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RAPID COMMUNICATION

Estimating Lung Cancer Mortality from Residential Radon Using Data for Low Exposures of Miners

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Some recent estimates of lung cancer risk from exposure to radon progeny in homes have been based on models developed from a pooled analysis of 11 cohorts of underground miners exposed to radon. While some miners were exposed to over 10,000 working level months (WLM), mean exposure among exposed miners was 162 WLM, about 10 times the exposure from lifetime residence in an average house and about three times the exposure from lifetime residence at the "action level" suggested by the U.S. Environmental Protection Agency. The extrapolation of lung cancer risk from the higher exposures in the miners to the generally lower exposures in the home is a substantial source of uncertainty in the assessment of the risk of indoor radon. Using the pooled data for the miners, analyses of lung cancer risk were carried out on data restricted to lower exposures, either <50 WLM or <100 WLM. In the pooled data, there were 115 lung cancer cases among workers with no occupational WLM exposure and 2,674 among exposed miners, with 353 and 562 lung cancer cases in miners with <50 WLM and <100 WLM, respectively. Relative risks (RRs) for categories of WLM based on deciles exhibited a statistically significant increasing trend with exposure in each of the restricted data sets. In the restricted data, there was little evidence of departures from a linear excess relative risk model in cumulative exposure, although power to assess alternative exposure–response trends was limited. The general patterns of declining excess RR per WLM with attained age, time since exposure and exposure rate seen in the unrestricted data were similar to the patterns found in the restricted data. Risk models based on the unrestricted data for miners provided an excellent fit to the restricted data, suggesting substantial internal validity in the projection of risk from miners with high exposures to those with low exposures. Estimates of attributable risk for lung cancer (10–14%) in the U.S. from residential radon based on models from the unrestricted data were similar to estimates based on the data for miners receiving low exposures.

One source of uncertainty in using risk models based on data for miners for estimating lung cancer risks from exposure to radon progeny in homes is that miners typically experience much higher exposures than individuals exposed to radon in homes. Although radon concentrations in some houses can reach or exceed levels measured in mines, mean cumulative exposure in a recent pooled analysis of data for miners was 162 working level months[1] (WLM), about an

[1]One work level (WL) equals any combination of radon progeny in one liter of air which results in the ultimate emission of 130,000 MeV of energy from α particles. WLM is a time-integrated exposure measurement and is the product of time, in units of working months, which is taken to be 170 h, and working levels. Radon concentration is measured in units of becquerels per cubic meter (Bq/m^3), where 1 Bq equals 1 disintegration per second. Under standard occupancy conditions, residence in a house at 1 Bq/m^3 results in approximately 0.005 WLM per year of exposure. The U.S. Environmental Protection Agency (EPA) recommends the testing of all homes and the mitigation of those that exceed 150 Bq/m^3.

order of magnitude greater than exposure from lifetime occupancy in an average U.S. home (46 Bq/m^3 $\times$ 0.005 WLM/Bq/m^3-year $\times$ 70 years = 16 WLM).

The National Research Council's Fourth Committee on the Biological Effects of Ionizing Radiation, known as the BEIR IV Committee, developed a model for the relative risk (RR) of lung cancer (called the BEIR IV model) based on an analysis of four studies of miners (*1*). This model has been updated in an analysis of pooled data from 11 cohort studies of underground miners exposed to radon, including the data used by the BEIR IV Committee (*2, 3*). The newest models specify a linear relationship in the excess RR (ERR) for cumulative WLM, with the exposure response modified by temporal variation in the effects of past exposure (time-since-exposure windows), current age and exposure rate. Extrapolation to indoor exposures also requires describing the exposure–dose relationship in homes compared to mines. This has been done with the K-factor, the ratio of exposure to dose for the general population in homes compared to male miners (*4*). However, while the models directly reflect risk at levels of exposures found in miners, the model-based estimates of risk may not accurately reflect risk at low exposures and low exposure rates, corresponding to residential levels of exposure.

One approach to assessing risk in homes would limit the use of data for miners to levels of exposure that are more typically experienced in residences. In this report, we use data from the pooled miner studies and compare models based on exposure data either under 50 WLM or under 100 WLM with models based on all the data for exposure of miners.

The 50 WLM and 100 WLM restrictions for the data for miners were selected for several reasons. These limits cover the range of cumulative exposures for most of the general population and minimize the inverse exposure-rate (protraction enhancement) effect, while still allowing for sufficient numbers of lung cancer cases. To compare, an individual living 30 years in a house at the EPA action level of 150 Bq/m^3, a concentration equaled or exceeded in about 6% of U.S. homes (*5*), receives about 20–25 WLM. Somewhat less than 1% of U.S. homes have radon concentrations of 370 Bq/m^3 or greater, and 30 years' residence in a house at this level results in 50–60 WLM of exposure.

