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LUNG CANCER MORTALITY AMONG FEMALES IN QUEBEC'S CHRYSOTILE ASBESTOS-MINING AREAS COMPARED TO THAT PREDICTED BY THE U.S. E.P.A. EXPOSURE-EFFECT MODEL

MORTALITÉ PAR CANCER PULMONAIRE DES FEMMES DES RÉGIONS MINIÈRES DE L'AMIANTE DU QUÉBEC COMPARÉE AUX PRÉVISIONS DU MODÈLE EXPOSITION-EFFET DE L'E.P.A.

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A Thesis submitted to the Faculty of Graduate Studies and Research in partial fulfillment of the requirements for the degree of Doctor of Philosophy.

December 1996

Lung cancer mortality among females in Quebec's asbestos-mining areas

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Résumé

Avec l'amélioration des conditions d'hygiène dans l'industrie de l'amiante et l'observation universelle d'amiante dans l'environnement général, la controverse de l'amiante est devenue « environnementale ». Pour évaluer les risques posés par les expositions environnementales à l'amiante, on a extrapolé linéairement les risques professionnels élevés vers des niveaux d'exposition 1 million de fois plus faibles dans la population générale. Malgré leur impact sur la santé publique et l'économie, de telles évaluations de risques n'ont jamais été validées.

Ceci est la première étude à comparer le risque de cancer du poumon d'une population exposée non professionnellement à l'amiante avec le risque prédit par le modèle exposition-effet de l'Environmental Protection Agency (EPA). L'étude a porté sur la population féminine des deux agglomérations minières de l'amiante au Québec, où la pollution de l'air par l'amiante était visible de 1882 à 1975 environ. Six petites études ont été réalisées pour évaluer les expositions passées dans l'air des villes minières et dans les maisons. Cinq experts reconnus internationalement ont évalué les niveaux d'exposition passés à partir des six études. Leurs estimations ont été couplées aux histoires d'expositions résidentielles et domestiques de 817 résidantes des agglomérations minières afin d'évaluer l'exposition cumulative des femmes de plus de 30 ans y ayant résidé entre 1970 et 1989. L'exposition moyenne s'élevait à 35 années-fibres-par-millilitre d'air, niveau auquel le modèle de l'EPA prévoyait un risque relatif de cancer du poumon de 2,47. La mortalité de 1970 à 1989 des femmes de plus de 30 ans dans les agglomérations de l'amiante a été comparée à celle des femmes de 60 agglomérations comparables. Le SMR s'élevait à 0,99 (IC95%: 0,78-1,26) et le PMR à 1,10 (IC95%: 0,88-1,38) pour le cancer du poumon. Selon le SMR, aucun excès de décès par cancer du poumon n'a été observé, à comparer à un excès de 105 prédit par l'EPA; selon le PMR, un excès de 6,5 décès était observé à comparer à 95 prédits par l'EPA. L'évaluation de risque de l'EPA a grandement surestimé le risque de cancer de poumon attribuable aux expositions à l'amiante dans cette population.

Abstract

With the improvement of working conditions in asbestos industries and the recognition that asbestos exposure is widespread, the asbestos controversy has shifted in the 1980s to the general environment. To assess lung cancer risks due to environmental asbestos exposure, risks in past asbestos workers have been extrapolated linearly to exposures 100,000 times smaller than historical occupational levels. Despite their enormous health and economic impact, such risk assessments have not been validated to this day.

This is the first study to compare the risk of lung cancer in a population non-occupationally exposed to asbestos with those predicted by the U.S. Environmental Protection Agency's (EPA) exposure-effect model. The study was carried out among the female population of Quebec's two chrysotile-mining agglomerations, where asbestos pollution has been visible commonly from 1882 to 1975. Six small exposure studies were conducted. These studies were synthesized by a panel of 5 international experts to estimate the historical levels of asbestos in the three main mining towns as well as the asbestos pollution brought home on the clothes of workers. These estimates were combined with a survey of lifetime neighbourhood and household exposures of female residents to assess the cumulative exposure of the females aged 30+ who resided in the mining agglomerations between 1970 and 1989. The average was 35 continuous fiber-years/mL. On the basis of the equivalent occupational exposure, the EPA model predicted a lung cancer relative risk of 2.47. Mortality of the female population of the asbestos-mining agglomerations aged 30+ was compared over the 1970-1989 period to that of 60 comparable agglomerations of Quebec. The lung cancer SMR was 0.99 (95%CI: 0.78-1.26) and the SPMR was 1.10 (95%CI: 0.88-1.38). Although the EPA model predicted 105 excess lung cancer deaths based on SMRs, none were observed in this population; based on SPMRs, an excess of 95 excess deaths was predicted, but only 6.5 observed. The EPA risk assessment on asbestos greatly overestimated the risk of lung cancer attributed to environmental asbestos exposure in this population.

THESIS OUTLINE

This research comprises three major components: 1) an assessment of the population's cumulative asbestos exposure, 2) the prediction of lung cancer relative risks resulting from the application of several asbestos exposure-effect gradients to this population's cumulative exposure, and 3) a mortality analysis against which to compare the predictions. While interconnected, these components are distinct enough to be presented in separate sections with separate Methods, Results and Discussion sections. The most crucial and sensitive component of the thesis is a retrospective asbestos exposure assessment embracing nearly 100 years of history of asbestos exposure in the female residents of Quebec's asbestos-mining towns. In the absence of historical direct exposure measurements, the exposure assessment combined numerous, dissimilar, indirect and unfamiliar approaches and methods. It was impractical to describe all these variegated methods in a monolithic Materials and Methods chapter of the thesis and then to present the results of all substudies based on these methods in a separate Results section. Therefore, only a cursory methodological overview is presented in the Overview of Study Design chapter in the Introduction part of the dissertation. The methodology of each substudy in the exposure assessment is presented in its own chapter in the Exposure Assessment part of the thesis.

Note Regarding the Study Period

Originally, the study period was designed to be 1950-1989. The exposure assessment was carried out successfully over that period. However, while mortality ratios had been computed for the whole follow-up period, decade-specific mortality analyses showed abnormally low mortality rates over the 1950-1969 period, at least for the agglomeration of Asbestos. Statistical variability could not account for these anomalies. The problem was likely due to weaknesses of the Provincial mortality datasets before 1970: a) municipality geocodes were not available for the mortality data before 1966 and we had to guess municipality names from abbreviated spellings, b) causes of death were not available on Provincial data records and were merged from Federal mortality datasets on the basis of individual record identification numbers, c) in the past, residents of the agglomeration of Asbestos were mostly treated in

Sherbrooke and Montreal hospitals at a time when municipality of residence and municipality of death were often confused on death certificates. To identify and correct the cause of the problem would involve a substantial effort with unknown chances of success.

In the present dissertation, the mortality analyses are presented only for the 1970-1989 period, for which the database and the analyses were definite[1]. Relative to previous studies, this 20-year follow-up period was still longer than in any previous non-occupational lung cancer study on Quebec's asbestos-mining towns [Loslier, 1983], agglomerations [Pampalon et al., 1982] or census divisions [Wigle, 1977; Graham, 1981]. Although dropping the 1950-1969 period reduced the number of person-years by 40%, statistical power was reduced only marginally because female lung cancer incidence was much lower across all ages and because the age distribution of the population was much younger in the 1950-1969 era than after 1970. As a result of the combination of these two factors, about 88% of all lung cancer deaths expected over the 1950-1989 period were in fact expected over the 1970-1989 period. Finally, it seems unlikely that the complete and corrected 1950-1989 data would produce materially different SMR and SPMR estimates from those presented here, since the 1950-1969 data for Thetford Mines which seemed all right did not exhibit a different distribution of mortality causes nor a different total mortality rate relative to the 1970-1989 period.

The Introduction and Exposure Assessment parts of the thesis refer to the tentative 1950-1989 follow-up period. From the Mortality Study chapter on, the results, discussions and the final conclusion pertain to the 1970-1989 period only.

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[1] We compared the number of deaths ≥ 30 years of age computed by agglomeration for "all cancer sites", "g.i.t. cancer", "respiratory cancer" and "non-malignant respiratory diseases" in the present study with similar agglomeration-specific mortality data computed for all ages by other investigators for the 1966-77 [Pampalon et al., 1982], 1974-78 [Pampalon, 1985] and 1979-1983 [Pampalon, 1986] periods respectively. The number of lung cancer deaths was only 1% lower in our data, while the number of deaths due to non-malignant respiratory diseases was 5% lower. This data check and other spot-checks corroborated our results for the 1970-1989 period.

Measure what is measurable and
make measurable what cannot be measured.

Galileo Galilei, 1564-1642

As far as the laws of mathematics refer to reality, they are not certain, and as far as they are certain, they do not refer to reality.

Albert Einstein, 1879-1955

The goal [in risk assessments] should be to bound the set of not clearly incorrect answers, rather than to focus solely on the most likely answer statistically.

Nicholas Ashford, professor, M.I.T., 1985

The movement of asbestos from the occupational to the non-occupational environment is a case study that will undoubtedly be followed in the future by other potentially toxic materials.

