, ^{210}Pb and ^{210}Po determined in the specimens would have been constant in animals for most of their lifetime. In a population of hares on Manitoulin Island, 40 km or so South of the experimental sites, only 3 to 17% of the animals were three or more years old in a 1959 study (Newson and De Vos 1964). Considering the published annual adult survival rates (see Meslow and Keith 1968), which range from 0.03 to 0.64 but lie mostly between 0.2 and 0.3, it seems doubtful that

many hares live beyond three years in the wild. Consequently, the lifetime cumulative dose was based on a three year span and was calculated using Equation 4 and t_{max} = 1095 days.

These calculated doses for snowshoe hares are below the threshold dose rates and lifetime cumulative doses of 3.9 x 10^{-5} Gy.d^{-1} and 0.8 Gy (man), 1.1 x 10^{-4} Gy.d^{-1} and 0.5 Gy (dog), and 1.6 x 10^{-3} Gy.d^{-1} and 1.1 Gy (mouse), required to produce osteosarcoma (Raabe et al. 1983). It seems unlikely therefore that the observed ^{226}Ra and its daughter products will adversely affect the hares of the area. As skeletal tissue is less easily assimilated by predators and scavengers the spread of these radionuclides through trophic levels is probably quite slow.

As ^{226}Ra and other isotopes may not distribute evenly in the skeleton, some areas (hot spots) may in fact have a higher concentration than measured and therefore be subject to a higher radiation dose (Momeni et al. 1976).

CONCLUSIONS

Snowshoe hares living in the immediate vicinity of revegetated uranium tailings have elevated amounts of ^{226}Ra, ^{210}Pb and ^{210}Po in their skeletons, this finding reflects the situation in meadow voles (Microtus pennsylvanicus) established on the same tailings (Cloutier et al. 1985). Observed levels of ^{210}Po and ^{210}Pb were also higher and comparable at local and distant control sites. No significant accumulation of U and Th isotopes in bones was measured.

The calculated lifetime cumulative doses of 2.6 x 10^{-3} to 3.5 x 10^{-2} Gy (based on the average ^{226}Ra levels in the hind leg), and 8.2 x 10^{-3} to 4.2 x 10^{-2} Gy (including ^{210}Po levels) are believed too low to produce osteosarcoma in animals of the area. The measured levels of ^{226}Ra, ^{210}Po and ^{210}Pb in the skeletal tissues (20 to 275 mBq.g^{-1} d.w.) are too small, compared to levels in

the tailings, to contribute significantly to the environmental transport of radioactivity from the tailings to secondary consumers, including man.

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LIST OF ILLUSTRATIONS

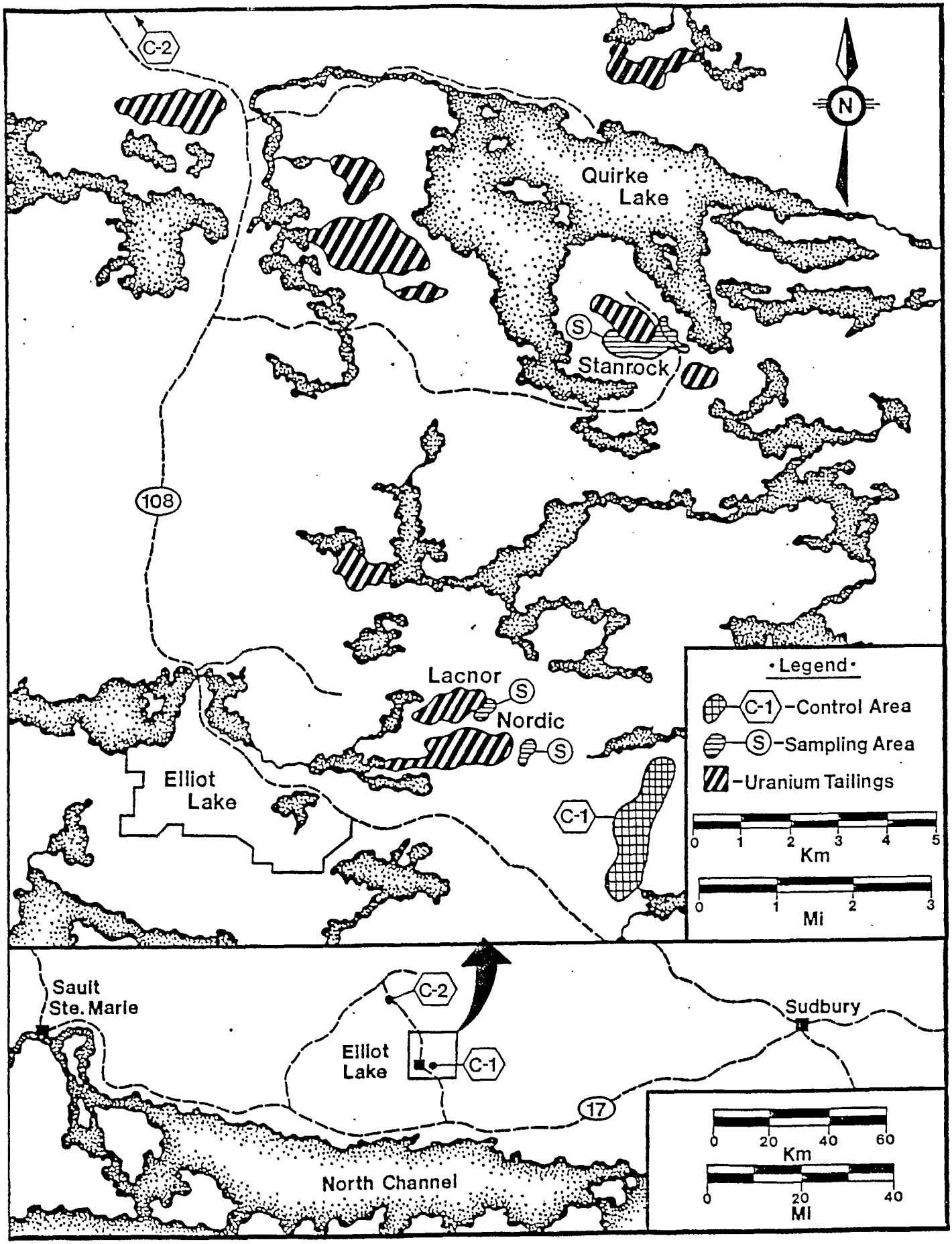
C-2
N
Quirke
Lake
S
Stanrock
108
Lacnor
S
Nordic
S
Elliot
Lake
C-1
·Legend·
C-1 -Control Area
S -Sampling Area
-Uranium Tailings
0 1 2 3 4 5
Km
0 1 2 3
Mi
Sault
Ste. Marie
C-2
Elliot
Lake
C-1
Sudbury
17
North Channel
0 20 40 60
Km
0 20 40
Mi

