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28 June 2002

Mr. Richard R. Faucher
Président
Niocan Inc.
2000 rue Peel, Suite 560
Montréal, Québec H3A 2W5

Fax: (514) 843-4809

RE: Commitments to BAPE

Dear Mr. Faucher:

During our meeting with BAPE on Friday 20 June, 2002, BAPE requested information on a number of subjects from SENES, Roche and Niocan. Dr. Lowe and I have attempted to provide the information requested by BAPE in this letter. Our response follows in the order set out in your memorandum of June 25, 2002.

1) Transfer of Radioactivity Through the Food Chain

There is a considerable literature on this subject; however, data are periodically reviewed by international and national agencies and compiled in convenient summary form. A copy of data presented in Report No. 123I of the United States National Council on Radiation Protection (NCRP) is given in Appendix 1. Similar data can be found in reports of the International Atomic Energy Agency (IAEA) and the United Nations Scientific Committee on the Effect of Atomic Radiation (UNSCEAR), among others.

It should be noted that data on the levels of natural radioactivity in radon, soils, air and various foodstuffs were provided in Appendix E of my 25 April 2002 submission to the Administrative Tribunal.

2) Exposure Pathways Analysis

In response to a question from BAPE, we have completed a screening level exposure pathways analysis (Appendix 2) for an adult and a child who live on the carbonatite and inhale airborne radioactive dust, consume vegetation, meat and dairy products grown on the carbonatite and who

are exposed to gamma radiation from uranium-238 and thorium-232 series radionuclides in the soil. Other pathways of exposure such as cosmic radiation, inhalation of radon (and its decay products) and consumption of well water are not included in the screening analysis since:

- cosmic radiation is extra-terrestrial and is not in any way affected by Niocan;

- radon exposure is, for practical purposes, from exposure to radon (and its decay products) while indoors. Indoor radon levels are determined by factors which control the flux of radon from soil and rocks under a home into the home and, in our opinion, are not related to potential Niocan operations.

  There are considerable data on radon in houses in Oka which are available to BAPE, including for example, the DSP report "Le radon à Oka" and data from the Geological Survey of Canada (gamma.gsc.nrcan.gc.ca:80radon_html).

  In addition, I have attached copies of two (2) papers which might be of interest to BAPE "Indoor Exposure to ^{222}Rn: a public health perspective" [Ayotte et al., Health Physics 1998] and "Radon in Residences: Influences of Geological and Housing Characteristics" [Lévesque et al., Health Physics 1977].

- Uranium levels in process slurry at Niocan are well within the range of natural background. Thus, in our opinion, the Niocan project would have no impact on the level of uranium in groundwater.

As shown in Appendix 2, we have considered three (3) scenarios:

- Scenario 1: Base Case with (estimated) current radioactivity levels for average conditions over the carbonatite (i.e. uranium at 5.3 ppm and thorium at 23.8 ppm).

- Scenario 2: As for Scenario 1, but for people living on and consuming food from the (approximately) 130 ha of the carbonatite with natural uranium levels of 15 ppm, and natural thorium levels of 62.4 ppm.

- Scenario 3: With maximum predicted off-site air concentrations and deposition from Niocan added to the Base Case.

Details are given in Appendix 2. However, in brief, doses were estimated to both adults and children who live on the carbonatite and eat food grown on farms on the carbonatite. Radionuclides in both the uranium-238 and thorium-232 decay chain were considered. The exposure pathways evaluated included:

- Inhalation of airborne dust;

- Ingestion of:
  - Above-ground produce
  - Below-ground produce
  - meat;
  - dairy;
  - incidental soil.
- external gamma radiation.

Overall, the results are as follows:

TABLE 1
RESULTS OF EXPOSURE PATHWAYS ANALYSES

	Adult	Child
SCENARIO 1 (average uranium and thorium at 5.3 ppm uranium and 23.8 ppm thorium	505 µSv/y	809 µSv/y
Scenario 2 (130 ha of carbonatite at 15 ppm uranium and 62.4 ppm thorium)	1206 µSv/y	1852 µSv/y
Scenario 3 (average over carbonatite plus maximum incremental Niocan)	520 µSv/y	836 µSv/y

It should be emphasized that such estimates are approximations and that the numbers shown in the table are simply the result of mathematical operations and are not intended to represent significant figures. In reality, the doses to the adult and child receptors under Scenario 3 (i.e. with Niocan) and Scenario 1 (without Niocan) are effectively the same.

As I stated to BAPE (and previously to the Tribunal), any potential radiological impacts from Niocan operation are expected to be insignificant and not distinguishable from natural variation in background radioactivity.

3) Radioactivity in Uranium Tailings

Table 2 ~~provides~~ summarizes the uranium and radioactivity content of a variety of uranium mill tailings site throughout the world.

TABLE 2
URANIUM RADIOACTIVITY IN ORE AND TAILINGS

Mill Location	Approximate Ore Grade and Radioactivity (% U) [a]	(Bq U-238/g) [b]	Tailings U-238 Series Radioactivity [c] (Bq/g)
Key Lake, Sask.	13	1599	16310
Rabbit Lake, Sask.	1.9	234	2384
Cluff Lake, Sask.	0.51	63	640
Elliot Lake, Ont.	0.09	11	113
Akouta, Niger	0.43	53	539
Rossing, Namibia	0.029	3.6	36
Niocan tailings [d] at 10 ppm U	0.001	0.123	1.7

a. Uranium ore data from SENES 1998. *Long-Term Population Dose Due to Radon (Rn-222) Released from Uranium Mill Tailings*. Prepared for The Uranium Institute (now the World Nuclear Association), April.

b. Based on 1.23×10^4 Bq U-238 per g of natural uranium.

c. Assuming 95% recovery of U (four radionuclides down to U-234) and no revovery of ten additional radionuclides in U-238 series.

d. Radioactivity based of 14 radionuclides in equilibrium in U-238 series.

4) Dr. Chambers Testimony to the Tribunal

I believe that Niocan has already provided this to BAPE.

5) Uranium and Thorium

I believe that this question relates to the relative hazard from uranium and thorium series radionuclides.

The potential effects from inhaling or ingesting uranium-238 and thorium-232 series radionuclides are considered in the exposure pathways analysis (Appendix 2). The external gamma dose from uranium and thorium is also considered in the exposure pathways. The gamma radiation levels used in the exposure pathways are those measured by the GSC for Oka and reflect the total actual gamma radiation of the two decay series.

With respect to groundwater, thorium (relative to uranium) is immobile in neutral pH waters. This is the situation in Oka. As noted earlier, the uranium level in process slurries are low with respect to background levels.

Although not considered in the screening pathways analysis presented in Appendix 2, it should be noted that the risk per unit exposure from thoron and decay products) is about ¼ of that for radon (and its decay product). For example, ICRP 50 ("Lung Cancer Risk from Indoor Exposure to Radon Daughters", ICRP 1986) suggests relative risks (per working level month-WLM) of 0.019 and 0.0064 from radon-222 and radon-220, respectively.

Finally, we have provided a short discussion of "Radon Concentration, Exposure and Dose" in Appendix 3.

We hope these data are useful to BAPE. As always, we would be pleased to provide clarification on any of these issues or to assist in discussion of other topics that may arise.

We appreciate the opportunity to assist Niocan and BAPE.

Yours very truly,

SENES Consultants Limited

for Douglas B. Chambers, Ph.D.
Principal, Director of Risk and Radioactivity Studies

APPENDIX 1

TRANSFER COEFFICIENTS

NCRP REPORT No. 123 I

SCREENING MODELS FOR RELEASES OF RADIONUCLIDES TO ATMOSPHERE, SURFACE WATER, AND GROUND

Recommendations of the
NATIONAL COUNCIL ON RADIATION
PROTECTION AND MEASUREMENTS

Issued January 22, 1996

National Council on Radiation Protection and Measurements
7910 Woodmont Avenue / Bethesda, MD 20814-3095

TABLE 5.2—*Element specific transfer factors for terrestrial foods.*[a]

Element	B_w[b] (fresh vegetables)	B_{veg}[c] (dry forage)	F_m (milk) d L^{-1}	F_f (beef) d kg^{-1}
Ac	0.001	0.1	2E-6	2E-5
Ag	0.004	0.1	0.006	0.003
Al	0.004	0.1	0.0002	0.0005
Am	0.001	0.1	2E-6	5E-5
Ar	0	0	0	0
As	0.08	0.2	0.0001	0.02
At	0.2	0.9	0.01	0.01
Au	0.1	0.4	1E-5	0.005
B	0.01	0.1	0.003	0.0008
Ba	0.01	0.1	0.0005	0.0002
Be	0.004	0.1	2E-6	0.005
Bi	0.1	0.5	0.001	0.002
Bk	0.001	0.1	2E-6	2E-5
Br	0.4	2	0.02	0.05
C	—[d]	—[d]	—[d]	—[d]
Ca	0.5	5	0.003	0.002
Cd	0.5	1	0.002	0.001
Ce	0.002	0.1	3E-5	2E-5
Cf	0.001	0.1	2E-6	6E-5
Cl	20	100	0.02	0.04
Cm	0.001	0.1	2E-6	2E-5
Co	0.08	2	0.002	0.03
Cr	0.01	0.1	0.002	0.03
Cs	0.2	1.0	0.01	0.05
Cu	0.05	0.8	0.002	0.01
Dy	0.002	0.1	6E-5	0.002
Er	0.002	0.1	6E-5	0.002
Es	0.001	0.1	2E-6	2E-5
Eu	0.002	0.1	6E-5	0.002
F	0.02	0.1	0.007	0.02
Fe	0.001	0.1	0.0003	0.03
Fm	0.002	0.1	8E-6	0.0002
Fr	0.03	0.1	0.008	0.03
Ga	0.003	0.1	1E-5	0.0003
Gd	0.002	0.1	6E-5	0.002
Ge	0.4	4	0.01	0.2
H	—[d]	—[d]	—[d]	—[d]
Ha	0.002	0.1	5E-6	5E-6
He	0	0	0	0
Hf	0.003	0.1	2E-5	0.0004
Hg	0.3	1	0.0005	0.01
Ho	0.002	0.1	6E-5	0.002
I	0.02	0.1	0.01	0.04
In	0.003	0.1	0.0002	0.004
Ir	0.03	0.2	2E-6	0.002

TABLE 5..