Morton Corn, president of the American Industrial Hygiene Association, 1986

ACRONYMS, ABBREVIATIONS, SYMBOLS

ACRONYM, ABBREVIATION, SYMBOL or IDIOM	DEFINITION
µm	micrometre
µg/m^3	micrograms per cubic metre of air (usually total dusts <100 µm diameter)
Ø	diameter of a fibre or other particle
AB	asbestos body, also a measure of lung burden: median typical asbestos body count per g dry lung tissue
ACM	asbestos-containing material
ae	aerodynamic equivalent (diameter of a particulate aerosol)
Agglo.	agglomeration (def. p.)
AI	Asbestos Institute
Asb.	either the municipality or the agglomeration of Asbestos
ASEM	analytic scanning electron microscopy
ATEM	analytic transmission electron microscopy
ATSDR	Agency for Toxic Substances and Disease Registry, U.S. Department of Health and Human Services
chry	chrysotile
CI	confidence interval
CL	confidence limit of the confidence interval (boundary)
CPSC or C.P.S.C.	U.S. Consumer Product Safety Commission
croci	crocidolite
D_{ae}	aerodynamic equivalent diameter
dom.	domestic or household (exposure to asbestos), i.e. living with an asbestos workers
DWLS	distance-weighted least squares (criterion for fitting a set of data points)
EDF	Électricité de France
EDXA	energy dispersive x-ray analysis
EM	electron microscopy
envir.	environmental (exposure)
EPA or E.P.A.	U.S. Environmental Protection Agency
E_{ty}	tons of total dust emitted in a year and town
f/µg	asbestos fibres per microgram of dry lung tissue (asbestos lung burden)
f/L	fibres (>5 µm) per litre of air (electron microscopy)
f/mL	fibres (>5 µm) per millilitre (usually optical microscopy)
f-y/mL	fibre-years per millilitre of air (cumulative exposure)
FR	U.S. Federal Register
F_{ty}	"efficiency" of filtration systems in a given year expressed as the gravimetric proportion of dusts retained by the filters
g	gram
geocode	Statistics Canada's geographic code identifying any municipality in Canada
GDF	Gaz de France
G	gravimetric "penetrance factor" of filtration bags and other dust controls, i.e. gravimetric proportion (%) of generated dusts that passes through dust controls and is emitted into the ambient air
gr	average annual improvement rate of the gravimetric penetrance factor in a given town; e.g. "gr=0.9" means that penetrance would improve or be reduced by 10% each year on average
GR	ratio of the penetrance factors at the beginning and at the end of a time period of many years

HEI-AR	Health Effects Institute - Asbestos Research
HSC or H.S.C.	U.K. Health and Safety Commission
HSE or H.S.E.	U.K. Health and Safety Executive
IARC	International Agency for Research on Cancer
ICD or I.C.D.	international classification of diseases
INSERM or I.N.S.E.R.M.	France's "Institut national de la santé et de la recherche médicale"
IRSST or I.R.S.S.T.	Quebec's "Institut de recherche en santé et sécurité au travail"
IRR	incidence rate ratio
ISC-LT	"Industrial System Complex - Long Term" (EPA's long term aerosol dispersion model)
K_d	complex conversions factor from *mpcf* exposure to f/µg deposition factor of aerosols breathing space
K_e	annual clearance rate of deposited particles in the lung
K_L	exposure-effect gradient between the relative risk of lung cancer and cumulative asbestos exposure
L	litre of air
L:	length of a fibre
L:Ø	aspect ratio of a particle: length divided by diameter
LCDC or L.C.D.C.	Laboratory for Control of Diseases classification (of diseases)
LCL	lower confidence limit (lower boundary of the CI)
ln	natural or neperian logarithm
LOD	limit of detection
MMMF	man-made mineral fibres
mpcf	million particles per cubic foot (total dusts)
mpcf.y	million particles per cubic foot - years (total dusts)
mg/m^3	milligrams per cubic metre of air (usually total dusts <100 µm diameter)
meso.	mesothelioma
MIT or M.I.T.	Massachusetts Institute of Technology
N	number of cases, respondents or subjects, sample size
N-Y	respondent-years (respondents times the number of years in the study base)
NAS or N.A.S.	National Academy of Sciences
neighb.	neighbourhood (exposure to asbestos)
ng/m^3	nanograms per cubic meter of air (usually asbestos fibres)
NIOSH	National Institute for Occupational Safety and Health
NRC	National Research Council
occup.	occupational (exposure to asbestos)
OPDQ or O.P.D.Q.	Office de planification et de développement du Québec
OR	odds ratio
ORCA	Ontario Royal Commission on Asbestos, it self short for: "Royal Commission on Matters of Health and Safety Arising from the Use of Asbestos in Ontario"
p.expos.	exposed population
p.refer.	referent population
PCM or pcm	phase-contrast optical microscopy (light microscopy)
PCME or pcme	PCM or PCOM equivalent
PCOM or pcom	phase contrast optical microscopy (light microscopy)
P_{exp}	proportion exposed (%) among cases
PMR	proportional mortality ratio
pö	tremolite-contaminated whitewash used inside houses in some villages of New Caledonia
pop.	population
P_{ty}	kilotons of asbestos produced in a year and town
PY or p-y	person-years
QAMA	Quebec Asbestos Mining Association

QHS	Quebec Health Survey ("Enquête Santé-Québec")
RR	risk ratio, or relative risk
r_{fv}	"reject fraction", ratio of useless dust aerosols (kg) generated by the production of 1 ton of commercial asbestos fibre; although this ratio varies between plants, it can be averaged over a mining town; it probably did not change significantly over time
s/L	structures per litre
Sc.#	scenario number
SEM	scanning electron microscopy
sublinear	sigmoid-shaped exposure-effect or dose-response curve
TEM	transmission electron microscopy
SMR	standardized mortality ratio (indirect standardization)
SNA	Société nationale de l'amiante
SPCMR	standardized proportional mortality ratio relative to cancer deaths (indirect standardization)
SPMR	standardized proportional mortality ratio (indirect standardization)
SRR	standardized rate ratio (direct standardization)
T.M.	Thetford Mines (either municipality or agglomeration)
t.w.a.	time-weighted average
trem	tremolite
UCL	upper confidence limit (upper boundary of the CI)

PART A. INTRODUCTION

A.1. BACKGROUND AND RATIONALE

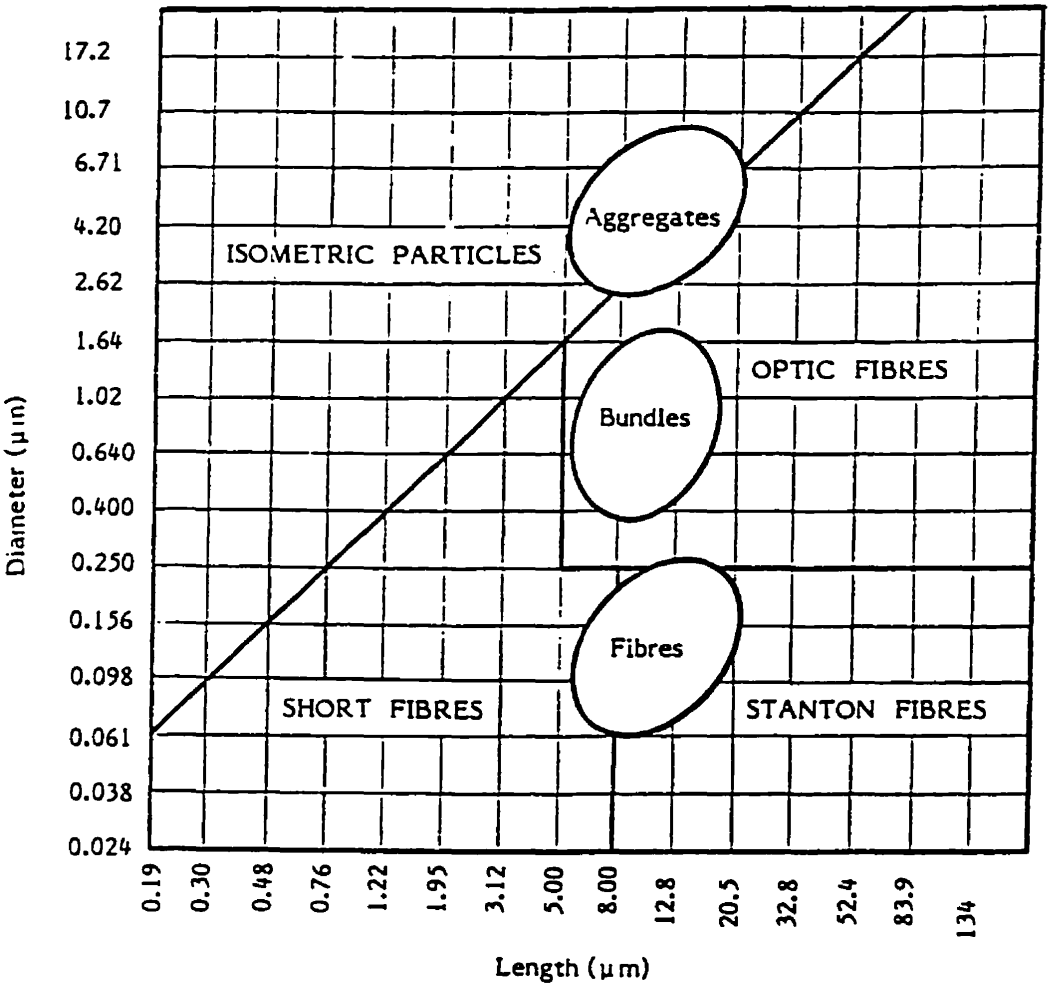
A.1.1. Preliminary Notions and Concepts

Asbestos is the "commercial and generally used name for fibrous varieties of naturally occurring silicate minerals of the amphibole or serpentine groups" [Skinner, 1988, p.192]. Asbestos usually occurs as veins in rocks. The important characteristics of asbestos as compared to non-asbestiform varieties of silicates is the presence of long, thin fibres that can be separated easily. Its fibrous nature, tensile strength, durability, flexibility, and resistance to heat, wear and corrosion have made asbestos one of the most useful and versatile of minerals. Six varieties of asbestos are of commercial importance. Chrysotile asbestos, which accounts approximately for 95% of the asbestos produced in the world, belongs to the serpentine group; it has curly and relatively ductile fibers. Crocidolite and amosite, the two other most commonly used fibres belong to the amphibole group, together with anthophyllite, tremolite and actinolite; amphibole fibers are needlelike, they are more resistant to heat and corrosion than chrysotile but they are less ductile. Through crushing, fiberization, spinning, weaving and other industrial processing, asbestos fibres break longitudinally and become thinner and thus more hazardous. Asbestos products include brake and clutch linings, water pipe, roofing materials, fireproofing, electrical insulation, various other building materials, floor tiles, and chemical filters.