An inverse exposure-rate (protraction enhancement) effect, i.e., for equal total exposure RRs are greater for exposures occurring at low exposure rate (and long duration) than for exposures occurring at a high exposure rate (and short duration), has been postulated (*6–8*) and has been observed in data from miner studies (*3, 9–12*). However, biophysical considerations suggest that 50 WLM and 100 WLM are about the levels of exposure where an inverse exposure-rate effect would be expected to diminish (*13*). An exposure-rate effect can occur only if a target cell, or the pool of cells with which it communicates, "experiences" more than one interaction with an α particle. At low exposures there is only a small probability of multiple α-particle traversals. This hypothesized diminution of the inverse exposure-rate effect at about 50–100 WLM has been observed in the data for the miners (*14*).

Comparisons between analyses of exposure-restricted data and unrestricted data were done in several ways. (1) Deviances (a measure of fit) for models derived from the unrestricted data and applied to the restricted data were compared to deviances of models fitted directly to the restricted data. This indicates how well models from the unrestricted data fitted the restricted data. Tests of homogeneity of effects by categories of WLM were also conducted. (2) Observed and fitted numbers of cases based on models from restricted and unrestricted data were compared for categories of WLM. (3) Attributable risks for lung cancer from indoor radon in the U.S. population were estimated based on models from restricted and unrestricted data.

DATA FOR THE MINERS AND METHODS

Data from the pooled analysis of 11 cohort studies of underground miners exposed to radon were used to evaluate the effects of exposure restrictions. The cohorts included studies of tin miners in Yunnan Province, China (*12*), iron miners in Malmberget, Sweden (*15*), fluorspar miners in Newfoundland, Canada (*16*), and uranium miners in the Czech Republic (*11, 17*), Colorado (*10, 18*), Ontario (*19*), New Mexico (*20*), Beaverlodge, Canada (*21*), Port Radium, Canada (*22*), Radium Hill, Australia (*23*), and France (*24*). These cohorts represent all known studies with estimates of WLM for individual miners. The pooling of data was described by Lubin *et al.* (*2, 3*).

The data from each study were summarized in a multi-way contingency table of observed lung cancer deaths and person-years of follow-up. The factors for each table included age, calendar year, cumulative WLM for temporal intervals prior to age at evaluation (referred to as exposure-time windows), duration of exposure and other factors. For time-varying factors, workers contributed to the appropriate category as time progressed. Thus most miners, regardless of their final cumulative exposures, contributed person-years to the restricted-exposure data sets. Only miners with cumulative exposures greater than 100 WLM prior to the start of follow-up failed to contribute at least some person-years to the restricted analysis. A 5-year lag interval for exposure was assumed. The individual tables were then appended to form the data set for the pooled analysis. Categorizations for the various factors in the tables depended on the cohort. For example, mean exposure for the Colorado miners was 807 WLM, while mean exposure for the Ontario miners was 31 WLM. It was therefore impractical to use the same categorization of WLM for all cohorts. Categories were selected to maximize information from each cohort. Although we refer to the complete pooled data as unrestricted, the Colorado study was limited to exposures under 3,200 WLM, a range across which the exposure–response trend was approximately linear.

For the restricted low-exposure analyses, the person-years table for each cohort was recomputed, emphasizing exposures under 100 WLM. For example, for the Colorado cohort, categories were specified as 0, 1–24, 25–49, 50–74 and 75–99 WLM. The combined person-years table used in the analysis of the low-exposure data was thus not a sub-table of the table used in the unrestricted analysis. This was necessary to limit the size of the summary table in the unrestricted analysis.

Since publication of the pooled analysis of 11 miner cohorts, 4 studies, the Chinese tin miners and the Czech, Colorado and French uranium miners, have been updated or modified. Modifications are described in Table I.

There has also been a reassessment of exposures for a nested case-control sample within the Beaverlodge cohort of miners, including all