Element
K
Kr
La
Li
Lr
Lu
Md
Mg
Mn
Mo
N
Na
Nb
Nd
Ne
Ni
No
Np
O
Os
P
Pa
Pb
Pd
Pm
Po
Pr
Pt
Pu
Ra
Rb
Re
Rf
Rh
Rn
Ru
S
Sb
Sc
Se
Si
Sm
Sn
Sr
Ta

TABLE 5.2—*Element specific transfer factors for terrestrial foods.*[a] *(Continued)*

Element	B_{vs}[b] (fresh vegetables)	B_{vss}[c] (dry forage)	F_m (milk) d L^{-1}	F_f (beef) d kg^{-1}
K	0.3	3	0.007	0.02
Kr	0	0	0	0
La	0.002	0.1	6E-5	0.002
Li	0.001	0.1	0.05	0.02
Lr	0.002	0.1	5E-6	0.0002
Lu	0.002	0.1	6E-5	0.002
Md	0.002	0.1	5E-6	0.0002
Mg	0.03	0.1	0.008	0.003
Mn	0.3	10	0.0003	0.001
Mo	0.1	0.4	0.002	0.001
N	7.5	20	0.01	0.01
Na	0.05	0.2	0.04	0.08
Nb	0.01	0.1	2E-6	3E-7
Nd	0.002	0.1	6E-5	0.002
Ne	0	0	0	0
Ni	0.05	1	0.02	0.005
No	0.002	0.1	5E-6	0.0002
Np	0.02	0.1	1E-5	0.001
O	0.6	2	0.02	0.2
Os	0.03	0.2	0.0001	0.002
P	1	3	0.02	0.05
Pa	0.01	0.1	5E-6	5E-6
Pb	0.004	0.1	0.0003	0.0008
Pd	0.1	0.5	0.0001	0.0002
Pm	0.002	0.1	6E-5	0.002
Po	0.001	0.1	0.0004	0.005
Pr	0.002	0.1	6E-5	0.002
Pt	0.1	0.5	0.0001	0.0002
Pu	0.001	0.1	1E-6	0.0001
Ra	0.04	0.2	0.001	0.001
Rb	0.2	2	0.01	0.03
Re	0.2	1	0.002	0.01
Rf	0.003	0.1	2E-5	0.0004
Rh	0.03	0.2	0.0005	0.002
Rn	0	0	0	0
Ru	0.03	0.2	2E-5	0.002
S	0.6	2	0.02	0.2
Sb	0.01	0.1	0.0001	0.001
Sc	0.002	0.1	6E-5	0.002
Se	0.1	0.5	0.01	0.1
Si	0.02	0.1	2E-5	0.0003
Sm	0.002	0.1	6E-5	0.002
Sn	0.3	1	0.001	0.01
Sr	0.3	4	0.002	0.01
Ta	0.002	0.1	5E-6	5E-6

...ds.[a]

F_f (beef) d kg^{-1}
2E-5
0.003
0.0005
5E-5
0
0.02
0.01
0.005
0.0008
0.0002
0.005
0.002
2E-5
0.05
—[d]
0.002
0.001
2E-5
6E-5
0.04
2E-5
0.03
0.03
0.05
0.01
0.002
0.002
2E-5
0.002
0.02
0.03
0.0002
0.03
0.0003
0.002
0.2
—[d]
5E-6
0
0.0004
0.01
0.002
0.04
0.004
0.002

TABLE 5.2—*Element specific transfer factors for terrestrial foods.*[a] *(Continued)*

Element	B_{vs}[b] (fresh vegetables)	B_{vas}[c] (dry forage)	F_m (milk) d L^{-1}	F_f (beef) d kg^{-1}
Tb	0.002	0.1	6E-5	0.002
Tc	5	40	0.001	0.0001
Te	0.1	1.3	0.0005	0.007
Th	0.001	0.1	5E-6	0.0001
Ti	0.001	0.1	0.01	0.02
Tl	0.2	0.6	0.003	0.02
Tm	0.002	0.1	6E-5	0.002
U	0.002	0.1	0.0004	0.0008
V	0.002	0.1	0.0005	0.01
W	0.8	3	0.0003	0.04
Xe	0	0	0	0
Y	0.002	0.1	6E-5	0.002
Yb	0.002	0.1	6E-5	0.002
Zn	0.4	1	0.01	0.1
Zr	0.001	0.1	6E-7	1E-6

[a]Values derived from Baes *et al.* (1984), Coughtrey and Thorne (1983), Coughtrey *et al.* (1983), Frissel (1989), IAEA (1982; 1993), Ng *et al.* (1977; 1982a; 1982b).

[b]Contains provision for inadvertent soil ingestion from unwashed vegetables. Minimum set to 0.001.

[c]Contains provision for soil uptake by grazing animals. Minimum set to 0.1.

[d]Tritium and carbon concentrations can be found by the specific activity method.

The concentration of a radionuclide in forage, $C_{forage,i}$, was estimated using Equations 5.2a and 5.2b. The procedure is essentially the same as that for estimating the concentration of a radionuclide in vegetables. Values for $C_{forage,i}$ were obtained in a manner similar to $C_{veg,i}$ in Equations 5.2a, 5.2b and 5.3 by adopting parameter values for forage plants taken from Table 5.1 (NCRP, 1984). The elemental plant/soil concentration ratios for dry forage, B_{vas}, were obtained from the same references that yielded B_{vs} values for vegetables taken from Table 5.2 (Baes *et al.*, 1984; IAEA, 1982; Ng *et al.*, 1982a). The ratio B_{vas} for dry forage includes provision for the uptake of contaminated soil by the grazing animal. For screening, a minimum value of 0.1 has been given to this ratio. This parameter is not permitted to go below 0.1 in order to account for the effect of soil ingestion by grazing animals. For screening, this lower limit is a reasonable value. The concentration for the milk pathway was calculated by substituting F_m for F_{animal}, where F_m is the transfer coefficient of the radioelement to milk (Table 5.2). The concentration for the meat pathway was calculated by substituting F_f for F_{animal} where F_f is the transfer coefficient of the radioelement to beef (Table 5.2). Values of F_m and F_f were taken from various publications of Ng and colleagues (Ng, 1982; Ng *et al.*, 1977; 1982b). Values of Q_{mf} and Q_{ff}

APPENDIX 2

SCREENING LEVEL EXPOSURE PATHWAYS ANALYSIS

APPENDIX 2: SCREENING LEVEL EXPOSURE PATHWAYS ANALYSIS

Methodology

The potential exposures of humans to sources of radioactivity in the environment may be estimated using exposure pathways analysis. To illustrate this concept, dose estimates were made for people assumed to live continually over the carbonatite. For the Base Case analysis, the average uranium (U) and thorium (Th) concentrations in the soil, based on the airborne survey of the Geological Survey of Canada (GSC), were assumed to be 5.3 ppm U and 23.8 ppm Th. The external gamma radiation exposures measured by the GSC were also used in the analyses. The exposure pathways examined were:

- Inhalation of airborne dust;
- Ingestion of produce;
- Ingestion of beef and milk;
- Incidental ingestion of soil; and
- External gamma radiation.

In addition to the Base Case (Scenario 1), Scenario 2 based on soil concentrations of 15 ppm U and 62.4 ppm Th was analyzed. These concentrations were in the mid-range of the values measured over the carbonatite by the GSC, and represent about 130 ha of the total of more than 1300 ha surveyed by the GSC. Finally, Scenario 3, which represents the Base Case plus the maximum off-site incremental air and soil concentrations estimated to result from 17 years of Niocan operations was analyzed (see letter of 6 June 2002 from SENES to R. Faucher of Niocan, which gave maximum on-site deposition rates).

For purposes of this study, the analysis focused on terrestrial exposure pathways since these pathways could be potentially impacted by Niocan operations. Exposures to indoor radon and to groundwater, which are highly variable from home to home, were not considered. (In any case, exposures via these pathways would not be significantly impacted by Niocan operations.) The potential exposures were maximized by assuming that the produce was grown in the area of the carbonatite, and the beef and milk cattle obtained their food from this same area. The analyses were done for both an adult and a 5-y old child since exposure estimates vary with age.

A summary of the dose estimates for this Base Case and the Input Parameters required for the assessment are shown in the attached tables. The results for Scenarios 2 and 3 are also attached. The reference sources for the majority of the input parameters are also given. It is beyond the scope of the present analysis to explain the basis for all of the numerous parameter values. However, specific details on any aspect of this analysis can be provided to the BAPE if required.

Results

The Base Case analysis (Scenario 1) estimated total doses of 505 and 809 μSv/y for the adult and child, respectively. The most significant exposure pathways for the adult were external gamma radiation and consumption of produce, while the consumption of milk (dairy) was also significant for the child.

For Scenario 2, based on the higher uranium and thorium levels measured over the carbonatite, the estimated doses increased proportionately to 1206 μSv/y and 1852 μSv/y for the adult and child, respectively. This comparison shows the potential variability of background radiation exposures in this area.

For Scenario 3, based on the estimated incremental impacts of Niocan operations, the resulting exposures increased to 520 μSv/y and 836 μSv/y for the adult and child , respectively. (Because of the very small potential increase in soil concentrations (< 0.5% averaged over 15 cm depth at the location of maximum off-site deposition), the corresponding very small increase in external gamma radiation levels (also < 0.5%) was not included.) Based on the conservative assumptions used in the analyses, the incremental doses are less than 30 μSv/y, a dose considered trivial or insignificant by many radiation protection authorities.

NIOCAN **BASE CASE - Average over Carbonatite**

Summary of Dose Estimates (μSv/y)

Pathway		Adult	Child
Inhalation		18.7	29.9
Ingestion	Above-Ground Produce	173	276
	Below-Ground Produce	70.9	110
	Meat	22.3	38.6
	Dairy	18.6	117
	Soil	1.85	16.1
External Gamma		199	221
Total		505	809

Summary of Adult Dose Estimate

		U-238 Series					Th-232 Series				Total
		U-nat	Th-230	Ra-226	Pb-210	Po-210	Th-232	Th-228	Ra-228	Ra-224	
Inhalation		0.10	0.08	0.06	16.5	1.27	0.22	0.14	0.35	0.03	18.7
Ingestion	Above-Ground Produce	0.69	0.79	39.8	81.5	16.7	1.28	3.84	15.2	13.7	173
	Below-Ground Produce	0.53	0.59	31.7	7.82	3.40	0.97	2.90	12.1	10.9	70.9
	Meat	0.16	0.04	2.25	3.84	14.1	0.07	0.20	0.86	0.77	22.3
	Dairy	0.14	0.00	6.83	4.30	2.32	0.01	0.02	2.60	2.35	18.6
	Soil	0.04	0.10	0.13	0.32	0.55	0.16	0.47	0.05	0.04	1.85
External Gamma											199
Total		1.66	1.60	80.7	114.3	38.4	2.69	7.55	31.1	27.8	505

Summary of Child Dose Estimate

		U-238 Series					Th-232 Series				Total
		U-nat	Th-230	Ra-226	Pb-210	Po-210	Th-232	Th-228	Ra-228	Ra-224	
Inhalation		0.17	0.11	0.09	26.3	2.06	0.26	0.23	0.58	0.04	29.9
Ingestion	Above-Ground Produce	0.61	0.58	44.0	130	30.7	0.97	9.46	23.2	36.8	276
	Below-Ground Produce	0.48	0.44	35.1	12.5	6.23	0.73	7.13	18.5	29.4	110
	Meat	0.14	0.03	2.49	6.13	25.9	0.05	0.48	1.31	2.08	38.6
	Dairy	0.51	0.01	30.3	27.4	17.0	0.02	0.16	15.9	25.3	117
	Soil	0.19	0.35	0.71	2.51	5.03	0.59	5.76	0.37	0.59	16.1
External Gamma											221
Total		2.10	1.53	113	205	86.9	2.63	23.2	59.8	94.3	809

INPUT PARAMETERS - BASE CASE

RECEPTOR CHARACTERISTICS

		Adult (20-39y age bracket)	Child (5 year old)	
Air intake rate:	indoors (m^3/h)	0.5	0.4	U.S. EPA 1997; sedentary activities
	outdoors (m^3/h)	1.6	1.2	U.S. EPA 1997; moderate activities
	daily average (m3/d)	14.7	12.1	calculated
Produce intake rate:	above-ground (g/d)	149	75	HC 1994; child is 0.5 adult
	below-ground (g/d)	119	60	HC 1994; child is 0.5 adult
Intake of beef (g/d)		183	91.5	HC 1994; child is 0.5 adult
Intake of dairy products (g/d)		283	566	HC 1994; child is 2x adult
Fraction of time spent at site		0.96	0.96	MOE 2001
Hours per day outside		2.5	3.15	MOE 2001
Days per week outside		7	7	MOE 2001
Weeks per year outside		52	52	MOE 2001
Fraction of site time outdoors		0.104	0.131	calculated
Fraction of site time indoors		0.896	0.869	calculated
Dirt ingested when outdoors	(mg/day)	20	50	HC 1994 (value for 7mo-4 year used for child)
Fraction of produce from site		1	1	assumed
Fraction of beef and milk from local sources		1	1	assumed