Canada has been one of the leading asbestos-producing countries; it supplied more than half of the world's asbestos production from 1876 to 1957, and it remained the world's largest producer until 1974. In particular, the Eastern Townships in Southern Quebec, the area which is the object of the epidemiological study of the present thesis, has always produced and exported most of Canada's asbestos. Quebec's Eastern Townships produced mainly untransformed asbestos fibres for exportation.

Health risks due to occupational exposure to asbestos have been well documented. Asbestosis, lung cancer and mesothelioma are established health effects of occupational asbestos exposure, and there is some unconvincing evidence that lymphomas, laryngeal and gastro-intestinal cancers and a few other cancer sites may also be associated with occupational asbestos exposure [Doll and Peto, 1987]. Smoking has been shown to be strongly synergistic with asbestos exposure for lung cancer [Selikoff et al., 1968] but not for mesothelioma [Selikoff, 1979]. It is believed that asbestosis probably occurs only at high cumulative occupational exposures, and a threshold of 25 fibre/mL-years has been suggested [Royal Commission on Matters of Health and Safety Arising from the Use of Asbestos in Ontario et al., 1984; Doll and Peto, 1985]; however, an exposure-effect relation has been consistently observed in occupational cohorts [Becklake, 1983]. Still, there is yet no suggestion that asbestosis could be induced by low environmental exposures in the general population, a notion corroborated by the rapid drop in asbestosis morbidity and mortality in the asbestos industry since the reduction of asbestos exposure levels [Becklake, 1991]. There has been and there still is strong controversy as to whether respiratory and mesothelial cancers can be induced by very low cumulative environmental asbestos exposures [Abelson, 1990a; Mossman et al., 1990; Nicholson et al., 1990; Sterling et al., 1993; Upton and Shaikh, 1995]. Some researchers believe that asbestos-related lung cancers can only develop if there is asbestosis or fibrotic lung tissue [Hughes and Weill, 1991; Jones, 1992; Weiss, 1994], but there is evidence of the contrary [Becklake, 1991]. Others believe that there is epidemiological evidence of a threshold for both lung cancer and mesothelioma among chrysotile miners and millers [Liddell, 1993; Liddell, 1994]. Still others [Abraham, 1994; Nurminen and Tossavainen, 1994; Roggli et al., 1994; Sterling et al., 1993] oppose these threshold theories and believe in a linear or possibly supralinear exposure-effect relationship. However, most experts seem agnostic with respect to this issue and would side with cancer risk assessors who use a non-threshold linear model not by conviction but by public health cautiousness.

Figure A-1. Size Distribution of Fibres, Bundles and Aggregates[1]



(The elipsoids contain about 95 % of total particles measured on all the membrane filters of the mining town region.)

[1] Reproduced from[Sébastien et al., 1986] with author's permission.

Long thin fibres (L > 8 µm, Ø < 0.2 µm) are believed to be the most carcinogenic fraction of asbestos aerosols when they reach the target organ [Pott and Friedrichs, 1972; Stanton and Wrench, 1972; Pott, 1978]. However, shorter fibers reach mesothelial tissues more easily [Sébastien et al., 1980b]. The physical dimension classes of asbestos fibres are represented schematically on the graph in Figure A-1 [Sébastien et al., 1986]. By contrast with these toxicity characteristics of asbestos fibres, industrial hygiene measurements of occupational asbestos exposures have been limited up to the 1960s to counts of total dust particulates and, after 1970, to counts of respirable fibres visible by optical microscopy[1].

Most experts believe that crocidolite fibre is more toxic than chrysotile, particularly with respect to mesothelioma. However, the extent of a differential toxicity by mineralogical type, the degree to which other amphiboles such as amosite would be more toxic than chrysotile, remains a hotly debated issue among experts [HEI-AR et al., 1991, p.6-23]. The toxicity of short asbestos fibres (L<5µm) and the carcinogenic mechanism of asbestos are still unknown.

A.1.2. Social, Public Health and Scientific Issues

In the wake of the rising environmental consciousness in the 1960s and with the improvement of working conditions in the asbestos industry in the 1970s, it has become recognized that asbestos exposure among city dwellers is widespread [Chatfield, 1979; Nicholson et al., 1980; Nielsen, 1986; Case et al., 1988; Nicholson, 1989], shifting the asbestos controversy from the workplace to the general environment in the 1980s [Becklake, 1979]. For instance, asbestos-related litigation is no longer limited to occupational exposures, and in the U.S.A. alone tens of

[1] L:Ø aspect ratio > 3:1; L > 5 µm; Ø > 0.25 µm, look like fibres but may be asbestos or not.

thousands of claims have been filed by household contacts of asbestos workers and by members of the general public [Feder, 1981; Stone, 1991; Nicholson and Landrigan, 1994]. Fears engendered by asbestos have led to panics [Girard, 1989; Pitt, 1989b; Pitt, 1989a; Perr, 1994] and to colossal expenditures to eliminate asbestos from public places [Anonymous, 1981; Anonymous, 1989].

Over the last decade, asbestos has been the object of a proliferation of reviews, risk assessments and governmental regulations in many countries. On the basis of risk projections from cohorts of asbestos workers to the general environmentally exposed population, environmental regulations on asbestos have become more stringent throughout the industrialized world. Based on its own risk assessment in 1986 [Nicholson, 1986], the U.S. Environmental Protection Agency (EPA) promulgated regulations to phase out the importation of asbestos and to progressively replace extant asbestos products with man-made mineral fibre products. Although some scientists subscribed to the EPA's view [Selikoff, 1989; Brody, 1990; Nicholson et al., 1990; Landrigan and Kazemi, 1991; Selikoff, 1991], others endorsed the conclusion of an international symposium on asbestos in buildings in 1989 that mesothelioma and lung-cancer risk projections from exposures to indoor asbestos for school-age children and the general population were truly "quite small", i.e. two orders of magnitude lower than those posed by second-hand cigarette smoke or radon [Anonymous, 1989; Harvard Conference on Asbestos and Asbestos Institute, 1989]. In the winter of 1990, the latter viewpoint was endorsed by an editorial in *Science* [Abelson, 1990a; Mossman et al., 1990]. Later that year, the U.S. Court Of Appeal, Fifth District (New Orleans) accepted the arguments of the EPA's critics and nullified the proposed ban on asbestos. Finally, a recent review of risk assessments and environmental exposure data on asbestos was issued by the Health Effects Institute - Asbestos Research panel [HEI-AR, 1991]; assuming the same exposure-effect gradient as was proposed by the U.S. EPA in 1986, but relying on a more complete environmental exposure assessment, the HEI-AR predicted lifetime risks of asbestos-induced cancer deaths of 10 to 140 per million in the general population. Whether this is a "small risk" is a social value issue. Over and above the subjective interpretation of what is a "reasonable",

"acceptable", "small" or "insignificant" risk, the scientific controversy itself remains undiminished and opposing views continue to be voiced for and against the positions stated in Science and in the HEI-AR report[Abelson, 1990b; Abelson, 1990a; Anonymous, 1990; Sterling et al., 1994; Upton and Shaikh, 1995].

Given the overwhelming experimental and occupational evidence about the carcinogenicity of asbestos, why not simply adopt a "zero tolerance" or "no risk" policy? Among the costs of an overly aggressive asbestos-elimination policy are the following: important job losses in the asbestos industry, dramatic drop in asbestos exports for countries such as Canada and Russia, the diversion of limited educational and public health funds to remove asbestos from schools ($150 billion estimated in the U.S.A.), additional asbestos exposure that may be generated by asbestos removal, the non-zero and unknown disease potential of asbestos substitutes [Abelson, 1990a; Mossman et al., 1990], and the greater cost and health risks possibly associated with substitutes to asbestos in brake linings and, for developing countries, in asbestos-cement water-ducts. From a public health viewpoint, it is not obvious therefore whether a zero tolerance policy would be the best or even the safest policy.

How then should cancer risks associated with asbestos use be estimated and interpreted in making public health decisions? In the absence of direct epidemiological evidence concerning the quantitative relation between asbestos-related cancers and non-occupational exposure to asbestos, there has been no choice but to rely on environmental cancer risk assessments synthesizing all existing knowledge on the toxicity and the environmental exposure to asbestos and making quantitative risk estimates for the general population. All these assessments use linear excess risk models justified from multistage carcinogenesis theory. The models are fitted on up to 14 occupational asbestos cohorts, and lung and mesothelial cancer risks are extrapolated from these past high-exposure occupational settings (10-500 f/mL) to the present general low-exposure environment (0.00001-0.005 f/mL)[1].

[1] The "f/mL" unit of airborne asbestos concentration is the equivalent of the past "f/cc" unit of measurement and has been used mostly in the asbestos industry. This unit has been traditionally associated with phase-contrast optical microscopy counting (PCM). By contrast, "f/L" has been the unit used to measure concentrations in the general environment with transmission electronic microscopy (TEM). In the dissertation, I have followed

Such risk projections are inevitably laden with error. First, the very carcinogenesis "dose-response" model which justifies linear "exposure-effect" extrapolations on the basis of a low-dose approximation to the Doll-Armitage multistage model is uncertain and may not be generalizable from cellular-level doses and events to whole-body exposures and population-level risks. Second, the occupational data used to fit the exposure-effect model are neither precise nor consistent. The estimated exposure-effect gradients differ 670-fold between cohorts. The 95% confidence intervals associated with each study's lung cancer SMRs are one or two orders of magnitude wide. Each study's exposure assessment is itself laden with error, often being based on poor proxy asbestos exposure measurements. Confounding, the healthy worker effect, and other comparability issues bias most of these studies which have used external reference populations. Finally, there are important dissimilarities between historical occupational cohorts studied and targeted general populations in terms of age, sex, overall health status, exposure-time patterns, etc. As a result of these uncertainties, the plausible range around the environmental low-exposure risk projections to the general population is orders of magnitude wide. Thus, the scientific basis for any public health policy on asbestos is meager and very controversial, as asbestos risk assessments are both necessary and problematic.