TABLE I
New Data Available for the Analysis of Underground Miners

Cohort	Updated information	Related reference
China tin miners	New information indicated miners worked 313 days/year before 1953, 285 days/year from 1953–1984 and 259 days/year from 1985.	Unpublished information
Czech uranium miners	Exposure histories re-evaluated and follow-up improved. There were 705 lung cancer cases, compared to 661 in the previous analysis. Cohort was enlarged from 4,284 to 4,320 miners, including all miners who entered 1948–1959.	(*17*)
Colorado uranium miners	Follow-up extended from December 31, 1987, to December 31, 1990. In updated data, there were 336 lung cancer deaths <3,200 WLM used in the pooled analysis (and 377 total cases), compared to 294 lung cancer deaths <3,200 WLM (and 329 total cases) previously.	(*18*)
Beaverlodge uranium miners	Recalculation of WLM exposures, but were limited to a nested case-control sample and not used in the current analysis.	(*25*)
French uranium miners	Corrections of some (non-lung cancer) outcomes and exposure data. Changes were minor.	Unpublished information

lung cancer cases and matched control subjects (*25*). A linear RR model for cumulative WLM, w, was fitted,

$$RR = 1 + \beta \times w, \tag{1}$$

where β is the ERR per WLM. Although mean exposures were about 60% greater after reassessment, the estimate of β was also greater, increasing from 0.027/WLM to 0.033/WLM. Because of difficulties merging case-control data with cohort data, only the cohort data with the original exposure information were included in the current analysis.

In the pooled analysis (*2*), final models for risk prediction were based on the ERR and had the form:

$$RR = 1 + \beta \times (w_{5-14} + \theta_{15-24} w_{15-24} + \theta_{25+} w_{25+}) \times \phi_{age} \times \gamma_z, \tag{2}$$

where β is the exposure–response parameter, w $(= w_{5-14} + w_{15-24} + w_{25+})$ is partitioned in temporal exposure windows with w_{5-14}, w_{15-24} and w_{25+} defining the exposure incurred 5–14, 15–24 and 25 years and more prior to current age, and θ_{5-14}, θ_{15-24} and θ_{25+} represent the relative contributions to risk from exposures 5–14, 15–24 and 25+ years prior with $\theta_{5-14} = 1$. The parameters ϕ_{age} and ϕ_z define modifications to the exposure response and represent multiple categories of attained age and z, which is used to represent either exposure rate in WL or exposure duration. Model (1) is obtained from model (2) by fixing $\theta_{15-24} = \theta_{25+} = 1$ and $\phi_{age} = \gamma_z = 1$ for all categories. For some analyses, model (2) was fitted using only a single modification factor.

In analyses of the unrestricted data, the exposure–response estimate β was heterogeneous across the cohorts. Therefore, β in model (2) was replaced with a separate parameter for each cohort, $\beta_1,...,\beta_{11}$. A summary estimate for the exposure response was obtained as a weighted mean of the cohort-specific estimates, using inverse variances as weights, with variances adjusted for random effects for the individual estimates (*26, 27*). In the restricted analyses, the test of homogeneity of $\beta_1,...,\beta_{11}$ was not significant ($P = 0.16$ and 0.18 for <50 WLM and <100 WLM, respectively). Therefore, unless otherwise noted, we fitted a single β parameter. Confidence intervals (CI) for estimates of β are based on multiplicative standard errors adjusted for random effects.

Deviances were used to compare models. Deviances are related to twice the maximized log-likelihood and provide a general measure of the overall fit of the model. For nested models, the difference in the deviances provides a test of statistical significance for the added parameters. Under the null hypothesis of no effect, this difference is asymptotically χ^2 distributed with degrees of freedom equal to the number of added parameters.

RESULTS

Model (2) with exposure rate and with exposure duration, denoted TSE/AGE/WL and TSE/AGE/DUR in Table II, respectively, was fitted to the updated data. Parameter estimates changed only slightly from the estimates given in ref. (*2*).

Table III shows summary information for the unrestricted and restricted data sets. There were 274,161 person-years of observation among nonexposed workers with 115 lung cancer cases. (Using the unrestricted data, there were 266,547 person-years and 113 cases among "nonexposed" workers. The difference is due to the different categorizations used in the creation of the person-years tables.) For exposures <100 WLM, there were 564,772 person-years (64% of exposed person-years) and 562 lung cancer deaths (21% of exposed cases), while for exposures <50 WLM, there were 453,604 person-years (51% of exposed person-years) and 353 lung cancer deaths (13% of exposed cases). In the full data set, there were a total of 2,674 exposed lung cancer cases and 888,906 person-years. For cases, mean WLM, WL, duration of exposure, exposure rate and years since last exposure were markedly less in the restricted data, while age at lung cancer death was similar in the various groups. For the data for <50 WLM and <100 WLM, mean exposures for cases were 20 WLM and 40 WLM, respectively.