PHYSICAL SITE DATA

Outdoor total dust concentration (ug/m^3)	50
Contaminated fraction of airborne dust	0.50
Fraction of outdoor dust indoors	0.75
Respirable fraction of airborne dust	0.75

EXTERNAL GAMMA EXPOSURE

Fraction of outdoor time over contamination	1
Reduction factor when indoors	0.33

FOLIAR DEPOSITION FACTORS

Deposition velocity (m/s)	0.01
Fraction of dust intercepted	0.2
Produce yield (g/m^2)	1400
Weathering constant (1/d)	0.05
Vegetation growth period (d)	60

INPUT PARAMETERS - BASE CASE

ANIMAL DATA

Intake of vegetation by beef cattle (kg WW/d)	36.0	IAEA 1994
Intake of vegetation by dairy cattle (kg WW/d)	80.5	IAEA 1994
Intake of soil by cattle (kg/d)	0.4	McKone 1992
Fraction of time spent at site	1	

FOOD DATA

Fraction of contamination remaining

After Food Preparation:	above-ground cont.	1.00
	below-ground cont.	1.00

TRANSFER FACTORS

(taken from NCRP 1996)	U-nat	Th-230	Ra-226	Pb-210	Po-210	Th-232	Th-228	Ra-228	Ra-224
Soil-to-plant transfer factor (Bq/g ww per Bq/g soil)	2.0E-03	1.0E-03	4.0E-02	4.0E-03	1.0E-03	1.0E-03	1.0E-03	4.0E-02	4.0E-02
Food-to-beef transfer factor (Bq/g ww per Bq/d)	8.0E-07	1.0E-07	1.0E-06	8.0E-07	5.0E-06	1.0E-07	1.0E-07	1.0E-06	1.0E-06
Food-to-milk transfer factor (Bq/g per Bq/d)*	4.0E-07	5.0E-09	1.0E-06	3.0E-07	4.0E-07	5.0E-09	5.0E-09	1.0E-06	1.0E-06

* assumes density of milk same as water 1000 g/L

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RADIATION DOSE CONVERSION FACTORS (uSv/Bq)

		Uranium-238 Series					Thorium-232 Series			
		U-nat	Th-230	Ra-226	Pb-210	Po-210	Th-232	Ra-228	Th-228	Ra-224
Adult:	ingestion (sol)	4.7E-02	2.1E-01	2.8E-01	6.9E-01	1.2E+00	2.30E-01	6.90E-01	7.20E-02	6.50E-02
	ingestion (insol)	4.7E-02	2.1E-01	2.8E-01	6.9E-01	1.2E+00	2.30E-01	6.90E-01	7.20E-02	6.50E-02
	inhalation	8.7E+00	1.4E+01	9.5E+00	5.6E+00	4.3E+00	2.50E+01	1.60E+01	4.00E+01	3.40E+00
Child:	ingestion (sol)	8.4E-02	3.1E-01	6.2E-01	2.2E+00	4.4E+00	3.50E-01	3.40E+00	2.20E-01	3.50E-01
	ingestion (insol)	8.4E-02	3.1E-01	6.2E-01	2.2E+00	4.4E+00	3.50E-01	3.40E+00	2.20E-01	3.50E-01
	inhalation	1.8E+01	2.4E+01	1.9E+01	1.1E+01	8.6E+00	3.70E+01	3.20E+01	8.20E+01	5.90E+00

Source: ICRP Publication 72 (1996)

Gamma Radiation (Sv/Gy)

Adult	0.7
Child	0.8

Source: UNSCEAR (2000)

Measured Data - GSC Airborne Survey (Average Values)

Soil		Ratio to U in air (UNSCEAR 2000):	
U	5.3 ppm	Pb-210	500
Th	23.8 ppm	Po-210	50
Gamma	114.4 nGy/h		

Radionuclide Levels

		U-238 Series					Th-232 Series			
		U-238	**Th-230**	**Ra-226**	**Pb-210**	**Po-210**	**Th-232**	**Th-228**	**Ra-228**	**Ra-224**
Soil	Bq/g	0.06519	0.06519	0.06519	0.06519	6.52E-02	9.66E-02	9.66E-02	9.66E-02	9.66E-02
Air - Outdoor	Bq/m3	1.63E-06	1.63E-06	1.63E-06	8.15E-04	8.15E-05	2.42E-06	2.42E-06	2.42E-06	2.42E-06
Air - Indoor	Bq/m3	1.22E-06	1.22E-06	1.22E-06	6.11E-04	6.11E-05	1.81E-06	1.81E-06	1.81E-06	1.81E-06

	U-238 Series					Th-232 Series			
	U-238	Th-230	Ra-226	Pb-210	Po-210	Th-232	Th-228	Ra-228	Ra-224
VEGETATION									
Above-ground									
Dep. rate of contaminated dust (Bq/m²/d)	1.4E-03	1.4E-03	1.4E-03	7.0E-01	7.0E-02	2.1E-03	2.1E-03	2.1E-03	2.1E-03
Veg. concentration due to deposition (Bq/g ww)	3.8E-06	3.8E-06	3.8E-06	1.9E-03	1.9E-04	5.7E-06	5.7E-06	5.7E-06	5.7E-06
Veg. concentration due to uptake from soil (Bq/g ww)	1.3E-04	6.5E-05	2.6E-03	2.6E-04	6.5E-05	9.7E-05	9.7E-05	3.9E-03	3.9E-03
Vegetation concentration (Bq/g ww)	1.3E-04	6.9E-05	2.6E-03	2.2E-03	2.6E-04	1.0E-04	1.0E-04	3.9E-03	3.9E-03
Below-ground									
Vegetation concentration (Bq/g)	1.3E-04	6.5E-05	2.6E-03	2.6E-04	6.5E-05	9.7E-05	9.7E-05	3.9E-03	3.9E-03
ANIMAL PRODUCTS									
Intake by beef cattle (Bq/d)	30.9	28.6	120.1	104.3	35.3	42.3	42.3	178.0	178.0
Beef concentration (Bq/g ww)	2.5E-05	2.9E-06	1.2E-04	8.3E-05	1.8E-04	4.2E-06	4.2E-06	1.8E-04	1.8E-04
Intake by dairy cattle (Bq/d)	36.9	31.6	236.3	200.9	46.7	46.9	46.9	350.2	350.2
Milk concentration (Bq/g ww)	1.5E-05	1.6E-07	2.4E-04	6.0E-05	1.9E-05	2.3E-07	2.3E-07	3.5E-04	3.5E-04

NIOCAN **SCENARIO 2: Higher Levels in 130 ha of Carbonatite**

Summary of Dose Estimates (μSv/y)

Pathway		Adult	Child
Inhalation		52.0	82.9
Ingestion	Above-Ground Produce	431	655
	Below-Ground Produce	153	212
	Meat	59.8	102
	Dairy	43.5	254
	Soil	4.17	33.2
External Gamma		463	513
Total		1206	1852

Summary of Adult Dose Estimate

		U-238 Series					Th-232 Series				Total
		U-nat	Th-230	Ra-226	Pb-210	Po-210	Th-232	Th-228	Ra-228	Ra-224	
Inhalation		0.29	0.23	0.16	46.7	3.58	0.57	0.14	0.35	0.03	52.0
Ingestion	Above-Ground Produce	1.94	2.23	113	231	47.3	3.35	3.84	15.2	13.7	431
	Below-Ground Produce	1.51	1.68	89.8	22.1	9.62	2.53	2.90	12.1	10.9	153
	Meat	0.44	0.11	6.36	10.9	40.0	0.17	0.20	0.86	0.77	59.8
	Dairy	0.41	0.01	19.3	12.2	6.55	0.01	0.02	2.60	2.35	43.5
	Soil	0.12	0.27	0.36	0.89	1.55	0.41	0.47	0.05	0.04	4.17
External Gamma											463
Total		4.70	4.54	229	323	109	7.05	7.55	31.1	27.8	1206

Summary of Child Dose Estimate

		U-238 Series					Th-232 Series				Total
		U-nat	Th-230	Ra-226	Pb-210	Po-210	Th-232	Th-228	Ra-228	Ra-224	
Inhalation		0.47	0.32	0.26	74.5	5.82	0.69	0.23	0.58	0.04	82.9
Ingestion	Above-Ground Produce	1.74	1.65	125	368	86.8	2.55	9.46	23.2	36.8	655
	Below-Ground Produce	1.35	1.24	99.4	35.3	17.6	1.93	7.13	18.5	29.4	212
	Meat	0.39	0.08	7.04	17.35	73.4	0.13	0.48	1.31	2.08	102
	Dairy	1.45	0.03	85.7	77.54	48.1	0.04	0.16	15.9	25.3	254
	Soil	0.54	1.00	2.00	7.11	14.2	1.55	5.76	0.37	0.59	33.2
External Gamma											513
Total		5.94	4.33	319	580	246	6.89	23.2	59.8	94.3	1852

NIOCAN **SCENARIO 3: Base Case + Niocan**

Summary of Dose Estimates (μSv/y)

Pathway		Adult	Child
Inhalation		24.4	38.4
Ingestion	Above-Ground Produce	181	290
	Below-Ground Produce	71.5	111
	Meat	23.1	40.1
	Dairy	19.0	119
	Soil	1.86	16.2
External Gamma		199	221
Total		520	836

Summary of Adult Dose Estimate

		U-238 Series					Th-232 Series				Total
		U-nat	Th-230	Ra-226	Pb-210	Po-210	Th-232	Th-228	Ra-228	Ra-224	
Inhalation		0.24	0.20	0.13	16.8	1.44	1.63	1.04	2.61	0.22	24.4
Ingestion	Above-Ground Produce	0.72	0.87	40.2	83.5	19.1	1.83	5.48	15.4	13.9	181
	Below-Ground Produce	0.54	0.60	32.0	7.92	3.44	0.97	2.91	12.1	11.0	71.5
	Meat	0.16	0.04	2.27	3.93	14.8	0.07	0.20	0.87	0.78	23.1
	Dairy	0.15	0.00	6.91	4.40	2.48	0.01	0.02	2.64	2.38	19.0
	Soil	0.04	0.10	0.13	0.32	0.56	0.16	0.47	0.05	0.04	1.86
External Gamma											199
Total		1.85	1.80	81.7	117	41.8	4.66	10.1	33.7	28.3	520

Summary of Child Dose Estimate

		U-238 Series					Th-232 Series				Total
		U-nat	Th-230	Ra-226	Pb-210	Po-210	Th-232	Th-228	Ra-228	Ra-224	
Inhalation		0.40	0.27	0.22	26.9	2.34	1.96	1.70	4.35	0.31	38.4
Ingestion	Above-Ground Produce	0.65	0.64	44.5	133	34.9	1.39	13.49	23.5	37.4	290
	Below-Ground Produce	0.48	0.44	35.4	12.6	6.31	0.74	7.17	18.5	29.5	111
	Meat	0.14	0.03	2.51	6.26	27.1	0.05	0.50	1.32	2.11	40.1
	Dairy	0.52	0.01	30.6	28.0	18.2	0.02	0.18	16.1	25.7	119
	Soil	0.19	0.36	0.71	2.55	5.09	0.60	5.78	0.37	0.60	16.2
External Gamma											221
Total		2.39	1.75	114	210	94.0	4.75	28.8	64.2	95.6	836

APPENDIX 3

RADON CONCENTRATION, EXPOSURE AND DOSE

APPENDIX 3: RADON CONCENTRATION, EXPOSURE AND DOSE

Although one usually speaks of the risks of exposure to Rn-222 (radon), it is actually the short-lived decay products of radon that are the source of the potential risk. This is because the decay products are radionuclides of solid elements (polonium, bismuth and lead) that attach to dust particles and the surface of the lungs when inhaled, while the inert gas radon does not remain in the lungs.