A.1.3. The Need to Study Cancer Risks Associated With Low to Intermediate Non-Occupational Exposures

What avenues are open to reduce the uncertainty of environmental risk assessments on asbestos, given that chrysotile is the main asbestos material to which the general population is exposed today and that it is very possible that chrysotile and amphibole fibres may have partially different toxicities?

the convention used by the HEI-AR [1991] expressing all measurements in "f/mL" where possible. I have used qualifiers such as "optical" and "electronic" when relevant.

Animal and occupational studies are not likely to shed new light on low-exposure risks. Animal experiments require huge exposure levels or unaffordable sample sizes to observe statistically significant cancer risks and are thus more remote from the exposure levels in the general population than occupational cohorts are, even notwithstanding the interspecies extrapolation problem. Regarding occupational studies, pooling data from the lowest exposure groups of historical occupational cohorts could give insights in the effects of intermediate asbestos exposures, but pooling would be hampered by exposure measurement insufficiencies and discrepancies between cohorts; even if those problems were overcome, the generalizability of the results from male workers to the general populations would still be limited. An original contribution of occupational studies to environmental risk assessment on asbestos could come from the follow-up of today's asbestos workers and of workers of secondary asbestos transformation and utilization industries; but statistically useful observations could not be expected before at least two decades. Occupational lung burden studies might help to evaluate the comparability of the exposure assessments between different occupational cohorts, or between asbestos workers and the general population (in terms of exposure levels, but more specifically in terms of fibre physical and chemical characteristics), but they would not reduce the uncertainty due to the down-scaling magnitude of extrapolations and due to the dissimilarity between workers and the general population.

From the viewpoint of environmental risk assessments on asbestos and cancer, the most efficient way to bridge the gap between extrapolations from occupational studies and the very low but continuous lifelong asbestos exposures of the general population is to directly estimate the exposure-effect gradient in populations close to the target general population in terms of exposure intensity, exposure time patterns and background risk factors and host characteristics. Likewise, after reviewing the asbestos literature and risk assessments to estimate the risks of occupants of public and commercial buildings, the HEI-AR's first recommendation for future research into the health effects of low exposures to asbestos fibres was the following:

Epidemiological studies of carefully selected populations should be carried out to explore further the long-term effects of low to intermediate levels of exposure ... (HEI-AR, 1991, p.6-79)

With respect to fibre species, Langer and Nolan[Langer and Nolan, 1989] concluded after analyzing 54 occupational lung burden samples:

Although 95% of the fibre used in the US over the past 5 decades has been chrysotile, the 2 other commercially important fibres, amosite and crocidolite, were commonly found in the lungs of workers studied. (...) In the cases studied, tremolite tends to occur with chrysotile exposure and anthophyllite and actinolite with amosite exposure. (...)

The assessment of risk to asbestos disease in the general population of the US, exposed to chrysotile, should be based on appropriate chrysotile-exposed cohorts.

The present thesis addresses these research needs by studying the asbestos exposure and mortality experience of a population of asbestos-mining town residents with long-standing continuous exposure to intermediate levels of airborne chrysotile, and by directly estimating the exposure-effect gradient in this population.

A.1.4. The Present Study of Quebec Non-Occupationally Exposed Asbestos-Mining Town Residents

In the asbestos-mining region in Quebec's Eastern Townships, there is a unique opportunity to measure the impact of non-occupational exposure to chrysotile asbestos. This area was the world's largest asbestos mining and exporting region until 1974. Before the installation of modern emission control technology was completed on all asbestos mills and dryers at the end of the 1970s, there was apparently a fairly high level of asbestos exposure in the general environment of this area. Anecdotes tell of visible 24-hour asbestos dust depositions on the ground, floors and cars as a commonplace feature of life there. Quebec's asbestos-mining towns have likely been much more exposed to asbestos than other general populations, yet less than past cohorts of asbestos workers. Moreover, the type of asbestos mined in the area is chrysotile, a fibre that makes up more than 95% of all asbestos products in the world. This setting thus provides an opportunity to investigate the risks of exposure to intermediate non-occupational chrysotile levels. Moreover, available indirect exposure data enabled a retrospective exposure assessment which would allow to characterize the exposure-effect relationship in intermediate non-occupational chrysotile exposure circumstances.

To be congruous with the target population of environmental risk assessments, the effect of non-occupational exposure should not be mixed with the effect of occupational exposure. However, 70% of the male population of Quebec's asbestos-mining towns worked in the industry some 15-45 years. Among women on the other hand, less than 10% worked for the asbestos industry and those who did were in less dusty jobs for short employment periods. Consequently, the study was restricted to females to focus on a population non-occupationally exposed to asbestos almost continuously and around-the-clock since childhood.

An ecological study was designed to determine whether the female population of the Quebec asbestos-mining agglomerations experienced excess lung cancer mortality and, if so, to relate the excess risk to the population's cumulative asbestos exposure. Lung cancer mortality is the only health outcome other than mesothelioma for which the exposure-effect relationship has been quantified by environmental risk assessments on asbestos. Given that lung cancer deaths can be readily identified from the Quebec Mortality Registry far back in time and that most cases die within five years of diagnosis, lung cancer mortality was used as the proxy for lung cancer incidence. The study was commissioned by Health and Welfare Canada.

A few studies [Thériault and Grand-Bois, 1978; McDonald and McDonald, 1980; Pampalon et al., 1982] have examined the mortality of residents of this area, but they had low statistical power, they did not characterize the asbestos exposure of the study population, and one focused on the health effects of asbestos in drinking water [Wigle, 1977]. In the present study, mortality was ascertained over the 1950-1989 period, encompassing 3-5 times more person-years than previous studies, and an intensive effort was made to retrospectively estimate the cumulative exposures of the female residents of these areas. In fact, the exposure assessment constituted the most original contribution and the main challenge and effort of this project.

Mesothelioma incidence was not part of the thesis research. Due to its diagnostic complexity and uncertainty, neither mortality registries nor usual tumor registries can be trusted for the identification of cases. Mesotheliomas must be determined by thorough ascertainment and pathological review procedures, an endeavour that has been undertaken by a team including Drs. J. Siemiatycki, B. Case and myself in an ongoing study to be completed in 1997.

A.2. OBJECTIVES

The thesis aimed to achieve the following objectives:

1. Exposure assessment:

   To estimate the average cumulative asbestos exposure experienced by the female population of Quebec's asbestos-mining agglomerations.

2. Risk prediction:

   To compute the relative risk of lung cancer predicted by the EPA model on the basis of the population's estimated cumulative asbestos exposure.

3. Risk observation:

   To determine this exposed population's relative risk of lung cancer mortality in comparison to the female population of other comparable agglomerations in Quebec.

4. Validation of the EPA model:

   To compare lung cancer relative risk predicted by the EPA linear excess relative risk model with the relative risk observed in this non-occupationally exposed population.

A.3. LITERATURE REVIEW

A.3.1. Scope of the Review

The present study was conceived to provide the first validation of quantitative environmental risk assessments on asbestos and lung cancer directly in a non-occupationally exposed population. Accordingly, the literature review has borne on non-occupational epidemiological studies on asbestos and lung cancer and on quantitative risk assessments of environmental asbestos related lung cancers, more specifically on the "exposure-effect assessment" part of these risk assessments. However, given the scarce evidence on lung cancer and non-occupational asbestos exposure, studies on malignant mesothelioma in non-occupational exposure settings have been added to the review, as markers of the potential excess risk of lung cancer attributable to asbestos exposure. Indeed, mesothelioma is the main detectable cancer risk possibly associated to low and intermediate asbestos exposures. The non-occupational studies are not reviewed in detail but rather globally because most studies are of limited validity and none has quantified asbestos exposure, thus none can contribute to the assessment of the exposure-effect relationship. Occupational and experimental studies on asbestos were excluded.

The present review discusses and updates the results of two recent comprehensive reviews on respiratory and mesothelial cancer risks associated with environmental asbestos: 1) a 1989 review paper by Gardner and Saracci on the *Effects on health of non-occupational exposure to airborne mineral fibres* [Gardner and Saracci, 1989], and 2) the 1991 Health Effects Institute-Asbestos Research review of environmental risk assessments entitled *Asbestos in Public and Commercial Buildings: A Literature Review and Synthesis of Current Knowledge* [HEI-AR, 1991] The review included in the *Institut national de la Santé et de la recherche médicale* 's risk assessment in 1996 [INSERM et al., 1996] was used to complete the present review.

Preliminary Definitions: Sources of Non-Occupational Asbestos Exposures

Some of the terminology used regarding different sources of asbestos exposure can be rather confusing. In this review and in the rest of the thesis, I have adopted the following conventions.

Neighbourhood exposure was defined as having breathed air contaminated by nearby asbestos man-made emission sources such as asbestos natural outcrops, mines and mills, asbestos-product factories, and shipyards which used asbestos paints and insulation in shipbuilding. It also comprised the exposure of populations having used asbestos-containing whitewash and stucco to whiten or build their houses. Neighbourhood exposures has been experienced both outside and inside homes since indoor air would inevitably be contaminated by indoor-outdoor air exchanges. I avoided the expression "residential exposure" used by some authors to designate neighbourhood exposure; it does not characterize the proximity of residence with respect to pollution sources and it can be confused with "household exposure".

Household-contact exposure - or *household exposure* - was defined as the excess indoor asbestos exposure experienced by persons living in the household of an active asbestos worker. Persons with household exposure have been termed *household contacts.* The following expressions may have been used by various authors in regard to household exposure but have been avoided in this text: "domestic", "cohabitation", "housemate", "bystander", "worker-mediated" and "para-occupational" [Gardner and Saracci, 1989] exposure.

Some persons have experienced both neighbourhood and household exposures and this must be considered when evaluating the respective risks of each type of exposure. As well, some study subjects may have been occupationally exposed to asbestos.