TABLE II
Comparison of Parameter Estimates from Summary Models[a] Using Previous and Updated Pooled Data for Miners

TSE/AGE/DUR model			TSE/AGE/WL model		
	Original data	Updated data		Original data	Updated data
$\beta \times 100$	0.39	0.55	$\beta \times 100$	6.11	7.68
Time since exposure (TSE) windows					
θ_{5-14}	1.00	1.00	ϕ_{5-14}	1.00	1.00
θ_{15-24}	0.76	0.72	ϕ_{15-24}	0.81	0.78
θ_{25+}	0.31	0.44	ϕ_{25+}	0.40	0.51
Attained age					
$\phi_{<55}$	1.00	1.00	$\phi_{<55}$	1.00	1.00
ϕ_{55-64}	0.57	0.52	ϕ_{55-64}	0.65	0.57
ϕ_{65-74}	0.34	0.28	ϕ_{65-74}	0.38	0.29
ϕ_{75+}	0.28	0.13	ϕ_{75+}	0.22	0.09
Duration of exposure (DUR)			Exposure rate (WL)		
$\gamma_{<5}$	1.00	1.00	$\gamma_{<0.5}$	1.00	1.00
γ_{5-14}	3.17	2.78	$\gamma_{0.5-1.0}$	0.51	0.49
γ_{15-24}	5.27	4.42	$\gamma_{1.0-3.0}$	0.32	0.37
γ_{25-34}	9.08	6.62	$\gamma_{3.0-5.0}$	0.27	0.32
γ_{35+}	13.6	10.2	$\gamma_{5.0-15.0}$	0.13	0.17
			γ_{15+}	0.10	0.11

[a]Fitted model had the form: $RR = 1 + \beta \times (w_{5-14} + \theta_{15-24}w_{15-24} + \theta_{25+}w_{25+}) \times \phi_{age} \times \gamma_z$, where β is the exposure–response parameter, cumulative exposure is partitioned into w_{5-14}, w_{15-24} and w_{25+} defining exposure incurred 5–14, 15–24 and 25 years and more prior to current age, θ_{5-14}, θ_{15-24} and θ_{25+} are the relative contributions from exposures 5–14, 15–24 and 25+ years prior, with $\theta_{5-14} = 1$, and ϕ_{age} and γ_z denote parameters for multiple categories of attained age and z, which represents either exposure rate in WL or exposure duration. TSE/AGE/DUR denotes the model with z representing duration of exposure, and TSE/AGE/WL denotes the model with z representing exposure rate in WL.

TABLE III
Numbers of Lung Cancer Cases and Person-Years, and Exposure Information for Cases Used in the Pooled Miner Analysis and in Analyses Based on Cumulative WLM Restrictions

	<50 WLM	<100 WLM	No restrictions
	Number of lung cancer deaths		
Cohort			
Yunnan, China	77	116	980
West Bohemia, Czech Republic	15	77	705
Colorado	15	22	336
Ontario, Canada	180	231	291
Newfoundland, Canada	21	24	118
Malmberget, Sweden	17	36	79
New Mexico	8	11	69
Beaverlodge, Canada	42	49	65
Port Radium, Canada	20	25	57
Radium Hill, Australia	52	53	54
France	22	33	45
	All data combined[a]		
Lung cancer deaths			
Nonexposed	115	115	113
Exposed	353	562	2674
Person-years			
Nonexposed	274,161	274,161	266,547[b]
Exposed	453,604	564,772	888,906
Mean values for exposed lung cancer cases			
WLM	19.7	40.0	493.6
WL	0.9	1.2	4.1
Years since last exposure	17.0	17.4	13.8
Duration of exposure	5.4	6.6	14.1
Attained age	58.0	58.6	58.5

[a]Totals exclude 115 workers and 12 lung cancer cases who were in both the Colorado and New Mexico studies. A total of 102 miners had exposures ≥100 WLM, including 11 lung cancer cases.

[b]The numbers of "nonexposed" person-years and lung cancer cases differ due to the cruder classifications for WLM within exposure time windows, age, year and other factors that define the person-years table.

TABLE IV
Numbers of Lung Cancer Cases and Relative Risks of Lung Cancer for Nonexposed and for Categories of Cumulative Working Level Months (WLM) Based on Deciles of Case Exposures, Using Data Pooled for Miners under 100 WLM

	Cumulative WLM										
	0	0.1–3.5	3.6–6.9	7.0–15.1	15.2–21.2	21.3–35.4	35.5–43.5	43.6–59.4	59.5–70.3	70.4–86.5	86.6–99.9
Cases	115	56	56	56	57	56	57	56	56	56	56
Person-years	274,161	111,424	95,727	72,914	67,149	57,890	42,068	25,622	40,220	28,076	23,682
Relative risk	1.00	1.37	1.14	1.16	1.45	1.50	1.53	1.69	1.78	1.68	1.86
95% CI	—	1.0–2.0	0.8–1.7	0.8–1.7	1.0–2.2	1.0–2.2	1.0–2.2	1.1–2.5	1.2–2.6	1.1–2.5	1.2–2.8
Mean WLM	0.0	2.4	5.3	12.4	17.3	33.1	38.6	53.2	63.3	81.1	91.4

Table III illustrates that restrictions on WLM alter the relative contribution of each cohort to the combined data. For example, in the unrestricted data, the Ontario study contributed about 10% of the lung cancer cases, while in the <50 WLM restriction the contribution from the Ontario study was nearly 40% of the cases. Similarly, the Beaverlodge, Port Radium, Radium Hill and French studies contributed *in toto* a substantially greater percentage of cases (29%) to the restricted analysis than to the unrestricted analysis (8%).