The evaluation of the exposure to radon and its decay products must therefore take account of the concentration of the radon decay products. This concentration is expressed in terms of the total alpha particle energy in units of J/m^3, or in the historical unit, working level (WL). One WL is equivalent to an activity concentration of approximately 3700 Bq/m^3 of radon in equilibrium with its decay products, i.e.

$$1 \text{ WL} = 3700 \text{ Bq/m}^3 \text{ radon and decay products in equilibrium}$$

Because of normal air movement, radon decay products are seldom in equilibrium with radon. The equilibrium factor F is defined as the ratio of the total potential alpha energy of the given radon decay product concentration to the total potential alpha energy of the decay products if they were at equilibrium with the radon. It therefore follows that:

$$\text{Concentration of radon decay products (WL)} = \text{radon concentration (Bq/m}^3\text{)} \times F / 3700$$

F is variable but a typical annual average indoor value is F = 0.4 (UNSCEAR 2000). The equilibrium equivalent concentration (EEC) is that concentration of radon at equilibrium that would result in the same radon decay concentration, or

$$\text{EEC (Bq/m}^3\text{)} = \text{radon concentration (Bq/m}^3\text{)} \times F$$

Exposure to radon decay products is historically expressed in terms of working level months (WLM) where 1 WLM is exposure to 1 WL for one working month (170 hours). Therefore,

$$\text{WLM} = \text{WL} \times \text{exposure duration (h)} / 170 = (\text{Rn (Bq/m}^3\text{)} \times F / 3700) \times \text{duration (h)} / 170$$

Based on the risks of exposure to radon derived from epidemiological studies, the dose from exposure to 1 WLM is considered equivalent to 4 mSv for a member of the public (ICRP 1993):

$$1 \text{ WLM} = 4 \text{ mSv}$$

Example Calculation of Radon Dose

To illustrate the above concepts, the annual indoor exposure (assuming 90% of the year (8766 h) indoors) to a typical indoor radon concentration of 50 Bq/m^3 at 40% equilibrium (F = 0.4) is given by:

Annual indoor exposure (WLM/y) = (50 x 0.4 / 3700) x (8766 x 0.9) / 170 = 0.25 WLM/y

The annual dose from this exposure is:

Annual indoor dose (mSv/y) = 0.25 WLM/y x 4 mSv/WLM = 1.0 mSv/y

Typical outdoor radon concentrations (20 Bq/m^3) result in annual doses less than 0.2 mSv/y.

Thoron

These same concepts also apply to thoron (Rn-220) but with different coefficients because of the different alpha energies of the thoron decay products. Typical exposures to thoron indoors (0.3 Bq/m^3) result in annual doses of about 0.08 mSv/y. Outdoor exposures to thoron result in doses that are typically less than 0.01 mSv/y (UNSCEAR 2000).

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Paper

INDOOR EXPOSURE TO ^{222}Rn: A PUBLIC HEALTH PERSPECTIVE

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Abstract—**The objective of this study was to assess the lung cancer risk resulting from indoor radon exposure in the province of Québec, Canada, and to evaluate the efficacy of mitigation measures to reduce this exposure. Concentrations of radon were determined in a representative sample of houses, and the corresponding lung cancer risk estimates were generated using the BEIR IV model, taking into account smoking, residential mobility, and regional variations in radon concentrations. Mean (geometric) radon concentrations in basements (n = 418) and on first floors (n = 319) were, respectively, 34.4 (95% CI—30.6 to 38.8) and 16.5 Bq m^{-3} (14.2 to 19.3). A total of 109 deaths from lung cancer are predicted to occur as a result of this exposure in a cohort of 60,000 people. Detecting all residences with high radon concentrations (equal to or above 200 Bq m^{-3}) and implementing mitigation measures in each of them would reduce by 4 the number of lung cancer deaths attributable to indoor radon exposure. A reduction of 0.05% in the prevalence of smoking would prevent as many deaths from lung cancer as would radon mitigation. From a public health perspective, in order to reduce mortality from lung cancer, most efforts should be focused on smoking, not on the relatively minor and hardly preventable population risk arising from household radon exposure.**
Health Phys. 75(3):297–302; 1998

Key words: ^{222}Radon, indoor; cancer; risk analysis; health effects

INTRODUCTION

Radon (^{222}Rn) is a radioactive gas derived from the decay of radium (^{226}Ra), which itself is a decay product of uranium (^{238}U). These elements are present in all soils (Samet 1989). ^{222}Rn seeps into homes and may accumulate in concentrations substantially greater than those prevailing outdoors. A causal association between lung cancer and exposure to ^{222}Rn and its progeny has clearly been demonstrated in epidemiological studies on cohorts of miners (National Research Council 1988; Samet 1989; Lubin et al. 1995).

We conducted a survey in a representative sample of houses in the province of Québec (Canada) to document the extent of exposure to this radioactive gas and to estimate the associated health risks, taking into account regional variations in ^{222}Rn concentrations and the residential mobility in this population. The public health efficacy of mitigation measures designed to reduce ^{222}Rn levels in Québec houses also was assessed.

MATERIALS AND METHODS

Exposure assessment

The original plan for this study was to sample 1,000 homes located in two zones with different probabilities of finding elevated ^{222}Rn concentrations, based on geological indices. However, since only 1% of the Québec population resides in areas included in zone 1 ("high risk"), and because small communities in this zone were not randomly selected, but chosen according to geological indices, sampling for this risk analysis was restricted to houses located in zone 2 ("low risk"). Zone 2 houses were selected randomly among municipalities of more than 1,000 eligible homes, in all but two administrative regions of Québec (Lévesque et al. 1997). Two remote regions of northern Québec were excluded from this survey because of sampling difficulties. In each participating dwelling of the study, concentrations of ^{222}Rn were measured during two consecutive 6-mo periods by passive CR-39 alpha-track dosimeters (McGregor et al. 1987) in the master bedroom and at the lower level of the house. Geometric means of ^{222}Rn concentrations were calculated for each administrative region. Statistical comparisons between regions were effected by analysis of variance. More details regarding the selection of the population, ^{222}Rn measurements, and statistical analyses are available elsewhere (Lévesque et al. 1997).

Cancer risk assessment

Pulmonary cancer risk associated with indoor ^{222}Rn exposure was estimated using a multi-component model that incorporated a residential mobility model, a ^{222}Rn exposure model, and a cancer risk model. Monte-Carlo

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Table 1. Frequency distribution of annual ^{222}Rn concentrations in Québec homes.

^{222}Rn concentration (Bq m^{-3})	Basement (N = 418)			First floor (N = 319)			Second floor (N = 125)		
	n	%	cumulative	n	%	cumulative	n	%	cumulative
0–24	151	36.1	36.1	179	56.1	56.1	73	58.4	58.4
25–49	96	23.0	59.1	70	21.9	78.0	23	18.4	76.8
50–74	61	14.6	73.7	36	11.3	89.3	15	12.0	88.8
75–99	36	8.6	82.3	15	4.7	94.0	6	4.8	93.6
100–124	20	4.8	87.1	6	1.9	95.9	3	2.4	96.0
125–149	21	5.0	92.1	7	2.2	98.1	1	0.8	96.8
150–199	13	3.1	95.2	3	1.0	99.1	0	0.0	96.8
200–299	13	3.1	98.3	1	0.3	99.4	3	2.4	99.2
300–399	2	0.5	98.8	2	0.6	99.7	0	0.0	99.2
400–799	4	1.0	99.8	—	—	100.0	1	0.8	100.0
≥800	1	0.2	100	—	—	—	—	—	—
Geometric mean (Bq m^{-3})	34.4		(30.6–38.8)[a]	16.5		(14.2–19.3)	18.2		(14.0–23.5)

[a] 95% confidence interval.

simulations were performed using the @Risk software[§] in order to select at random values from predetermined statistical distribution of variables entering models.

The model describing the residential mobility of Canadians was designed to obtain plausible patterns of movement for individuals over a lifetime. Data on moves (frequency, distance) for various age groups were obtained from Statistics Canada. Since data were available only from individuals aged 15 y and older, residential mobility at younger ages was assumed to be similar to that of adults of childbearing age. The average number of moves for a lifetime was 14, ranging from 0 to 43. The location of the first house was selected at random from all administrative regions, with greater weight being allocated to more densely populated regions. Destinations were selected at random taking into account the distance and populations sizes for regions within reach. For each house occupied during lifetime, a ^{222}Rn concentration was selected at random from the distribution of first floor values documented in the present study for the specific region.

The lifetime risk of dying from lung cancer due to indoor ^{222}Rn exposure was estimated using the BEIR IV model (National Research Council 1988). For each year of life, the additional probability of dying from ^{222}Rn-induced lung cancer was computed, taking into consideration current and past ^{222}Rn exposure, sex, and the smoking status. These probabilities were then added over an assumed 80-y lifetime to obtain the additional risk of lung cancer for an individual. To link ^{222}Rn in mines to homes, a K-factor of 0.8 for children under age 10 y and of 0.7 for ages over 10 y was added to multiply the specific exposure in the home (National Research Council 1991). The prevalence of tobacco use in Québec was determined according to data from the 1992 Santé Québec Survey (Aubin et al. 1996).

The risk of dying from lung cancer was assessed under three different scenarios: 1) no residential mobility; 2) residential mobility; and 3) residential mobility and mitigation measures to reduce ^{222}Rn exposure. For each scenario, 60,000 simulations were performed, comprising 30,000 men and 30,000 women. Each sex group was divided up into three subgroups of 10,000 individuals according to smoking habits: non smokers, smokers and ex-smokers. The mitigation scenario assumed that measures to reduce ^{222}Rn levels exceeding 200 Bq m^{-3} would be introduced 1 y after moving into the house. It was further hypothesized that levels could be brought down to 74 Bq m^{-3}, which corresponds to the average level targeted in most cases of mitigation (Brill et al. 1994), but that can only be realistically achieved for 20% to 30% of homes (Oge 1994).

RESULTS

Frequency distributions of annual ^{222}Rn concentrations are presented in Table 1. The proportion of houses displaying ^{222}Rn concentrations greater than or equal to 200 (the Great Britain reference limit), 400 (the European reference limit), and 800 Bq m^{-3} (the Canadian reference limit) were, respectively, 4.8, 1.2, and 0.2% in basements, 0.9, 0.3, and 0% on first floors, and 3.2, 0.8, and 0% on second floors.

Regional variations of ^{222}Rn concentrations measured in basements and on first floors are shown in Table 2. Data for the second floor are not presented because of the small number of samples analyzed per region. Statistically significant differences were observed between regions for basement ($p < 0.0001$) and first floor ($p = 0.003$) measurements.