Both neighbourhood pollution and household-contact pollution may also entail extra indoor exposure for members of asbestos-polluted households who perform housework. This *housework asbestos exposure* would be due to a closer, more aggressive and more frequent

contact with asbestos laden objects such as floors, carpets, curtains and clothes. This exposure would occur indoor and would apply specifically to mature female exposed populations; they worked most of their lifetime as housewives and probably did a lot of housework even as maidens. Housework might induce a sex differential in respiratory cancer risks related to asbestos exposure.

Table A-1 depicts these types of exposures and is used as a conceptual framework in the review and throughout the dissertation.

Table A-1 Components and Sources of Asbestos Exposures Among Females[1] by Asbestos Exposure Status

Neighbourhood exposure status:	Unexposed			Exposed		
Household exposure status:	Unexposed	Exposed	Exposed	Unexposed	Exposed	Exposed
Occupational exposure status:	Unexposed	Unexposed	Exposed	Unexposed	Unexposed	Exposed
Place in the day[2]						
Outdoor (≈3 hrs.)	E_{oo}	E_{oo}	E_{oo}	E_{no}	E_{no}	E_{no}
Home (≈14 hrs.)	E_{oi}	$E_{oi} + E_{hi}$	$E_{oi} + E_{hi}$	E_{ni}	$E_{ni} + E_{hi}$	$E_{ni} + E_{hi}$
Work (≈7 hrs.)	E_{oi}	E_{oi}	$E_{oi} + E_{wi}$	E_{ni}	E_{ni}	$E_{ni} + E_{wi}$
Personal exposure[3] on an average day	PE_o	PE_h	PE_w	PE_n	PE_{nh}	PE_{nw}

- First subscript represents type of exposure.
- Second subscripts represents whether inside or outside when referring to exposure levels "E".
- Second subscripts represents an additional household or an additional work-related exposure when referring to personal exposure status "PE" of neighbourhood-exposed persons.

E_{oo} Average background outdoor exposure level for the general population.

E_{oi} Average background indoor exposure level for the general population in the absence of indoor sources of asbestos. Slightly lower than outside due to building or wall filtration effect.

E_{no} Average neighbourhood outdoor exposure level for a population residing near an asbestos emission source. It comprises the background level E_{oo} plus the added pollution by local sources of emissions. To simplify the table, these two elements (general background and local emissions) were merged in to E_{no}.

E_{ni} Average neighbourhood outdoor exposure level in the absence of indoor sources of asbestos for a population residing near an asbestos emission source. Slightly lower than outside due to building or wall filtration effect.

E_{hi} Household-contact exposure added to indoor background by the presence of an asbestos worker in the household. Occurs inside.

E_{wi} Occupational exposure added to indoor background by asbestos-related work. This may occur directly in the home. For instance, some women repaired jute bags at home for the asbestos industry.

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[1] The table and its elements refer to female exposures, which are different from male asbestos exposures. In particular, housework entails a higher exposure to dusts and asbestos due to re-aerosolization, yet it is often and it used to be exclusively the lot of females. Also, asbestos exposures at work would be different by sex.

[2] Hours per day averaged over a typical 7-day week. These are simply indicative rough estimates.

[3] This is a time-weighted average of the different asbestos exposures experienced over an average day.

A.3.2. Epidemiological Studies on Respiratory Cancer Risks Associated With Non-Occupational Exposures to Asbestos

Tables A-2, A-3, A-4, A-5 and A-6 summarize the results of 32 non-occupational studies and reports on the risks of lung cancer and mesothelioma in populations with neighbourhood exposure to airborne asbestos or with household-contact asbestos exposure[1,2].

The evidence of an association between lung cancer and non-occupational exposure is consistent with a positive effect of "non-occupational exposure to asbestos fibres". The pooled estimate for the five studies was 1.20 with an approximate 90%CI of 1.02 - 1.41. The New Jersey amosite study was the only study where the asbestos exposure was strictly of the neighbourhood type; it had also the only risk ratio smaller than 1.0. All four other studies involved significant household exposures and had RRs larger than 1.0. The pooled RR for these studies was 1.52 with a 90%CI of 1.24 - 1.86. The relative risk was higher in the populations which were apparently more exposed to amphibole fibres, particularly crocidolite. If the study on crocidolite-mining areas were excluded[3], the pooled estimate for the four other studies would not be statistically significant (RR = 1.09 with a 90%CI of 0.91 - 1.30). Overall, accounting for a 25-year latency, having lived with an asbestos worker before 1955 has been associated with an increased risk of lung cancer, apparently correlated with amphibole - particularly crocidolite - fibres. However, the evidence of an excess risk of lung cancer induced by chrysotile exposure is inconclusive, and there is yet no direct evidence that an excess risk of lung cancer may result from strictly neighbourhood exposure.

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1 In the tables, the symbol "≈" means that in the absence of a mesothelioma reference rate, I estimated a ratio and confidence interval assuming a background yearly rate of 2×10^{-6} or a background lifetime risk of 2×10^{-4}.

2 Some epidemiological studies were excluded because they overlapped with and were superseded in validity by the above studies: 4 studies in Quebec [Graham and et al., 1977; Wigle, 1977; Loslier, 1983; Toft et al., 1984], and 1 study in New Jersey [Joubert et al., 1991].

3 Excluding studies on crocidolite exposure is a valid alternative since there is hardly any crocidolite in the environmental exposures of the general population in North America today.

Table A-2 Epidemiological Studies on Non-Occupational Asbestos Exposures and Risk of Lung Cancer

Study[1] [author, year of publication]; country; follow-up; population; type of fibre; note	Relative risk [2]	90% CI	Observed cases	Exposed population	Exposure circumstance
[Magnani et al., 1993]; Monferrato, Italy; 1965-78; wives of asbestos-cement workers; chrysotile + 10% crocidolite; all household exposures	SRR = 1,50	0.7 - 3.3	6	1,964 wives	Household
[Botha et al., 1986]; South Africa; 1968-80; mining districts vs. neighbouring districts; crocidolite; neighbourhood, but must include occupational and household exposures;	SMR = 1.87	0.98 - 3.3	9	White female residents	Neighbourhood and some household
	SMR = 2.47	1.4 - 4.0	12	Other females residents	
	SMR = 2.17	1.4 - 3.1	21	Total: all females	
[Anderson, 1982]; New Jersey, USA; 1961-1980; families of factory workers; amosite; household exposure, may include some occupational exposure, unknown smoking	SMR = 1,85	1.2 - 2.8	20	2,218 household contacts	Household
[Pampalon et al., 1982; Siemiatycki, 1982]; Quebec, Can.; 1966-1977; female residents of mining towns; chrysotile; 70% with household exposure	SRR = 1.07	0.73 - 1.5	23	22,000 female residents	Neighbourhood and household
[Hammond et al., 1979]; New Jersey, USA; 1968-76; male residents less than 0.8 km from factory; amosite; only neighbourhood exposure, no smoking data	RR = 0.89	0.7 - 1.4	41	1,779 exposed 3,771 unexposed	Neighbourhood
POOLED ESTIMATE:	**RR = 1.20**	**1.0 - 1.4**	**111**		

[1] The studies are listed in inverse chronological order.

[2] SRR = standardized rate ratio; SMR = standardized mortality ratio; PMR = proportional mortality ratio; RR = relative risk; OR = odds ratio.

Table A-3 Epidemiological Studies on Household-Contact Exposure to Asbestos and Risk of Mesothelioma

Study[1] [author, year of publication]; country; follow-up; population; type of fibre; note	Relative risk [2]	Exposed cases	Population	Exposure circumstance
[Magnani et al., 1993]; Casale, Monferrato, Italy; 1965-88; wives of asbestos-cement workers; + 10% crocid.; -----	SRR = 7.9	4	1,964 wives	Household
[Anderson, 1982]; New Jersey; 197?-8?; families of amosite factory workers; amosite; possible occupational exposure	"PMR" ≈ 30	3	663 deaths among 2,218 house. contacts	Household
[Vianna et al., 1981]; New York; 1973-78; wives of asbestos factory workers; asbestos; -----	OR = 8.0	7	14 non-occ. cases	Household
[McDonald et al., 1980]; Canada and California; 1972-7?; having lived with a chrysotile miner or miller; chrysotile; -----	OR = 4.0	8	557 non-occ. cases 557 non-occ. controls	Household
[Vianna et al., 1978]; New York; 1973-78; females living with an asbestos factory worker; asbestos; -----	OR = 8.0	8	46 non-occ. cases 46 non-occ. controls	Household
[Newhouse et al., 1965]; London, U.K.; 1915-65; living with an asbestos factory worker; crocidolite; -----	OR = 9.0	9	45 non-occ. cases 45 non-occ. controls	Household

[1] The studies are listed in inverse chronological order.

[2] SRR = standardized rate ratio; SMR = standardized mortality ratio; PMR = proportional mortality ratio; RR = relative risk; OR = odds ratio.

Table A-4 Case Series on Household-Contact Exposure to Asbestos and Risk of Mesothelioma

Study[1] [author, year of publication]; country; follow-up; population; type of fibre; note	Proportion exposed non-occupationally	Exposed cases	Population	Exposure circumstance
[Bianchi et al., 1987]; Monfalcone, Italy; 1979-87; living in shipbuilding area; asbestos; -----	80 %	4	5 non-occup.	Household
[Bianchi et al., 1982; Giarelli et al., 1992]; Trieste, north-east Italy; 1968-87; living in shipbuilding area; asbestos; -----	25%	5	20 non-occup.	Household
[Milne, 1972]; Victoria, Australia; 1962-72; having lived with an asbestos worker; crocidolite; -----	17 %	1	6 non-occup. meso.	Household
[Wagner et al., 1960]; Cape Prov., S. Africa; living in asbestos-mining area; crocidolite; includes household exposures	18 % 91 %	4 m. 10 f.	22 males 11 females	Household + Neighbourhood

[1] The studies are listed in inverse chronological order.