Table IV shows numbers of cases and person-years by nonexposed and WLM categories, defined by deciles of case exposures. The RRs were uniformly greater than one, with the highest RRs occurring in the highest WLM categories, although the RRs were not strictly monotone.

Table V shows results from fitting model (1) and model (2) with one modification factor. For model (1), estimates of β for the <50 WLM and the <100 WLM data restrictions were 0.0117/WLM with 95% CI (0.002, 0.025) and 0.0080/WLM with 95% CI (0.003, 0.014), respectively, and appear to fit the observed RRs well (Fig. 1). For the unrestricted data, the estimate of β was 0.0044 with 95% CI (0.002, 0.010). Note that because the relationship of lung cancer risk and WLM involves a complex association with attained age, and other factors, these estimates of β are not directly comparable.

For <100 WLM, Fig. 1 appears to suggest some inadequacy in the fit of the linear ERR model. We fitted two additional models: (1) a linear ERR model with an estimated intercept, $RR = \exp\{\gamma \times I(WLM)\}(1 + \beta \times WLM)$, where $I(.)$ is an indicator of exposure, taking value zero if

TABLE V
Parameter Estimates and *P* Values for Test of Models[a] Fitted to Pooled Data for Miners with Restrictions on Cumulative WLM

	Data restriction: <50 WLM					Data restriction: <100 WLM					No data restriction				
$\beta \times 100$	1.17	1.91	1.11	0.95	2.51	0.80	1.18	0.65	0.48	1.97	0.44	0.64	0.82	0.13	1.62
Time since exposure (years)															
5–14		1.00					1.00					1.00			
15–24		0.45					0.58					0.75			
≥25		0.46					0.53					0.39			
Attained age															
<55			1.00					1.00					1.00		
55–64			0.92					1.26					0.45		
≥65			1.43					1.69					0.12		
Duration of exposure (years)															
<5				1.00					1.00					1.00	
5–14				1.51					1.88					2.98	
≥15				2.14					2.79					3.33	
Radon concentration (WL)															
<0.5					1.00					1.00					1.00
0.5–0.9					0.18					0.48					0.57
≥1.0					0.42					0.31					0.28
P value[b]	—	0.53	0.94	0.66	0.17	—	0.41	0.83	0.21	0.06	—	<0.001	<0.001	<0.001	<0.001

[a]Models fitted to data were $RR = 1 + \beta \times w^* \times \phi_z$, where ϕ_z denotes parameters for categories of a modifying factor z, age, duration of exposure or exposure rate, with $w^* = w_{5-14} + \theta_{15-34} w_{15-24} + \theta_{25+} w_{25+}$ and $\theta_{15-24} = \theta_{25+} = 1$. For time since exposure, θ_{15-24} and θ_{25+} are estimated and $\phi_z = 1$.

[b]*P* value for test of null effect of modifying factors.