The influence of residential mobility on lifetime indoor exposure to ^{222}Rn is presented in Fig. 1. Median concentration was increased from 19.3 to 26.3 Bq m^{-3}, reflecting the tendency of the mobility model to move people into areas with higher ^{222}Rn levels. The variance was reduced from 1,497 to 379 when mobility was considered in the model. This phenomenon translates into a decrease in the number of individuals exposed during their lifetime to mean ^{222}Rn concentrations ex-

[§] V. 3.5d, Palisade Corp., Newfield, New York

Table 2. Regional variations of annual ^{222}Rn concentrations in Québec homes.

	Basement			First floor		
Administrative region	n	Geometric mean	95% confidence interval	n	Geometric mean	95% confidence interval
		Bq m^{-3}			Bq m^{-3}	
Bas St-Laurent	23	34.2	15.2–77.1	17	17.7	9.3–33.6
Saguenay/Lac-St-Jean	22	24.2	13.9–42.0	21	16.4	9.1–29.7
Québec	49	45.1	32.6–62.4	34	25.0	17.3–36.0
Mauricie	38	20.1	16.0–25.4	35	9.1	6.4–13.0
Estrie	13	41.4	27.0–63.5	10	16.2	7.6–34.4
Montréal	28	21.1	14.2–31.5	15	6.1	2.4–15.3
Outaouais	12	36.9	18.1–75.2	8	14.3	6.7–30.3
Abitibi-Témiscamingue	13	14.6	7.0–30.4	14	6.4	3.6–11.4
Côte-Nord	22	40.1	27.2–59.1	17	19.9	10.6–37.2
Gaspésie	11	95.6	59.4–154.0	10	40.9	23.8–70.3
Chaudière-Appalaches	26	58.2	39.5–85.7	19	28.1	14.7–53.8
Laval	22	40.4	28.7–56.9	14	24.6	16.3–37.2
Lanaudière	24	33.4	19.3–57.7	21	15.0	6.3–35.5
Laurentides	18	13.6	5.2–35.7	14	13.0	6.1–27.9
Montérégie	97	43.4	35.2–53.6	70	19.9	13.8–28.5

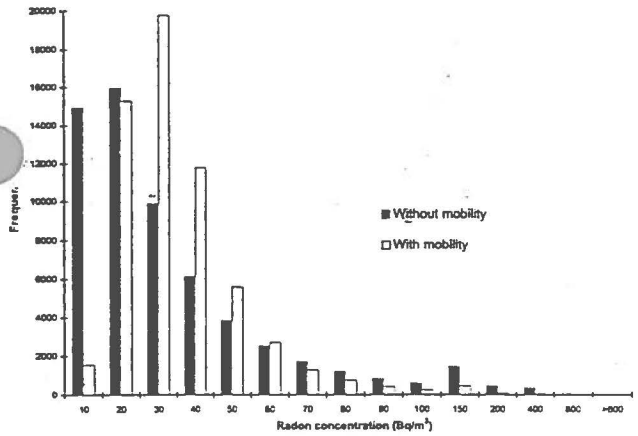

Fig. 1. Frequency distribution of lifetime average ^{222}Rn concentration in Québec homes according to residential mobility for a cohort of 60,000 individuals.

Table 3. Estimated number of individuals exposed to different lifetime concentrations of ^{222}Rn in Québec houses, considering or not residential mobility in the model.

	^{222}Rn concentration (Bq m^{-3})[a]		
Residential mobility	≥200	≥400	≥800
No	428	65	11
Yes	62	7	0

[a] Average concentration calculated over an 80-y lifetime; 60,000 simulations were performed.

eeding 800 Bq m^{-3}, 400 Bq m^{-3}, or 200 Bq m^{-3} (Table 3). For example, considering residential mobility reduces by a factor of seven the number of people exposed to a mean lifetime concentration of ^{222}Rn exceeding 200 Bq m^{-3} (0.7% of people in the model without mobility against 0.1% in the model with mobility).

Table 4 presents the number of additional lung cancer deaths predicted using the BEIR IV model, considering residential mobility and the prevalence of tobacco use in the Province of Québec during 1992 (non-smokers: 31.7%; ex-smokers: 34.7%; smokers; 34.3%). Taking mobility into account had little effect on the total number of expected lung cancer deaths, but the distribution of these deaths was different in relation to exposure (at age 30 y). Respectively, two (1.8%) and eight deaths (7.3%) out of the 109 additional lung cancer deaths were predicted to occur at mean lifetime concentrations over 400 and 200 Bq m^{-3} in the model without mobility, compared to 0 and two deaths (1.8%) in the model with mobility. In the latter case, 65% of deaths would occur at mean lifetime concentrations lower than 40 Bq m^{-3}, against 43% in the model without mobility.

This impact of mobility as well as the small number of homes with high ^{222}Rn concentrations explain the weak public health impact of mitigation measures applied to homes with more than 200 Bq m^{-3} (Table 5). While 109 lung cancer deaths were expected in a cohort of 60,000 people in the absence of mitigation measures, 105 deaths are still predicted to occur even if all residents moving into a house with high ^{222}Rn levels would adopt, after a 1-y delay, mitigation measures to reduce concentrations to 74 Bq m^{-3}. Results also indicate that in the model without mitigation, only 4.6% of deaths would occur among non smokers, compared to 63% in smokers and 32% among ex-smokers. The four deaths avoided when mitigation is considered are among smokers and ex-smokers.

DISCUSSION

Our sampling strategy allowed the extrapolation of ^{222}Rn indoor concentrations to all houses (excluding apartments) located in municipalities with 1,000 houses

Table 4. Lifetime ^{222}Rn-induced lung cancer mortality in a cohort of 60,000 individuals, considering or not residential mobility in the model.

	^{222}Rn concentration (Bq m^{-3})[a]				
Residential mobility	<40	40–199	≥200	≥400	Total
No	47	54	8	2	109
Yes	78	29	2	0	109

[a] Concentration in the residence occupied at age 30 y.

Table 5. Impact of mitigation measures on the number of deaths attributable to ^{222}Rn-induced lung cancer, according to sex and tobacco use.

	Without mitigation[a]	With mitigation
Men	74	71
Smokers	47	45
Ex-smokers	25	24
Non-smokers	2	2
Women	35	34
Smokers	22	21
Ex-smokers	10	10
Non-smokers	3	3
Total	109	105

[a] 60,000 simulations were performed for each mitigation scenario.

or more, which account for 72% of all homes in Québec. Few houses had concentrations exceeding 200 Bq m^{-3} on the first or second floors, where most residents are exposed to indoor ^{222}Rn. Comparisons by house levels clearly indicated greater concentrations in basements than on first and second floors. This is consistent with results demonstrated in other large surveys (Marcinowski et al. 1994; White et al. 1992).

In the United States, measurements conducted in 4,658 homes revealed median concentrations of 76.2 and 28.5 Bq m^{-3} for basements and first floors, respectively (Marcinowski et al. 1994). Schmier and Wicke reported median concentrations of 52 and 43 Bq m^{-3}, respectively, for basements and first floors of 6,000 houses in Germany (Schmier and Wicke 1985). In Switzerland, Crameri et al. obtained mean (geometric) concentrations for basements, first floors, and second floors of 234, 103, and 79 Bq m^{-3}, respectively, in 400 residences (Crameri et al. 1989). Finally, in Great Britain (Gunby et al. 1993) the mean (geometric) concentration measured for the most part on first or upper floors of 2,093 homes was 14.9 Bq m^{-3} (SD = 2.2), a value quite similar to that of 16.5 Bq m^{-3} (SD = 1.1) obtained in Québec for first floors. Globally, these comparisons indicate that average ^{222}Rn concentrations in Québec houses are among the lowest documented in studies using comparable methodologies.

Using the BEIR IV model and taking into consideration the prevalence of tobacco use, the number of lung cancer deaths in Québec attributable to indoor ^{222}Rn exposure was estimated at 109/60,000, or 182/100,000 (Table 4). ^{222}Rn levels documented on first floors were used in this modeling effort. Taking into consideration the occupancy time in basements would have led to somewhat higher risk estimates, since the mean ^{222}Rn level in basements is twofold greater than the first floor value. Unfortunately, no information is available regarding the time spent by Quebecers in basements. However, according to 1991 data from Statistics Canada, 41% of households in Québec are located in multilevel apartment buildings, where ^{222}Rn concentrations have been shown to be lower than in the basement or the main floor of single-family houses (Gunby et al. 1993; Lévesque et al. 1997). Therefore, it is likely that using first floor levels documented in single-family houses leads to an overestimation of the radon lung cancer risk for the entire Québec population.

The number of lung cancer deaths predicted were identical whether residential mobility was accounted for in the model or not. In a similar modeling effort, Warner et al. estimated 453 (with mobility) and 458 (without mobility) lung cancer deaths per 100,000 persons attributable to indoor ^{222}Rn exposure in the United States (Warner et al. 1995). The higher risk estimates reported by these authors are mostly due to the greater ^{222}Rn concentrations observed in the United States (Marcinowski et al. 1994) compared to those measured in the Québec survey.

Whereas residential mobility does not influence estimates of lung cancer deaths induced by exposure to residential ^{222}Rn, it does reduce markedly the number of expected deaths in the case of homes with high concentrations (Table 4). In a model that takes into account mobility, only two deaths from lung cancer out of 109 are expected to occur among individuals who, at 30 y of age, lived in a home with ^{222}Rn concentration exceeding 200 Bq m^{-3}. Warner and collaborators who also performed a cancer risk assessment taking into account residential mobility reported similar effects on ^{222}Rn-induced lung cancer risk for the American population (Warner et al. 1995, 1996).

Regarding the efficacy of mitigation measures, these authors stated that the universal adoption of the Environmental Protection Agency (U.S. EPA) recommendation, which calls for the mitigation of all houses displaying concentrations over 150 Bq m^{-3}, would be cost effective (Warner et al. 1996). This is likely not the case in Québec and in other countries such as Great Britain, where low concentrations of ^{222}Rn have been documented (Gunby et al. 1993). We estimated that mitigation measures adopted in all houses showing concentrations greater than 200 Bq m^{-3}, in order to reduce levels to 74 Bq m^{-3},

...ould at best prevent four deaths from lung cancer in a cohort of 60,000 people (Table 5). In fact, even this modest result may prove difficult to achieve, considering the American experience which showed the difficulty in setting up a program to reduce indoor exposure to ^{222}Rn.

In addition to the problem of establishing an adequate quality control program for companies performing ^{222}Rn analyses and mitigation measures, persuading the population to have their house screened for ^{222}Rn represents an important challenge. Despite very aggressive information campaigns conducted since the mid 1980's up until 1993, less than 6% of American home owners had verified ^{222}Rn concentrations in their houses (Field et al. 1993). Moreover, even if tests revealed high concentrations, mitigation measures were not more likely to be adopted (Field et al. 1993; Evdokimoff and Ozonoff 1992). Finally, Prill and coworkers reported that mitigation systems may deteriorate with time and therefore that ^{222}Rn concentrations must be monitored on a regular basis to verify their performance (Prill et al. 1990).