Table A-5 Epidemiological Studies on Neighbourhood Asbestos Exposures and Risk of Mesothelioma

Study[1] country; follow-up; exposure; type of fibre; note	Relative risk [2]	Exposed cases	Population	Exposure circumstance
[Magnani et al., 1995]; Monferrato, Italy; 1980-89; living ≤ 1 km from asbestos-cement plant; chryso.+10% crocidolite; strong design	SRR = 10	36	42,000	Neighbourhood
[Goldberg et al., 1991]; New Caledonia; 1978-87; Melanesian houses, huts; tremolite whitewash (pö); strong evidence, exposure levels	SRR = 9; RR > 30 in huts	12	Melanasian pop.; 7000 tremolite huts	Neighbourhood + Indoors
[Langer et al., 1987]; north-west Greece, Metsovo village; 1980-8?; whitewash inside houses; tremolite; -----	PMR > 50	6	600 deaths from all causes	Neighbourhood + Indoors
[Botha et al., 1986]; South Africa; 1968-80; mining vs. neighbouring districts; crocidolite; possible occupational exposures	SMR = 9.4	30	female residents	Neighbourhood
[Teta et al., 1983]; Connecticut; 1955-77; residence in "labor market area" near chrysotile products plant; chrysotile; -----	OR = 0.95	201	?	Neighbourhood
[McDonald et al., 1980]; Canada and U.S.A.; 1960-75; living < 20 km from chrysotile mines; chrysotile; -----	OR = 0.25	1	557 non-occ. cases 557 non-occ. refer.	Neighbourhood
[Yazicioglu et al., 1978; Yazicioglu et al., 1980]; south-east Turkey; 1977-78; whitewash inside houses; tremolite+chrysotile; -----	SRR ≈ 25	11	227,420 p.expos. 217,960 p.refer.	Neighbourhood + Indoors
[Hammond et al., 1979]; New Jersey; 1962-76; males living < 0.8 km from amosite factory between 1942 and 1954; amosite; -----	PMR ≈ 8	1	780 deaths expos. 1,735 deaths refer.	Neighbourhood
[Vianna et al., 1978]; New York; 197?-77; females living < 8 km from asbestos factory; asbestos; -----	Indefinite	1	46 non-occ. cases 46 non-occ. refer.	Neighbourhood
[Thériault et al., 1978]; Quebec; 1969-72; distance from chrysotile mines; ----; ----	Positive in males Negative in females		Concentric district areas vs. mines	Neighbourhood + Household?
[Newhouse et al., 1965]; London, U.K.; 1915-65; living < 0.8 km from asbestos factory; crocidolite + amosite + chrysotile; -----	OR = 2.2	11	45 non-occ. cases 45 non-occ. refer.	Neighbourhood

[1] The studies are listed in inverse chronological order.

[2] SRR = standardized rate ratio; SMR = standardized mortality ratio; PMR = proportional mortality ratio; RR = relative risk; OR = odds ratio.

Table A-6 Case Series on Neighbourhood Asbestos Exposures and Risk of Mesothelioma

Study[1] [author, year of publication]; country; follow-up; population; type of fibre; note	Proportion exposed non-occupationally	Exposed cases	Population or number of mesotheliomas	Exposure circumstance
[Boutin et al., 1989; Viallat et al., 1991]; north-east Corsica; 1978-89; living in tremolite-contaminated villages; tremolite+chrysotile; -----	---%	8	Corsica = 250,000 undefined # of non-occup. meso.	Neighbourhood
[McConnochie et al., 1987; McConnochie et al., 1989]; south Cyprus; 1977-86; living in tremolite-actinolite-contaminated villages + stucco; tremolite; ------	63%	5	Cyprus villages; 8 non-occup. meso.	Neighbourhood + Indoors
[Armstrong et al., 1984]; Western Australia; 1960-86; Wittenoom crocidolite mining area; crocidolite; -----	19 %	7	37 non-occ. meso.	Neighbourhood
[Baris et al., 1981; Baris et al., 1988]; central Turkey; 1979-83; living in tremolite-contaminated villages + stucco; tremolite (+ chrysotile); -----	100 %	117	117 non-occ. meso.	Neighbourhood + Indoors
[Hain et al., 1974]; Hamburg, Germany; 1960?-73; having lived ≥5 years <1 km from asbestos factory; crocidolite;---	31 %	20	65 non-occup. meso.	Neighbourhood
[Webster, 1973]; South Africa; 1956-70; crocidolite mining areas; crocidolite; some overlap with Wagner et al.[1960]	58 %	76	130 non-occ. meso.	Neighbourhood
[Lieben et al., 1967]; Pennsylvania; 196?-66; living in immediate neighbourhood of asbestos plant; asbestos; -----	25 %	8	32 non-occ. meso.	Neighbourhood
	9 %	3		Household

[1] The studies are listed in inverse chronological order.

Mesothelioma incidence is much more strongly associated than excess lung cancer with either household exposure, neighbourhood exposure or both exposures, probably due to the high specificity of mesothelioma with respect to asbestos exposure. Despite the underdiagnosis of mesothelioma in general populations, there are mostly positive biases affecting the comparisons of exposed and referent groups. As suggested in previous studies [McDonald, 1979; McDonald and McDonald, 1980], there probably was an overdetection bias of mesotheliomas in exposed areas or populations. By corollary, underdiagnosis may have been lower in exposed areas. Also, the inclusion or misclassification of occupational as non-occupational cases must have been more significant in case-control or case-series studies of mesothelioma than in ecological studies of lung cancer mortality. Indeed, given the difficult diagnosis of malignant mesothelioma, physicians and pathologists usually look for asbestos exposure histories, and such a history is known to influence the final diagnosis. Still, despite such positive biases, the relatively high specificity of mesothelioma with respect to asbestos exposure and its rare background occurrence make the detection of a similar number of asbestos-induced cases much more likely with mesothelioma than with lung cancer. The mesothelioma data must be more valid than the lung cancer data for these reasons alone.

Most but not all studies on mesothelioma indicate a large excess risk attributable to non-occupational asbestos exposure. All studies on household-exposure report an excess risk of mesothelioma whether the fibres involved are amphiboles or chrysotile, but the relative risks are higher with amphiboles and asbestos mixtures ($RR \geq 8$) than with "chrysotile only" (1 study: $RR=4$). Neighbourhood-exposure studies show apparently lower excess risks. All positive neighbourhood exposure studies ($RR \geq 2.2$) involve amphiboles - usually crocidolite or tremolite, but also amosite -, whereas all three negative studies ($RR<1$) involve "chrysotile only" exposures [Teta, 1983; McDonald, 1980; Thériault, 1978]. However, these risk differentials cannot be interpreted straightforwardly without quantitative exposure data.

Overall, the positive association between the risk of mesothelioma and both neighbourhood and household-contact asbestos exposures supports the apparent positive association between

household exposure and excess lung cancer, and suggests that neighbourhood exposure to asbestos may entail a real but as yet unmeasured excess risk of lung cancer. Both lung cancer and mesothelioma were associated more strongly with household exposure than with neighbourhood exposure, suggesting that household contacts would have experienced higher asbestos exposures than neighbourhood-only exposed persons. In terms of differential risks by mineralogical species, non-occupational studies seem to corroborate the apparent risk differentials in occupational studies, the few neighbourhood chrysotile exposure studies showing yet no excess risk of mesothelioma nor of lung cancer. Moreover, no mesothelioma case series in pathology units points to predominantly or exclusively chrysotile exposures.

The interpretation of these differences is ambiguous since they could be attributable to either: a) different toxicities of different fibre types, b) different cumulative doses, c) different exposure circumstances and environments, d) different study designs or background rates, and e) biases. The foremost obstacle to inference about the potency of non-occupational asbestos fibres and about potency differentials is the absence of quantitative exposure data. For instance, it has been implied and sometimes asserted that the high incidence of mesothelioma among household contacts indicated that existing risk assessments seriously underestimate the real cancer risks [Sterling et al., 1993]. Yet the absence of asbestos exposure assessments in these studies precludes such exposure-effect inference and any extrapolation to environmental risks of general populations.

A.3.3. Environmental Risk Assessments on Asbestos and Lung Cancer

In the absence of reliable empirical evidence on the risks due to non-occupational exposure to asbestos, a number of agencies have undertaken to estimate such risks by means of

"environmental quantitative risk assessments". Guidelines for health risk assessment were formalized in 1983 by the U.S. National Research Council [National Research Council, 1983] and augmented in 1986 by the U.S. Environmental Protection Agency [EPA, 1986], prescribing that health risk assessments should follow four steps: "hazard identification", "dose-response assessment", "exposure assessment" and "risk characterization".

On the basis of experimental and epidemiological evidence, the various risk assessment groups have inferred that the only pathway through which asbestos induced a material risk was inhalation and that the only quantifiable risks in the general population were the risks of malignant mesothelioma and lung cancer [NRC et al., 1984]. In all risk assessments to date [Consumer Product Safety Commission (U.S.), 1983; Health and Safety Executive (U.K.) et al., 1983; National Research Council (NRC) et al., 1984; Royal Commission on Matters of Health and Safety Arising from the Use of Asbestos in Ontario et al., 1984; Doll and Peto, 1985; Nicholson, 1986; HEI-AR et al., 1991; INSERM et al., 1996], the risks observed in historical cohorts of highly exposed asbestos workers[1] have been extrapolated more than 10,000-fold downward to the general population's very low environmental exposures today. These extrapolations are based on the one-hit linear dose-response approximation of the multistage carcinogenesis model [Armitage and Doll, 1957; Armitage and Doll, 1961; Armitage, 1982]. The justification for this model is threefold:

1) the linear model is supposedly "conservative" in that it should estimate higher risks in the low-exposure range than the contending threshold and sublinear models;
2) it is the most used mathematical carcinogenesis model, although no model has acquired universal acceptance yet; and
3) "the available occupational data suggest that the cancer risk is roughly proportional to the level of exposure" [HEI-AR, p.6-9] (e.g. Figure A-3 below).

[1] Following the NRC and the EPA guidelines, occupational and other human exposure-effect data are always more relevant to the target human populations than experimental data and should always be preferred to experimental data for that reason.