Figure 1

Table 1. Radionuclides (^{230}Th, ^{226}Ra, ^{210}Pb, ^{210}Po, ^{232}Th, ^{228}Th) and total U and Th contents of hind leg bones* of snowshoe hares, Lepus americanus.

Site	n	Radionuclide Concentration, Cd, $mBq.g^{-1}$ Dry Weight						Total U mg/g Dry Wt.	Total Th mg/g Dry Wt.
		^{230}Th	^{226}Ra	^{210}Pb	^{210}Po	^{232}Th	^{228}Th		
Stanrock tailings (chemically stabilized)	6	<10	220±32[a*]	210±80[c]	245±82[d]	<10	45±35[e]	N.D	N.D.
Lacnor tailings (revegetated)	4	12.5±5	275±94[a]	176±179[c]	125±44[d]	<10	65±58[e]	"	"
Nordic tailings (revegetated)	2	<10	20 (ave.)	100 (ave.)	95 (ave.)	<10	25±10[e]	"	"
Control C-1 (3 km E of Nordic)	6	<10	30±10[b]	154±97[c]	105±87[d]	<10	20±11[e]	"	"
Control C-2 (15 km NW of U mines)	5	<10	26±7[b]	112±16[c]	126±74[d]	<10	28±8[e]	"	"
Control C-3 (880 km E of U mines)	5	14±9	<3.7**	128±63[c]	130±93[d]	<10	40±16[e]	"	"

* Mean ± 1 S.E. indicated values with different superscripts significantly differ at 5% level (ANOVA/Student-Newman-Keuls range test, Nordic and Control C-3 omitted, see text).

** below detection limit.

N.D not detected.

Table 2. Grouped frequency distribution of ^{226}Ra levels ($mBq.g^{-1}$ dry wt.) in stomach contents of snowshoe hares, Lepus americanus, trapped around uranium tailings and control sites.

Site	Class Limits ($mBq.g^{-1}$ dry wt.) <3.7	3.8-35.0	35.1-55.0
Stanrock tailings	5	0	1
Lacnor tailings	3	0	1
Nordic Tailings	2	0	0
Control C-1	6	0	0
Control C-2	5	0	0
Control C-3	5	0	0

Table 3. Dose rate and life time cumulative dose received by leg bones of snowshoe hares, Lepus americanus

Site	From ^{226}Ra & its progeny excluding ^{210}Po		From ^{210}Po		Total
	Dose Rate $\frac{dD}{dt}$ $Gy.d^{-1}$	Cumulative Lifetime Dose D Gy	Dose Rate $\frac{dD}{dt}$ $Gy.d^{-1}$	Cumulative Lifetime Dose D Gy	Cumulative Lifetime Dose D Gy
Stanrock tailings (chemically stabilized)	$2.6x10^{-5}$	$2.8x10^{-2}$	$1.3x10^{-5}$	$1.4x10^{-2}$	$4.2x10^{-2}$
Lacnor tailings (revegetated)	$3.2x10^{-5}$	$3.5x10^{-2}$	$6.8x10^{-6}$	$7.4x10^{-3}$	$4.2x10^{-2}$
Nordic tailings (revegetated)	$2.3x10^{-6}$	$2.6x10^{-3}$	$5.1x10^{-6}$	$5.6x10^{-3}$	$8.2x10^{-3}$
Control C-1 (3 km E of Nordic)	$3.5x10^{-6}$	$3.8x10^{-3}$	$5.7x10^{-6}$	$6.2x10^{-3}$	$1.0x10^{-2}$
Control C-2 (15 km NW of U mines)	$3.0x10^{-6}$	$3.3x10^{-3}$	$6.8x10^{-6}$	$7.4x10^{-3}$	$1.1x10^{-2}$
Control C-3 (880 km E of U mines)	$< 4.3x10^{-7}$	$< 4.7x10^{-4}$	$7.0x10^{-6}$	$7.7x10^{-3}$	$8.2x10^{-3}$