Public health authorities set priorities by identifying major problems, efficient methods to prevent them, and, finally, ways to implement preventive measures. Lung cancer is a major health problem in industrialized countries. In the Province of Québec as well as in the rest of Canada, it was found to be the main cause of mortality by ...ancer (National Cancer Institute of Canada 1996). Lung ...ncer is a dreadful pathology for which screening and treatment are of questionable efficiency. Hence, strategies to reduce lung cancer mortality must focus on the prevention of plausible causes. The first and by far the most important cause is tobacco use. In second place comes ^{222}Rn exposure. Even when acknowledging uncertainties and approximations associated with the BEIR IV model, the residential mobility factor implies that almost all ^{222}Rn-induced cancers will result from low-level exposures, thereby rendering mitigation measures inappropriate to reduce the collective risk. In this context, and considering difficulties in setting up an efficient program to reduce indoor ^{222}Rn exposure, such a program is of questionable use from a public health perspective.

Again, according to the BEIR IV model, more than 95% of expected deaths by lung cancer following indoor exposure to ^{222}Rn would occur among smokers, and the four deaths avoided when stringent mitigation measures are adopted would be among smokers and ex-smokers. Moreover, based on lung cancer mortality data according to the smoking status among Caucasians in the United States (American Cancer Society 1992), a 0.05% reduction of smoking prevalence in the population of Québec would likely save as many lives as would mitigation measures aimed at reducing indoor exposure to ^{222}Rn in ...ll the province. In addition to diminishing the incidence ... lung cancer, this would also lead to a reduction in the incidence of other cancers (larynx, pharynx, lip, esophagus, pancreas, bladder, kidney), cardiovascular diseases, as well as chronic obstructive pulmonary diseases (Fielding 1992). Furthermore, past campaigns designed to fight tobacco use in North America have proved very effective. Indeed, between 1964 and 1987, the prevalence of tobacco use in the United States decreased by 0.5% annually (Fielding 1992). During the same period in Canada, smoking prevalence decreased by 43% in men and 19% in women (Brangker 1990).

While on a collective basis, efforts should be focused on the fight against tobacco smoking, for people currently living in homes with high ^{222}Rn concentrations and envisioning to stay there for a long period of time, mitigation can significantly reduce their lung cancer risk, especially for smokers (Warner et al. 1996). Even if it is very difficult to predict ^{222}Rn concentrations in homes (Lévesque et al. 1997), screening and mitigation could be indicated if the length of stay envisioned in the home is long, especially in areas showing geological abnormalities. Local public health authorities in these areas should evaluate the need to implement ^{222}Rn programs, considering the magnitude of the exposure, resources available, and priorities.

CONCLUSION

On a collective basis and from a public health perspective, radon-induced lung cancer risk appears of relatively minor importance in the province of Québec as in several other jurisdictions, when compared to smoking. Resources devoted to the fight against lung cancer must first and foremost focus on lowering the prevalence of tobacco use rather than on the hardly preventable household exposure to radon gas.

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Paper

RADON IN RESIDENCES: INFLUENCES OF GEOLOGICAL AND HOUSING CHARACTERISTICS

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Abstract—^{222}Rn is a radioactive gas emitted during the decay of ^{238}U. ^{222}Rn is a recognized lung carcinogen in humans and a common indoor air contaminant. This paper describes the results of research undertaken in 894 residences of the Province of Québec (Canada), in which one of the objectives was to evaluate the influence of geological and housing characteristics on ^{222}Rn levels. After a random selection of homes, ^{222}Rn concentrations were measured with alpha track detectors in the basement and the main bedroom during two consecutive 6-mo periods. Geological subsoil characteristics were determined from various sources (e.g., geological maps, databanks on uranium sampling in lake and stream sediments), and housing characteristics were documented with a questionnaire. Statistical variance analysis of data indicates that geological factors only explain 5% and 4.5% of the variations in ^{222}Rn concentrations, respectively, in the basement and on the first floor. When variables relative to housing characteristics are added, the analysis explains only 18% and 15% of the variations in ^{222}Rn concentrations in the basement and on the first floor. These results illustrate the difficulties in predicting ^{222}Rn concentrations in homes.

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Key words: radon; ^{222}Rn, indoor; geology; radioactivity

INTRODUCTION

^{222}Rn is a radioactive gas resulting from the decay of ^{226}Ra, the latter a product from the decay of ^{238}U. These two elements are present in most soils (Samet 1989). Since ^{222}Rn is a gas, it diffuses from soils and rocks to outdoor air, and it can also contaminate indoor air by infiltrating housing foundations.

Causal association between exposure to ^{222}Rn and lung cancer has been demonstrated in epidemiological studies carried out on cohorts of miners (Samet 1989, National Research Council 1988). This relation was also confirmed by meta-analysis of eleven studies carried out in these populations. Data summation of 2,700 cases of lung cancer among 68,000 workers demonstrated once again a linear dose-effect relationship between cases of lung neoplasms and exposure to ^{222}Rn (National Institute of Health 1994).

Many studies carried out around the world document the importance of ^{222}Rn housing contamination (International Agency for Research on Cancer 1988). In view of insufficient data relative to ^{222}Rn exposure in the Province of Québec (population 7 million) in Canada, we conducted a study on housing contamination. One of the objectives was to identify determining variables (i.e., geology, home characteristics) regarding housing contamination by ^{222}Rn. This paper discusses the results of this objective of the study, in particular in view of determining the predictive power of these indicators.

MATERIALS AND METHODS

Sampling

For the study, we planned to sample 1,000 homes, divided into two zones according to the theoretical probability of finding higher concentrations of ^{222}Rn based on known geological constraints: zone 1 = areas at risk, and zone 2 = areas without risk.

During the selection process, we tried to include all types of homes (bungalows, duplexes, row housing, trailer houses), excluding apartment buildings. However, because of methodological problems inherent to the selection process, it was impossible to exclude them, and some apartments (n = 25, 2.9%) located in a basement or on a first floor were therefore included in the study.

Random sampling was done through telephone and electoral lists. On the basis of a pre-test study conducted in Québec City on a sample of 90 homes, which led to the recruitment of 29 homes, a total of 3,500 letters were sent out to meet the study's 1,000 homes objective.

Zone 1. Sampling for zone 1 was done based on the mapping of regions at risk. From available data, we proceeded to determine areas more likely to be contaminated by ^{222}Rn using different criteria (Martel 1991).

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We used seven indicators to determine which communities to include in a zone at risk, namely:

- Favorable geological context likely to bear radioelements (R.F.C.) according to the judgment of an expert in the prospection of uranium;
- Uranium concentrations (measured or extrapolated) greater than 4 mg kg^{-1} in lake and stream sediments (L.S.S.);
- Equivalent uranium concentrations greater than 2 mg kg^{-1} evaluated by airborne radiometric mapping (A.R.M.);
- Uranium-bearing mineral deposits (U.M.D.);
- Uranium concentrations greater than 20 $\mu g\ L^{-1}$ in underground water (U.C.W.);
- Radioactive mining tailings (R.M.T.); and
- Favorable geological formations (F.G.F.) based on the study of geological maps.

According to these criteria, we selected 42 communities representing the major part of the area inhabited identified by these indicators. The number of homes selected in each of the communities was established according to the number of homes that fit the inclusion criteria; in all, 420 homes were included in zone 1.

Zone 2. For zone 2, the 500 homes required were divided up according to the number of eligible homes from each of the administrative regions of Québec, excluding northern regions because of sampling difficulties (see Fig. 1). Afterwards, in each region we chose communities of more than 1,000 eligible homes according to size of population and geographic situation. This procedure lead to a lack of complete coverage of the territory. However, it allowed an increase in precision regarding more densely populated regions. The number of participants selected per community was then established in accordance with the total allocated to a region and with the number of eligible houses in the town; in all, 474 houses were recruited in zone 2. The total for this project was 894 homes.

^{222}Rn concentrations measured in the air

To document ^{222}Rn levels, we used passive alpha-track type dosimeters made of CR-39 plastic. These devices were developed by Health Canada (McGregor et al. 1987) and recently used in a major epidemiological study in Winnipeg, Manitoba (Létourneau et al. 1992). The detection limit of these monitors is 47 KBq h m^{-3} (McGregor et al. 1987), which is sufficient to measure a concentration of 10 Bq m^{-3} for a 6-mo exposure.

We undertook sampling during two consecutive periods of approximately 6 mo, one during the cold season (November to April) and the other during the warmer season (May to October). Annual concentrations were quantified from results averaged by the number of days of exposure of each of the periods. A proportion of 54.8% (490/894) of the residences received four dosimeters to be installed by pairs: two in the master bedroom and two in the lower level of the house. For each pair, ^{222}Rn concentrations were established by calculation of the arithmetic mean of both readings. With these paired results, we calculated the correlation coefficient to establish the precision of the measurement. A 0.98 result was obtained indicating an excellent performance of the monitors. The other participants received two monitors, one for the master bedroom and the other for the lower level of the house.

For each home recruited, results were classified according to the floor on which the measurements were done (basement, first floor, and second floor) and according to the period of sampling (summer, winter, annual).

Characteristics of the homes

To document housing variables likely to influence residential ^{222}Rn concentrations (e.g., type of foundation, ventilation), the participants had to complete a self-administered form that was translated and adapted from the questionnaires developed, tested, and used by the California Department of Health Services (Liu et al. 1990).

Statistical analysis

Documented ^{222}Rn concentrations were stratified in three categories: summer, winter, and annual. For the three strata, data follow a log-normal distribution. Thus, geometric means were used for result description and statistical comparisons. Geometric means of ^{222}Rn concentrations were therefore compared by variance analysis for season, by geographical regions, and according to floor of sampling (Kleinbaum et al. 1987). Variables associated with housing were also examined in relation to the ^{222}Rn concentrations by one-way analysis of variance (Kleinbaum et al. 1987). Afterward, based on a p-value of 0.05, significant elements were included in an analysis of variance in order to control for confounding factors.

RESULTS

Table 1 describes the characteristics of the homes sampled. For each variable, percentages were calculated on the basis of the answers to the questionnaires.

Table 2 illustrates mean ^{222}Rn concentrations according to the floor and the season of sampling. As can be seen, annual arithmetic means invariably lie between summer and winter values. Surprisingly, this is not always the case for geometric means. This is due to the importance given to lower values in the calculation of the geometric mean, associated with the estimation of annual concentrations that integrate summer and winter data, as well as the more important imprecision of the measurements at lower concentrations. In certain cases, these latter two elements generated a larger number of zero values for winter data than for the annual data, therefore leading to a bias likely to slightly over-estimate the annual geometric mean.