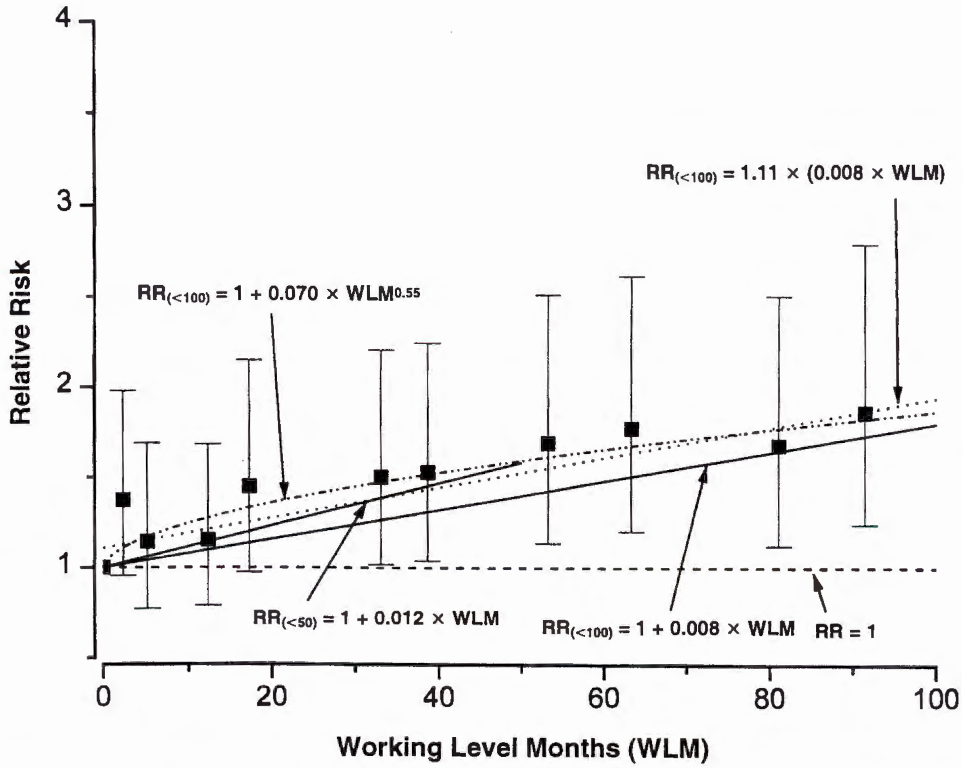

FIG. 1. Relative risks (RRs) of lung cancer from pooled data for miners, restricted to <100 WLM exposure, for zero exposure and categories of WLM exposure based on deciles. Also shown are, for data <50 WLM, the linear excess RR (ERR) model, RR = 1 + β × WLM and, for data <100 WLM, the linear ERR model, the linear ERR model with estimated intercept, RR = exp(γ)(1 + β × WLM), and a nonlinear RR model, RR = 1 + β × WLM$^{\kappa}$. The linear ERR model with intercept and the nonlinear model did not fit significantly better than the simple linear ERR model.

WLM = 0 and value one if WLM > 0; and (2) a nonlinear model, RR = 1 + β × WLM$^{\kappa}$. Neither the linear ERR model with estimated intercept (P = 0.58) nor the nonlinear model (P = 0.30) significantly improved the fit compared to model (1) (Fig. 1).

Time since exposure, attained age, exposure duration and exposure rate were individually significant effect modifiers in the unrestricted data, while in the restricted data the tests of homogeneity for each of these factors were not statistically significant (Table V). In line with biophysical predictions, the increasing trend with duration of exposure and decreasing trend with exposure rate suggest a diminution of the exposure-rate effect under 100 WLM, compared to the unrestricted data, although the variation was not statistically significant.

The patterns of modifications to the exposure response (β) in Table V for the restricted analyses were consistent with the patterns in the unrestricted data for all factors except attained age. The ERR declined with time since exposure and exposure rate and increased with exposure duration. Although exposure–response variations in the restricted data were nonsignificant, the exposure response increased with attained age for <100 WLM but decreased with attained age in the unrestricted data. To evaluate whether this result was due to correlations among the various factors, we fitted model (2) with reduced numbers of categories for age and duration as in Table V. With adjustment for time since exposure and duration of exposure, the exposure response decreased with attained age, much like the TSE/AGE/DUR model, although the improvement in the fit of the model was not significant.

The TSE/AGE/DUR and TSE/AGE/WL models (from the updated data, Table II) and the BEIR IV model were applied to the restricted data with parameter values fixed at their prescribed values and the deviances obtained. While formal statistical testing is not appropriate (the various models are not nested), the deviances for model (1) fitted to the restricted data were very similar to deviances obtained with the TSE/AGE/DUR and TSE/AGE/WL models (Table VI). The TSE/AGE/DUR model appeared to provide a marginally better fit than the TSE/AGE/WL model for data for <100 WLM, but there was little difference in the data for <50 WLM. The BEIR IV model (1) fitted the low-exposure data equally well.

The fit of the TSE/AGE/DUR and TSE/AGE/WL models based on the unrestricted data was evaluated further by estimating parameter values within the restricted data, using a reduced number of categories for age, duration and rate of exposure. Compared to model (1), there was no significant improvement in fit with the TSE/AGE/WL model (P = 0.54 for <50 WLM and P = 0.34 for <100 WLM) or the TSE/AGE/DUR model (P = 0.85 for <50 WLM and P = 0.51 for <100 WLM).