Using cumulative lifetime exposure as a proxy for dose, the linear exposure-effect model expresses the lung cancer rate in an exposed population as a function of the baseline or background lung cancer rate and of the exposed population's average cumulative asbestos exposure. The relation can be written in relative terms independent of absolute lung cancer rates as follows [HEI-AR report, 1991, p.6-11]:

$$\lambda_k = \lambda_0 \bullet \left(1 + K_L . \overline{cE}\right) \text{ or, equivalently,}$$

$$RR_s = {}^{\lambda_k}\!/_{\lambda_0} = 1 + K_L . \overline{cE}$$

where K_L = toxicity gradient, the increase in lung cancer excess relative risk per unit of "cumulative exposure"

λ_k = standardized lung cancer incidence or mortality rate in the exposed population

λ_0 = standardized background lung cancer incidence or mortality rate in an unexposed but otherwise comparable population with similar smoking habits

$\overline{cE}$ = occupationally equivalent (40hrs./wk) mean cumulative exposure $\left(\frac{\text{f-y}}{\text{mL}}\right)$

and RR_s = standardized rate ratios or relative risks (SMR, SRR, SPMR, OR, RR).

The toxicity gradient K_L represents the linear increase in excess relative risk associated with each unit increment of cumulative exposure[1]; thus a K_L of 0.01 means that a 1 f-y/mL increase of cumulative exposure increases the relative risk by 1%. This gradient is assumed to be independent of age, duration of exposure and time since first exposure, and ought to be the same among smokers and non-smokers[2]. K_L is estimated from selected occupational studies by fitting each study's exposure-stratified rate ratios with the linear model, using unweighted, "empirically" weighted[3] or iteratively reweighted linear regression. To avoid confounding,

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1 Because most occupational asbestos studies have relied on death certificate data, the cumulative exposure has usually been calculated after deducting the last 5-10 years of exposure to allow for the time between lung cancer induction and death.

2 Although Berry et al. [Berry et al., 1985] have estimated from pooling epidemiological studies that the relative risk of lung cancer might be twice as high in non-smokers as in smokers, the authors suggest that this difference could well be due to misclassification of 1-2% of smokers. If true however, due to the implied "less than multiplicative" interaction between asbestos and smoking, the model would tend to underestimate the risk of lung cancer in populations with lower cigarette smoke exposure.

3 A priori empirical precision weights are usually either person-years or the inverse variance of the observed rate ratio or relative risk measure. Empirical weights are usually less precise than "iterative reweighted least squares" (IRLS) which produces maximum likelihood estimates.

ideally, the referent population should represent the baseline incidence of lung cancer in the exposed population, thus it should be similar at least in terms of sex, age and smoking habits. Moreover, the model postulates that the effects of cigarette smoking and asbestos exposure on lung cancer risk are multiplicative[1]. From the resulting cohort-specific K_L estimates, risk assessors estimate a median, a weighted average or a confidence interval of K_L, or the full range of K_L estimates to characterize the risk of non-occupationally exposed populations on the basis of the exposure assessments of these populations.

a) Results of Exposure-Effect Assessments

The U.S. CPSC (1983) produced a 10-fold range of K_L estimates centered on the median (0.01) of 11 occupational cohorts. The U.S. NRC (1984) rounded the median gradient of 9 cohorts upward from 0.011 to 0.02 to obtain a supplementary margin of safety. The U.K. HSC (1985) averaged to 0.01 the gradients estimated from the purported only two occupational studies with reliable exposure estimates; these were two cohorts of chrysotile textile workers. The U.S. EPA (1986) excluded the studies on chrysotile miners and millers as low outliers and computed the inverse variance-weighted geometric mean (0.01) of the gradients estimated for the remaining 11 studies[2]. The HEI-AR review and the INSERM's risk assessment have used the EPA's exposure-effect K_L estimate without reviewing or re-analyzing original studies. Ontario's ORCA was the only group who did not synthesize the heterogeneous gradients estimated from 7 cohorts in a single estimate, opting for risk predictions specific to the type of asbestos fibres and industrial processes involved[3]. The U.K. Health and Safety Commission

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[1] Smoking and asbestos exposures are multiplicative [Hammond et al., 1979b] on an additive scale; but on a multiplicative or ratio scale, they are non-interacting cofactors. Smoking habit differences between study and referent populations are not likely to cause more than a 30% error in ratios [Asp, 1982; Blair et al., 1985; Siemiatycki et al., 1988; Axelson, 1989].

[2] The 11 studies represent 10 cohorts; 2 studies on a group of textile workers in Charleston (U.S.A.) overlapped.

[3] For instance, the ORCA used the K_L estimated from Selikoff's cohort of North-American insulators to estimate the cancer risks of commercial and public building occupants and of building maintenance, custodial and janitorial workers. (Note: this would tend to overestimate the risk of building occupants and users though it would be appropriate for building workers.)

produced a risk assessment in 1979 updated in 1983 [Acheson and Gardner, 1983] which does not appear in the EPA summary nor in the HEI-AR review; this assessment, like the ORCA's, used a discriminating approach, and did not produce a final universal point estimate.

Table A-7 was taken from the HEI-AR report (1991) and was adapted from Nicholson (1986). It shows the toxicity gradients K_L estimated from various studies of asbestos workers in five different risk assessments. For a given cohort, the gradients estimated by various risk assessment groups are relatively similar. The differences between assessments result from various factors: updated data and analyses, use of weighted or unweighted regression, adjustment or not for local rates, forcing or not the model through the zero intercept, and using or not relative slope estimates[1] [Hanley and Liddell, 1985; Liddell and Hanley, 1985]. As to the global point estimates, they also differ by the cohorts available at the time of the assessment and those selected for the pooled estimate of the exposure-effect gradient. For instance, the EPA reviewed 14 cohort-studies in 1986 but excluded the three cohorts of chrysotile miners and millers from its final exposure-effect estimate, whereas the HSE [Peto et al.] used only two cohorts of textile workers for which the exposure measurements were deemed adequate by the assessors. The global exposure-effect gradients estimated by the various risk assessment groups differ little and center around 1.0.

The range of cohort-specific exposure-effect gradients estimated by the EPA is 670-fold wide, with a minimum[2] $K_L = 0.0001$ in friction products workers and a maximum $K_L = 0.067$ in asbestos-cement workers. The updates available in 1991 to the HEI-AR for five studies were not significantly different from the data available to the EPA. The HEI-AR report mentions a study on Australian crocidolite miners not available to the EPA in 1986 [Armstrong et al., 1988; de Klerk et al., 1989], with an exposure-effect gradient estimated by conditional logistic regression at $K_L \approx 0.01$ (95%CI: 0.008-0.020). The risk gradients shown in Table A-7 have remained roughly the same after the few study updates published over the last decade.

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[1] This approach assumes that zero intercepts reveal external comparison biases. Accordingly, this bias is corrected by dividing a cohort's SMRs for all its exposure strata by the zero-dose intercept SMR.

[2] In fact, two estimates were negative by simple regression but were forced to a positive slope[Nicholson, 1986].

Table A-7 Comparison[1] of the Asbestos-Lung Cancer Exposure-Effect Gradients Estimated by Various Risk Assessment Groups From Cohorts of Asbestos Workers[2]

	Percent Increase in Lung Cancer Per f-y/mL of Exposure ($100 \times K_L$)				
Study	EPA[b]	CPSC[c]	NRC[d]	Ontario Royal Commission[e]	HSC[f]
Dement et al. (1983b)	2.8	2.3	5.3	4.2	
McDonald et al. (1983a)	2.5				1.25
Peto et al. (1985)[g]	1.1	1.0	0.8	1.0	0.54
McDonald et al. (1983b)	1.4				
Berry and Newhouse (1983)	0.058	0.06		0.058	
McDonald et al. (1984)	0.010				
McDonald et al. (1980)	0.06	0.06	0.06	0.020 – 0.046	
Nicholson et al. (1979)	0.17	0.12	0.15		
Rubino et al. (1979)	0.075	0.17			
Seidman (1984)	4.3	6.8[h]	9.1[h]		
Selikoff et al. (1979)	0.75	1.0	1.7	1.0	
Henderson and Enterline (1979)	0.49	0.50	0.3	0.069	
Weill et al. (1979)	0.53	0.31			
Finkelstein (1983)	6.7	4.8		4.2[i]	
Newhouse and Berry (1979)					
Males			1.3		
Females			8.4		
Values used for risk extrapolation	1.0	0.3 – 3.0	2.0	0.02 – 4.2	1.0

[a] Adapted from Nicholson (1986).

[b] U.S. Environmental Protection Agency (Nicholson 1986).

[c] U.S. Consumer Product Safety Commission (1983).

[d] National Research Council (1984).

[e] Ontario Royal Commission (1984).

[f] U.K. Health and Safety Commission (Doll and Peto 1985).

[g] Earlier reviews cited Peto (1978) or Peto (1980), and some noted that all men employed after 1951 suffered a higher dose-specific risk ($100 \times K_L = 1.5$).

[h] Data from Seidman and colleagues (1979).

[i] Unpublished data supplied to the Commission.