Figs. 2, 3, and 4 indicate frequencies of summer, winter and annual concentrations of ^{222}Rn for the whole sample, in the basement, on the first floor, and on the

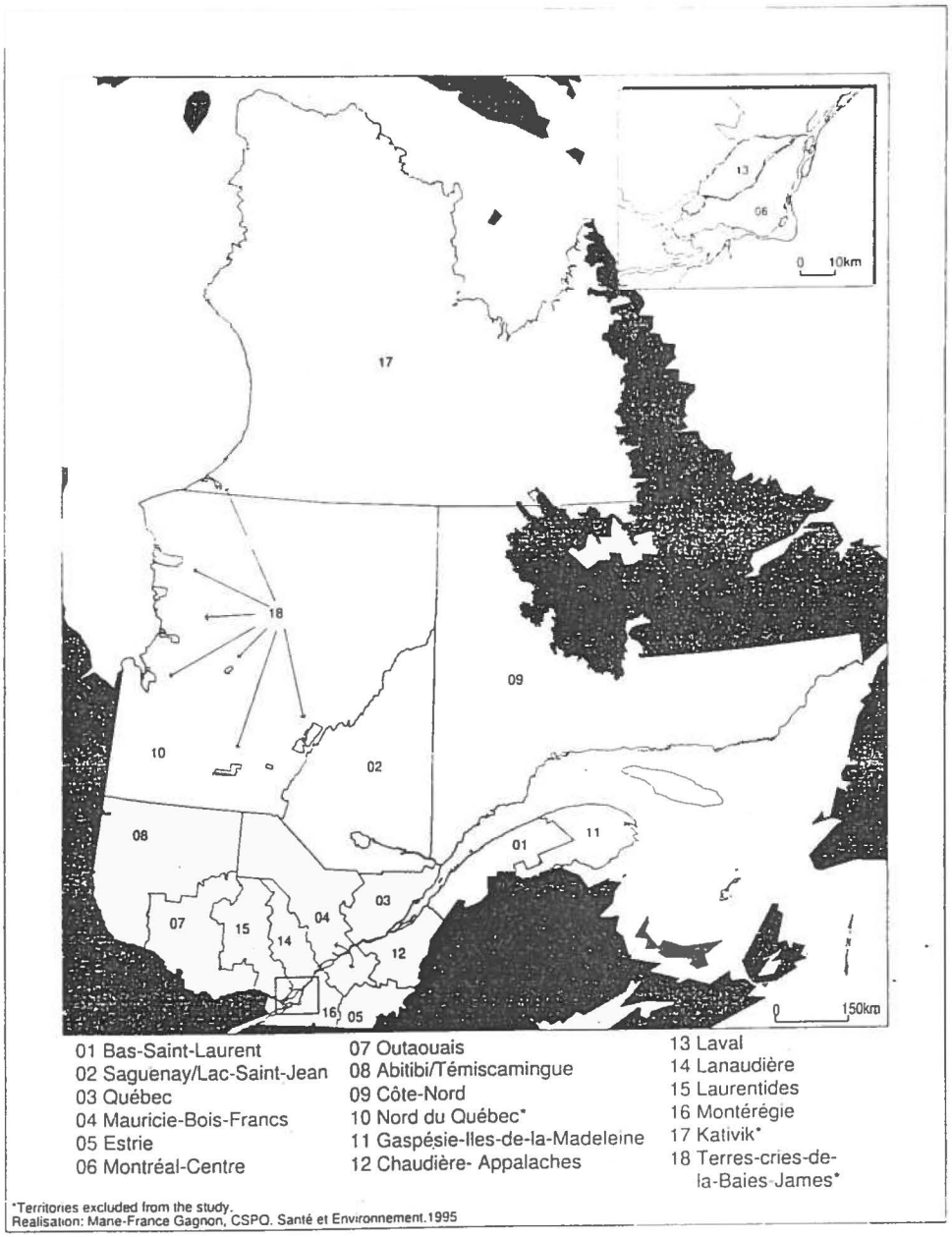

Fig. 1. Map of the administratives regions of the province of Québec.

second floor, respectively. In all cases, the data follow a log-normal distribution, and summer values are lower than winter values. This is confirmed by statistical analysis that reveals p-values of 0.0082 in the basement, ≤ 0.0001 on the first floor and ≤ 0.0001 on the second floor.

At the same time, concentrations found in the basement are clearly higher than those measured on the first floor without consideration for sampling period ($p \leq 0.0001$). Ratios of the annual geometric and arithmetic means of the values found in the basement to the ones measured on the first floor are 1.9 and 1.7. There is no difference between the means documented on the first floor and those of the second floor ($p = 0.68$). To simplify, we did not include data relative to the second floor in the following presentation of results.

Initially, we reexamined zone 1 according to the seven pre-defined indicators (R.F.C., L.S.S., A.R.M., U.M.D., U.C.W., R.M.T., F.G.F.). For each of these, annual geometric means of ^{222}Rn concentrations in homes classified as being at risk, in accordance with each criterion, were then statistically compared to the homes in zone 2. After examining the indicators used, houses itemized according to three of these have statistically higher mean concentrations ($p \leq 0.05$) than the homes in zone 2, in the basement (B) and/or on the first floor (FF).

Table 1. Housing characteristics.

Characteristics	*n*	%	Total respondents
Residence			873
Single-family home	747	85.6	
Duplex	77	8.8	
Apartment	25	2.9	
Row housing	24	2.7	
Market value ($)			771
<50,000	103	13.4	
50,000 to 99,999	435	56.4	
100,000 to 149,999	167	21.7	
≥150,000	66	8.5	
Exterior covering			887
Brick	336	37.9	
Vinyl	153	17.2	
Wood	117	13.2	
Aluminium	99	11.2	
Steel	48	5.4	
Stucco	35	3.9	
Stone	30	3.4	
Concrete	9	1.0	
Other	60	6.8	
Age of housing			891
0–4 y	52	5.8	
5–9 y	93	10.4	
10–25 y	361	40.5	
26–50 y	290	32.5	
>50 y	95	10.7	
Types of foundation			864
Crawlspace	34	3.9	
Basement	708	82.0	
Crawlspace and basement	39	4.5	
Crawlspace and concrete plate	79	9.1	
Piers	4	0.5	
Lived-in basement	634	78.2	811
Ventilation/air conditioning			
Kitchen fan-exterior ventilation	739	82.9	891
Fireplace	86	9.6	894
Wood stove	266	29.8	894
Thermal pump	60	6.7	894
Bathroom fan	338	38.0	889
Dryer exhaust exit	813	91.2	891
Air exchange system	242	27.4	883
Air conditioning	155	17.8	871
Gas central heating	25	2.8	894
Oil central heating	211	23.6	894
Electric central heating	618	69.1	894
Fixed electric heating	598	66.9	894
Air exchange system			240
Static	59	24.6	
Mechanical	154	64.2	
Heat	27	11.3	
Air conditioning			149
Heat pump	60	40.3	
Central	57	20.1	
Others	59	39.6	
Smokers in residence	332	37.2	892

The presence of a "favorable geological context likely to bear radioelements" (B: $p = 0.15$; FF: $p = 0.04$), of "concentrations equivalent in uranium greater than 2 mg kg^{-1} by airborne radiometric mapping" (B: $p = 0.11$; FF: $p = 0.02$), and especially, of "concentrations equivalent in uranium superior to 4 mg kg^{-1} in the sediments of lakes and streams" (B: $p \leq 0.0001$; FF: $p = 0.001$) seem to be associated with higher ^{222}Rn values in residences.

Geological formations that favored high ^{222}Rn concentrations were determined by projections from theoretical criteria (Martel 1991). These criteria were defined as sedimentary (black shales, phosphate) and igneous (silicic intrusive and alkaline intrusive) rocks known as containing high ^{238}U content (Wedepohl 1969). The sampling aimed, as with all other geological indicators, to validate these theoretical criteria. Furthermore, certain rock formations were chosen because, on a regional basis, they appeared to indicate higher levels of ^{222}Rn. In addition, on a provincial basis, we expected large variations between the various geological formations studied. Therefore, data were analyzed once again (statistically) by comparing documented concentrations in homes of each geological formation sampled to those in homes in zone 2. Housing of five geological formations out of eleven showed significantly higher concentrations ($p \leq 0.05$) than those in zone 2 for basements and/or first floors.

This new designation of favorable geological formations associated with other validated indicators (favorable geological context likely to bear radioelements, airborne radiometric mapping indicating uranium concentrations greater than 2 mg kg^{-1}, uranium concentrations greater than 4 mg kg^{-1} in lake and stream sediments) allowed us to redefine the zones at risk.

At the same time, housing characteristics were examined by univariate analysis. Significant statistical elements for housing characteristics were integrated in an analysis of variance to control for confounding factors to which we added indicators judged to be pertinent for determining zones at risk. Table 3 describes the two resulting models, one for the basement and the other for the first floor. These two models only explain 18% ($r^2 = 0.18$) and 15% ($r^2 = 0.15$) of the variation in ^{222}Rn concentrations in housing.

Results indicate that ^{222}Rn levels are higher in single-family homes built between 1982 and 1987 (age = 5–9 y). The presence of a crawlspace or a concrete slab associated with a basement increases the ^{222}Rn concentrations in the basement. Construction on hilly terrain is a significant positive factor as much for the basement as for the first floor, while a mechanical air exchange system is likely to decrease concentrations in the basement. The presence of a fireplace increases ^{222}Rn levels in the basement. Finally, housing fitted with central heating (mainly oil) is generally less contaminated in the basement and on the first floor. Of all the variables studied, this latter one is the most important for its influence on ^{222}Rn concentrations in the basement ($r^2 = 0.0312$), as well as on the first floor ($r^2 = 0.0242$).

Among the indicators used to determine zones at risk, the most sensitive are "the presence of uranium concentrations greater than 4 mg kg^{-1} in lake and stream sediments" (B: $r^2 = 0.264$; FF: $r^2 = 0.0225$), and "favorable geological formations" (B: $r^2 = 0.0198$; FF: $r^2 = 0.0145$). Concentrations measured on the first floor of housing in "favorable geological context likely to bear

Table 2. Mean ^{222}Rn concentrations according to season and floor level sampling.

Level[a]	Season	Arithmetic mean		CI[b]	Geometric mean	CI[b]	Maximum value
		n	Bq m^{-3}	(Bq m^{-3})	(Bq m^{-3})	(Bq m^{-3})	(Bq m^{-3})
0	Year[c]	781	69.6	62.1–77.0	34.6	31.6–38.0	1477.2
0	Summer[c]	785	62.1	54.9–69.4	24.8	22.1–27.8	1590.2
0	Winter[c]	811	77.0	68.7–85.3	32.8	29.4–36.5	1361.6
1	Year	616	39.9	35.5–44.4	18.0	16.1–20.3	653.3
1	Summer	628	27.4	23.9–30.9	9.4	8.2–10.8	471.1
1	Winter	661	53.5	47.4–59.6	19.7	17.2–22.6	877.2
2	Year	207	40.5	30.7–50.3	17.2	14.0–21.1	718.6
2	Summer	224	29.2	22.9–35.6	8.8	6.7–11.5	484.5
2	Winter	229	50.8	37.7–63.9	17.8	14.4–22.0	957.9

[a] 0 = basement, 1 = first floor, 2 = 2nd floor.
[b] 95% confidence interval.
[c] For a year, the summer or the winter.

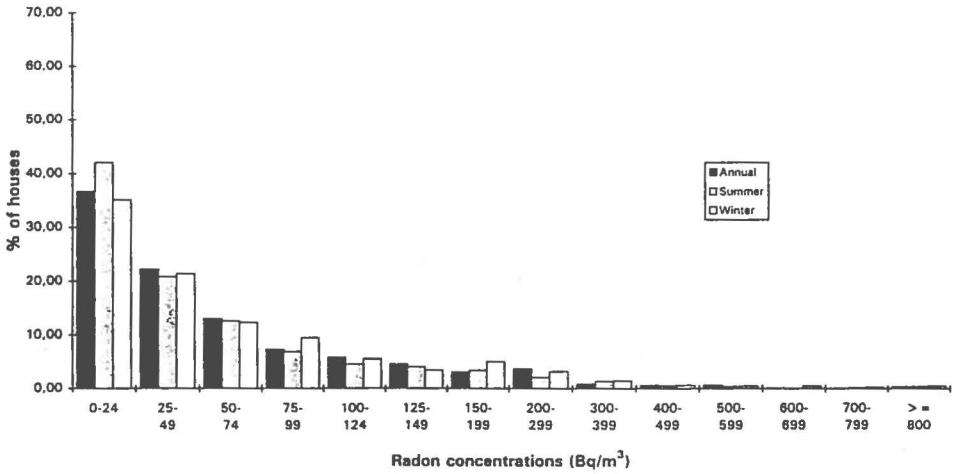

Fig. 2. Radon concentrations in basement (total sample).

radioelements" are equally statistically higher. The models do not reveal statistical differences for housing chosen according to airborne survey criteria, possibly because of limited sampling.