TABLE VI
Deviances for Models Fitted to Pooled Data for Miners with Restrictions on Cumulative WLM

Data restricted to:	Linear ERR[a]	TSE/AGE/DUR[b]	TSE/AGE/WL[b]
<100 WLM	3,089.7	3,086.8	3,095.2
<50 WLM	1,754.2	1,753.8	1,754.3

[a]Deviance based on maximum likelihood fit of the linear ERR model in cumulative WLM, i.e., RR = 1 + β × WLM, within the restricted data. There were 21,121 and 10,906 degrees of freedom for restrictions <100 WLM and <50 WLM, respectively.

[b]Deviance using model with parameters fixed at values derived from pooled miner data without restrictions. When parameters were estimated from the restricted data, deviances for the TSE/AGE/DUR and TSE/AGE/WL models were 3,084.4 and 3,082.9 for <100 WLM, respectively, and 1,751.3 and 1,749.0 for <50 WLM; with six degrees of freedom, these models did not improve fit compared to constant model.

Within WLM categories, the observed numbers of cases were similar to the estimated numbers of cases for the various risk models (Fig. 2).

Attributable risk of lung cancer for indoor radon was estimated using the TSE/AGE/DUR and TSE/AGE/WL models with parameter estimates from Table II, the BEIR IV risk model and the linear ERR model fitted to the <50 WLM data and to the <100 WLM data (*1*). We assumed a log-normal distribution for residential radon concentrations, with median 24.3 Bq/m^3 and geometric standard deviation 3.11 (*5*), and used 1985–1989 U.S. mortality rates for all causes and for lung cancer. Attributable risks for males based on unrestricted analyses (Table II) and the <50 WLM analyses were about 10–14% (Table VII). The 95% CIs (not shown) were about a factor of two, i.e. about 5–28%. Attributable risks for females were similar to males and were similar to prior estimates (*2*, *28*).

DISCUSSION

There are diverse uncertainties in using risk models based on data for miners for estimating lung cancer attributable to radon in the general population. One major uncertainty is the extrapolation of the generally higher radon-progeny exposures in miners to the lower exposures of the general population from residential radon. Maximum likelihood methods generally ensure that model-based predictions of the expected number of lung cancer cases for exposure levels

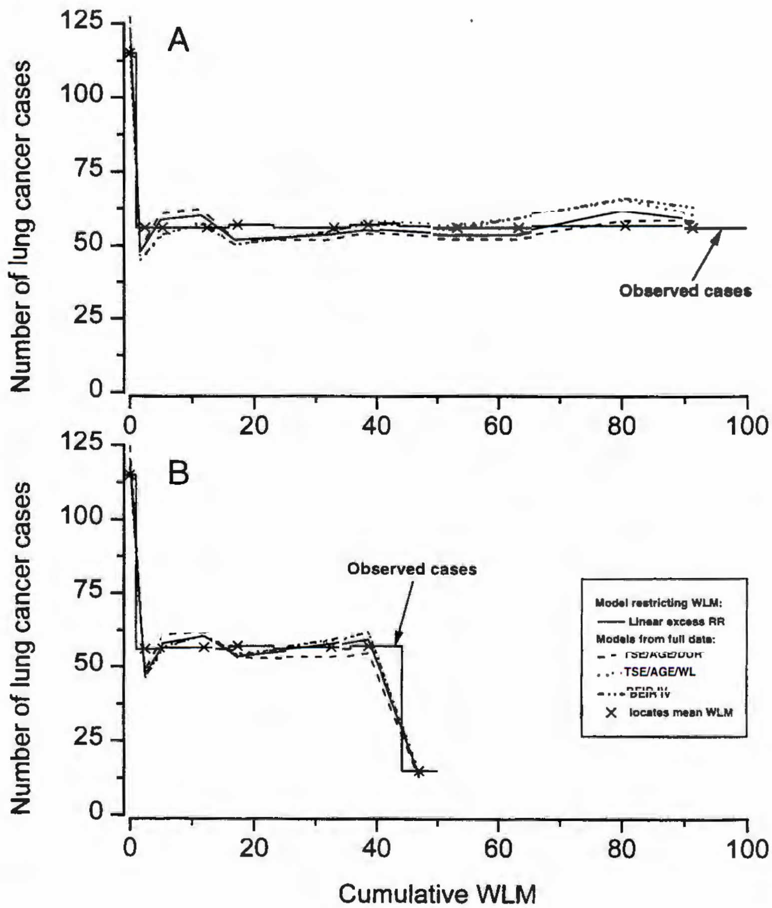

FIG. 2. Distributions of observed and model-generated fitted numbers of lung cancers by categories of zero exposure and deciles of WLM exposure within restricted data, <100 WLM (panel A) and <50 WLM (panel B).