DISCUSSION

Data on ^{222}Rn concentrations itemized by season and by floor level show that summer exposures are lower than in winter. Many authors observed the same relation in Europe (Ulbak et al. 1988), in the U.S. (Perritt et al. 1990; Mose et al. 1991a, b) and in Canada (Létourneau et al. 1992). The annual geometric and arithmetic mean ratios between concentrations in basements and first floors were 1.9 and 1.7. The first quotient is similar to the one documented by Perrit et al. in a study that generated data from more than 2,000 houses from the state of New York (Perritt et al. 1990). As for the second, it is similar to the one calculated from measurements done in 2,916 houses in Winnipeg (Létourneau et al. 1992). There is no significant difference between ^{222}Rn levels measured on the first floor and on the second floor. The literature shows results a bit lower on the second floor compared to the first floor. Again, differences are often not important enough to reach the statistically significant threshold (Wolfs et al. 1984; Schmier and Wicke 1985; Poffijn et al. 1985; Cohen 1991). As indicated by the confidence interval, the limited number of sampled houses might have induced a slight imprecision in the means calculated on the second floor.

It is well known that some geological characteristics (I.e., permeable soil containing elevated levels of ^{226}Ra, favorable geological formations, etc.) influence ^{222}Rn levels in homes (Nazaroff and Nero 1988). Also, pressure differential between the inside of the home and the surrounding soil gas influence overall ^{222}Rn levels in homes located on a favorable soil (Nazaroff 1988). With statistical tests, we validated our geological indicators and redefined the zones at risk. In accordance with this new classification, annual geometric means for the modified zone 1 are clearly higher (79.1 Bq m^{-3} in the basement and 45.5 Bq m^{-3} on the first floor) than for zone 2 (34.4 Bq m^{-3} in the basement and 16.5 Bq m^{-3} on the first floor). However, all indicators combined only explain 5% of the variation in ^{222}Rn concentrations in the basement and 4.5% on the first floor.

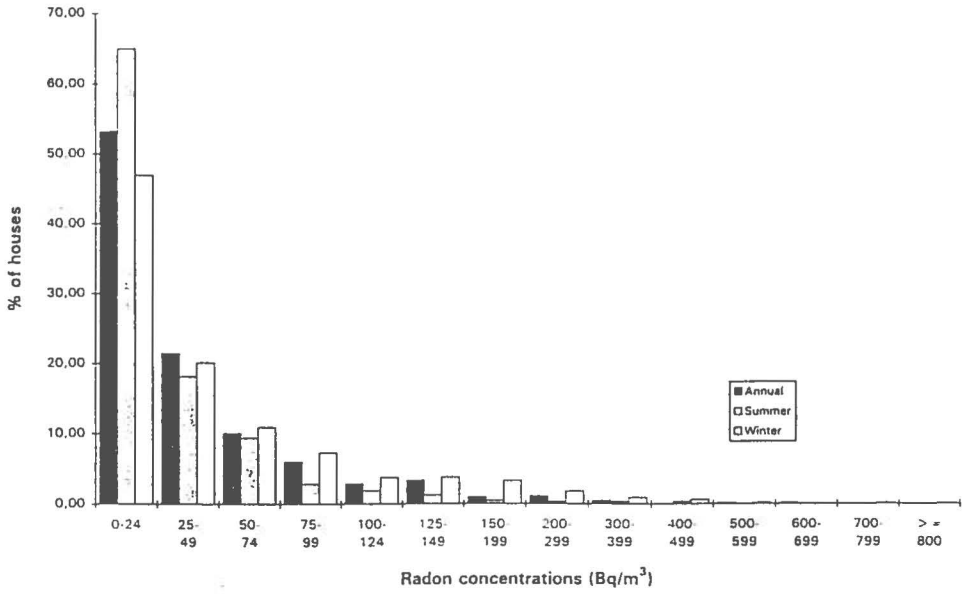

Fig. 3. Radon concentrations on the first floor (total sample).

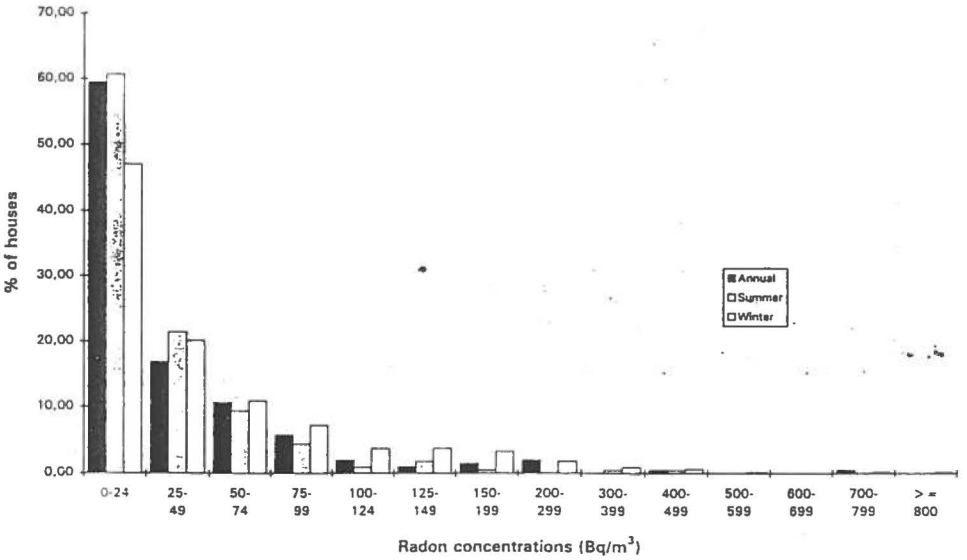

Fig. 4. Radon concentrations on the second floor (total sample).

Certain variables inherent to building characteristics also influence ^{222}Rn concentrations. Thus, the levels found in the basement and on the main floor of single-family homes are higher than in other types of housing. This was observed in Europe (Crameri et al. 1989; Gunby et al. 1993; Jönsson 1988) as well as in the U.S. (White et al. 1992), and even in China (Tianshan et al. 1987).

Age of housing also seems to be a contributing factor. Homes constructed between 1982 and 1987 (age = 5–9 y) seem to have higher ^{222}Rn levels. Cohen, in a study carried out in the U.S., also documented this same relationship (Cohen 1991).

The presence of a slab on grade or of a crawlspace favors the increase of ^{222}Rn concentrations in the adjoining basement. In the latter case, Field et al. (1993) made the same observation in Iowa, noting that if there is an opening between the crawlspace and the basement, logically, the most contaminated air of the crawlspace infiltrates by natural ventilation in the adjoining basement (Field et al. 1993). In Sweden, Swedjemark and Mjönes documented that houses built on concrete slabs had higher ^{222}Rn concentrations than those built on any other type of foundations (Swedjemark and Mjönes 1984).

Houses built on hilly terrain seem more contaminated than those set on even ground. Janssen et al. observed the same relation in Pennsylvania (Janssen et al. 1991). Under this condition, the high thermal gradient

Table 3. Variables influencing ^{222}Rn concentrations in housing.

Variables	Basement geometric mean				First floor geometric mean			
	n	Bq m^{-3}	*p*[a]	$r^{2[a,b]}$	*n*	Bq m^{-3}	*p*	r^2
Type of housing			0.05	0.0110			0.02	0.0186
Single-family home	670	36.5			534	19.1		
Duplex	71	25.6			43	14.6		
Apartment	13	20.6			18	8.9		
Row housing	22	19.9			8	11.5		
Age			0.01	0.0200			0.05	0.0175
0–4 years	49	38.7			29	20.0		
5–9 years	76	56.5			52	32.0		
10–25 years	324	30.2			285	15.6		
26–50 years	257	32.8			201	18.3		
>50 years	73	43.3			47	20.0		
Crawlspace with basement			0.01	0.0103			N.S.	0.035
Yes	39	64.2						
No	732	33.4						
Concrete slab with basement			0.01	0.0103			N.S.	0.020
Yes	77	54.3						
No	680	32.8						
Topography (layout)			0.01	0.0096			0.004	0.0151
On hillside	168	51.9			138	28.2		
level ground	600	30.8			469	15.7		
Air exchange system			0.01	0.0164			N.S.	0.0071
None	550	35.6						
Static	48	41.4						
Mechanical	146	27.6						
Heat	23	41.4						
Central heating			0.0001	0.0312			0.004	0.0242
None	122	44.1			96	18.3		
Gas	21	30.6			16	11.6		
Oil	191	23.1			147	12.9		
Electricity	447	38.7			357	21.1		
Fireplace			0.01	0.0104			N.S.	0.0046
Yes	75	46.2						
No	706	33.6						
Indicators								
R.F.C.[c]	20	52.2	0.1	0.0038	14	36.8	0.04	0.0080
M.G.F.[d]	61	78.5	0.0002	0.0198	52	42.7	0.0005	0.0145
A.R.M.[e]	6	78.9	0.65	0.0003	6	62.2	0.80	0.0001
L.S.S.[f]	29	112.7	0.0001	0.0264	28	61.1	0.0005	0.0225

[a] *P*-value and r^2 obtained by analysis of variance.
[b] Coefficient of partial determination.
[c] Favourable geological context likely to bear radioelements.
[d] Modified favourable geological formation.
[e] Equivalent uranium concentration superior to 2 mg kg^{-1} by airborne radiometric mapping.
[f] Uranium concentration superior to 4 mg kg^{-1} in lake and stream sediments.

between outdoor air and air in soil favors a larger air circulation, especially along the horizontal plan which is more permeable in stratified sediments. Also, the concrete slab of a house is more exposed to weathering and, therefore, more susceptible to cracking.

Improving ventilation by the use of a mechanical air exchange system seems to decrease ^{222}Rn concentrations in the basement. These results are similar to those demonstrated in Sweden by Swedjemark and Mjönes (1994).

The presence of central heating is, among all characteristics of housing studied, the one that influences ^{222}Rn concentrations the most. ^{222}Rn levels are lower in houses where central heating (mainly by oil and gas) is used. Researchers from Virginia also arrived at the same conclusion and attributed this fact to the negative pressure caused by the functioning of the furnace that requires incoming exterior air exceeding the influx of ^{222}Rn coming from the ground (Mose et al. 1991). Surprisingly, use of a fireplace seems to be a contributing factor in increasing ^{222}Rn levels in the basement.

Although the characteristics of the housing mentioned before are statistically associated with ^{222}Rn contents, their total impact on concentrations is not very important. Unfortunately, it is the same thing for geological indicators. In summary, models including all variables inherent to housing associated with geological indicators of exposure only explain 18% and 15% of the variation in ^{222}Rn concentra-

tions in the basement and on the first floor. To our knowledge, the only other authors who examined the effect of housing characteristics on ^{222}Rn contamination of interior air with the same statistical method (multivariate variance analysis) likely to control for confounding, could explain only 22% of the variance, even including the floor sampling as an independent variable in the model (Gunby et al. 1993). If we had not stratified for the floor sampling in our models, we would probably have explained a greater amount of the variation. These results emphasize the difficulties inherent to the prediction of ^{222}Rn concentrations in housing. Consequently, the data clearly indicate that when screening is recommended, geological or housing characteristics should not be considered as exclusion parameters.

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