within the bulk of the data for miners are close to the observed numbers of lung cancers. However, cumulative exposures from residential radon are generally much less than those received by miners, thus necessitating extrapolations to lower exposures. We reach several conclusions from the analysis of low exposures. (1) In our analyses there were substantial numbers of lung cancer cases and person-years in the data for miners restricted to <50 WLM and <100 WLM. These exposures can be accumulated from long-term residence in the highest 5–10% of U.S. homes. Duration of exposure can also be different in those exposed in mines and in homes. However, analyses of miners indicate that recent exposures (5–30 years) are the most relevant for lung cancer risk. In the data for <100 WLM, there were 64 lung cancer cases exposed to radon progeny for 15 years or more, 36 cases exposed for 20 years or more, and 11 cases exposed for 30 years or more. In the restricted data, the difference in exposure rates between homes and mines was also reduced. For exposed cases <50 WLM, mean exposure rate was about 3.6 WLM/year (= 19.7 WLM/5.4 years). For exposed cases <25 WLM (227 cases and 351,202 person-years), mean exposure rate was 2.0 WLM/year (mean exposure of 6.7 WLM and mean duration of 3.3 years). In comparison, living in a home with a radon level of 370 Bq/m^3 results in an exposure rate of about 1.8 WLM/year. (2) Current risk models which were developed using the full range of data for miners very adequately fitted the low-exposure data for miners. (3) In the restricted data, there was limited evidence for departures from a linear ERR model in cumulative exposure, but patterns of the variations in the exposure response with time since exposure, exposure rate, exposure duration and, after adjustment, attained age were similar to patterns in models from unrestricted data. However, data restrictions limited the power to assess variables as modifiers of the exposure response. (4) Attributable risk estimates for lung cancer deaths from indoor radon for the U.S. population based on models from unrestricted data for miners were similar to estimates from a linear ERR model in cumulative exposure from low-exposure data for miners.

Lubin and Boice (*29*) recently carried out a meta-analysis of eight indoor radon case-control studies of lung cancer, showing that miner-based estimates were generally consistent with results from the indoor studies. Our analysis of low-exposure data supports their conclusions, which were based on extrapolations using "high-exposure" miner-based models.

While our analysis demonstrated consistency in the miner-based extrapolations, it did not address limitations in the data, particularly the accuracy of exposure assessments, or the issue of whether the results from the analysis of the data for the miners could be generalized and applied to the progeny of radon in the home. Calculations of WLM for miners were often based on limited or no contemporary measurements of radon or of radon progeny, particularly in the earliest years of mining, and often relied on assumptions about equilibrium levels (for converting measured radon to radon progeny levels) and worker locations in miners[2] (*1*, *2*, *25*, *30–32*). Random error in exposure tends to depress the exposure–response relationship (*33*). However, patterns of exposure error were particularly complex. Errors were greatest in the early years of mining, when exposure rates were at their highest. This pattern would tend to reduce the effects of high exposure rates and thereby accentuate an inverse exposure-rate effect, although analyses using surrogate information indicated that it was unlikely that the inverse exposure-rate effect was entirely due to exposure errors (*2*, *3*). Systematic exposure measurement errors were also a potential problem in several cohorts[2] (*25*, *30*, *34*), but consequences are difficult to predict.

In summary, our analyses in miners using data for low cumulative exposures, which are more comparable to the level of cumulative exposures from homes, found estimates of lung cancer risks from radon-progeny exposure similar to those derived from all data for miners. Thus analyses suggested that there was substantial internal validity in the miner-based estimates for extrapolating from high to low cumulative exposures and that, while uncertainties in extrapolating risks to residential exposures remain, estimates from miners were relatively robust across a range of exposures which can be attained by long-term residents in high-radon houses.

TABLE VII
Comparison of Attributable Risk for the U.S. Male Population Estimated from Various Risk Models[a]

Model	Updated data	NCI Report (*2*)
TSE/AGE/DUR	0.099	0.102
TSE/AGE/WL	0.141	0.131
BEIR IV	0.081	
Linear ERR (<100 WLM)	0.078	
Linear ERR (<50 WLM)	0.109	

[a]Models of the form $RR = 1 + \beta \times w \times \phi_{age} \times \gamma_z$ where ϕ_{age} denotes parameters for categories of attained age, γ_z denotes parameters for categories of either duration of exposure or radon concentration, and $w = w_{5-14} + \theta_{15-24}w_{15-24} + \theta_{25+}w_{25+}$. The BEIR IV is specified by setting $\gamma_z = 1$ and $\theta_{15-24} = \theta_{25+}$, and the linear ERR model is specified by additionally setting $\theta = 1$ and $\phi_{age} = 1$. Calculations used 1985–1989 U.S. male mortality rates and a national radon concentration survey (*5*). Attributable risks for females were similar to males.

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