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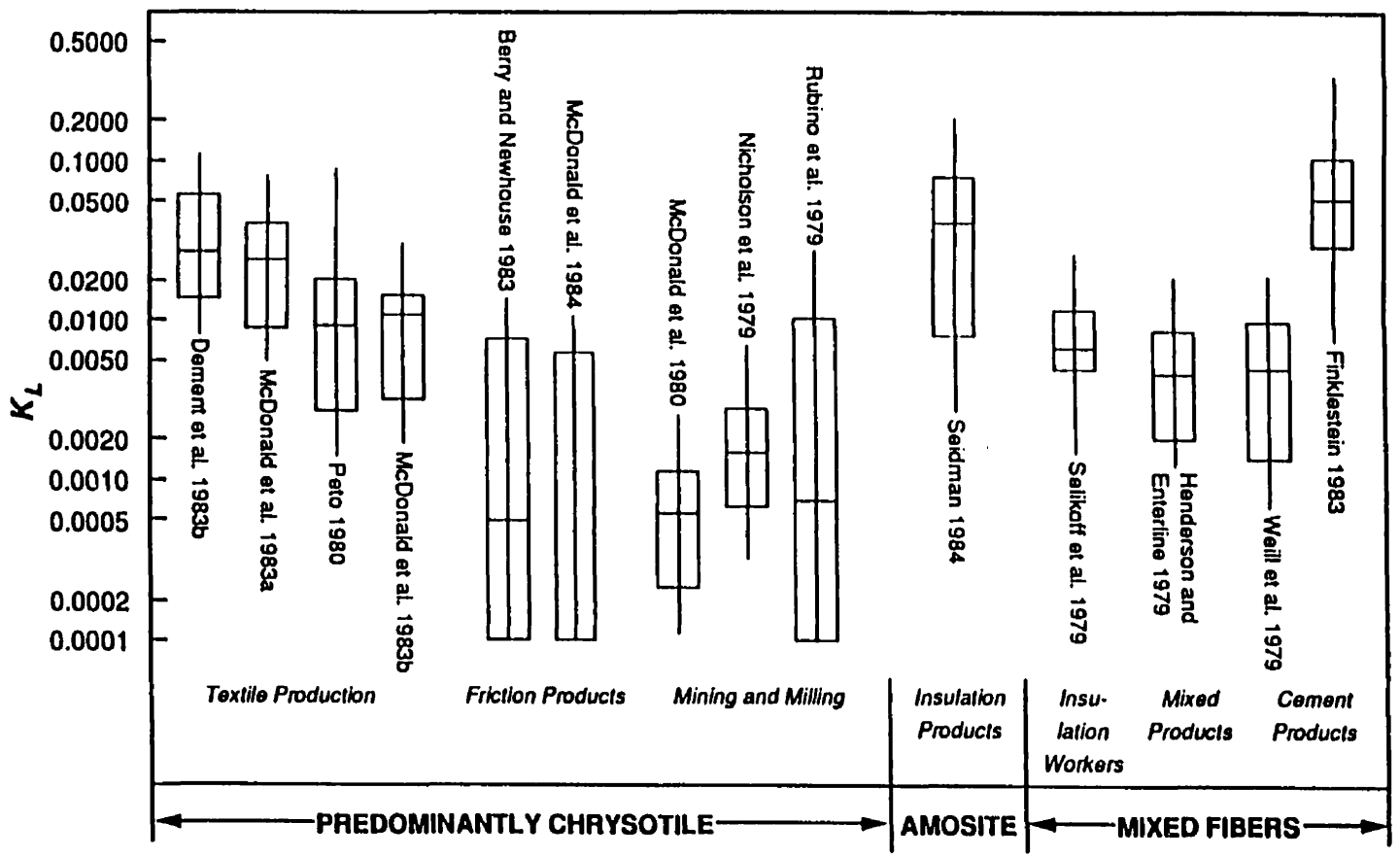
[1] Table taken from the HEI-AR report [1991, p. 6-10], itself adapted from the EPA report [Nicholson, 1986, p.52].

[2] One occupational study should be added. 92 lung cancer deaths were observed in a cohort of 6,500 male Australian crocidolite ex-miners and millers[Armstrong et al., 1988]. The cohort's exposure-effect gradient K_L was estimated at 0.01 (95%CI 0.008-0.020). [de Klerk et al., 1989]

Figure A-2 Asbestos-Lung Cancer Exposure-Effect Gradients[1] and Confidence Intervals Estimated for Cohorts of Asbestos Workers by the EPA[2]



1 A linear exposure-response gradient or slope K_L represents the excess relative risk of lung cancer added by each f-y/mL unit of cumulative asbestos exposure.

2 One occupational study should be added to the graphic. 92 lung cancer deaths were observed in a cohort of 6,500 male Australian crocidolite ex-miners and millers[Armstrong et al., 1988]. That cohort's exposure-effect gradient K_L was estimated at 0.01 (95%CI 0.008-0.020). [de Klerk et al., 1989]

Figure A-2 shows the EPA's exposure-effect estimates [Nicholson, 1986] for each study. The cohorts are grouped by type of fibre and by industrial process, and the Australian crocidolite miners study [Armstrong et al., 1988; de Klerk et al., 1989] would be represented by a 15th bar in a fibre/process category of its own[1]. There are two indicators of the uncertainty of K_L for each cohort. The open bar represents the uncertainty of K_L attributable to the 95%CI of the ratio measure of effect. The vertical line combines the measure of effect's 95%CI with a subjectively estimated geometric standard error of 2 due to exposure measurement error in each study and extends two-fold either side of the open bar. The uncertainty or "plausibility" interval around each individual study's K_L estimate was estimated between 25-fold wide and 200-fold wide according to Nicholson's estimates [Nicholson, 1986].

The exposure-effect gradients were lowest for chrysotile friction-products workers and chrysotile miners and millers; the gradients were largest[2] for amosite-products workers and "chrysotile textile" workers[3], followed by crocidolite miners and millers, insulators and other workers exposed to chrysotile-amphibole mixtures. The seeming toxicity differentials by fibre type and industrial process were not clearly significant statistically and were interpreted differently by various risk assessors. In the EPA's assessment, Nicholson [1986] excluded miners and millers from the precision-weighted geometric mean[4] of K_L because fibres in mines and mills would supposedly be larger, less respirable and less carcinogenic than in other asbestos industries with fibre size distributions supposedly closer to that of environmental fibres to which the general population is exposed. Concomitantly, the asbestos-cement workers and amosite workers studies were included despite their moot exposure assessments and the asbestos-cement eccentric exposure-effect pattern[5].

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1 The cohort's exposure-effect gradient K_L was estimated at 0.01 (95%CI 0.008-0.020).

2 Although the gradient estimated by Nicholson for Finkelstein's cohort of cement workers was the highest, this may well be an artifact since the most straightforward analyses showed a negative gradient.

3 5-15% of the asbestos used in the chrysotile textile industry was crocidolite or amosite.

4 The toxicity gradient originally estimated by Nicholson for the EPA would have been around K_L=0.0065 instead of K_L=0.01 if the three cohorts of chrysotile miners and millers had not been excluded from the final estimation.

5 This was the only study whose gradient estimate departed significantly from the central estimate $K_L = 0.01$. However, exposure estimates were dubious [HEI-AR, 1991], the exposure-effect pattern had the shape of a

The uncertainty of the EPA's final K_L estimate was represented by a 7-fold wide 95%CI centered on a geometric mean K_L=0.01, and the uncertainty of predicting an individual case was represented by a 100-fold wide 95% prediction interval. The CPSC presented a 10-fold range of K_L centered on a geometric mean K_L=0.01, whereas the ORCA presented a range of study-specific gradient estimates, abstaining from making a bottom-line point estimate[1].

b) Discussion

Major uncertainties underlie the exposure-effect gradient assessments. They have been recognized by most risk assessors and have been summarized by the HEI-AR panel [1991, p.6-9] as: 1) the untestable exposure-effect linearity in the low-dose range, 2) risk differential of different asbestos species, 3) the inadequacy of the exposure data for all industrial cohorts so far studied, and 4) the assumption that fibres longer than 5 µm represent most of the lung cancer risk. Furthermore, the uncertainty of the final risk estimate may not be symmetric, and heterogeneity of the data may not allow summarizing the data in a single point estimate.

i. Linear Exposure-Effect and Exposure-Dose Assumptions

As stated above, asbestos risk assessments use a linear *exposure*-effect[2] model based on the one-hit linearized multistage carcinogenesis *dose*-response model [Armitage and Doll, 1961]

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convex parabola, and a simple weighted linear regression gave a negative slope [Nicholson, 1986]. Other authors dismissed the study as too unreliable for consideration [Doll and Peto, 1985; Liddell and Hanley, 1985]. Similarly, the study of amosite-workers had a dubious exposure proxy (recent measurements taken in 2 plants other than the shut down plant of the study's cohort) and a high excess risk in a short-term and less exposed group.

[1] The first U.K. H.S.C. assessment had previously adopted the same approach [Acheson and Gardner, 1980; Acheson and Gardner, 1983].

[2] Terminology: For consistency and specificity, the use of the term "response" is restricted to indicate the result of an experimental administered exposure or dose ("challenge"). "Effect" denotes *ex post* observed results of involuntary exposure. More importantly, the usage of "dose" should be restricted to indicate the quantity of material entering the body ("administered dose") or the quantity of the more carcinogenic component of that material reaching the target tissue or cells of the target organ ("biologically effective dose"). "Exposure" represents levels (instantaneous or cumulative) in the immediate external environment of the exposed person or animal, that could likely be absorbed by the individual's body.

for its simplicity and statistical convenience, its rough consistency with occupational data, and its tendency to overestimate risks in the low-exposure range [NRC, 1984]. Even though the correct form of a dose-response model is very uncertain when risk extrapolations are made orders of magnitude downward from high doses which often produce physiological responses that do not occur at lower doses [Munro and Krewski, 1983], the choice of the model is crucial. For instance, a linear model can predict risks 5 orders of magnitude larger than those predicted by a "log-probit" model at doses 5 orders of magnitude smaller[1] than those at which models would be fitted and would coincide [Brown and Mantel, 1978]. Unfortunately, risk assessments on asbestos do not discuss the *likelihood* nor the degree of "conservativeness" of the linear model with respect to alternative models. For instance, risk assessments might discuss the exposure-effect issue along lines similar to the following overview which does not aim to be complete but rather to propose an approach and to explore issues which should be discussed more competently by experts and risk assessors.

1.a) At the cellular level, models that are or tend to be sublinear have been gaining popularity among experts over the one-hit multistage model: the initiation-promotion clonal growth model [Moolgavkar and Knudson, 1981] and other variants of the multistage model [Hoel et al., 1983; Gaffney and Altshuler, 1988; Hoel, 1991; Stein, 1991; Moolgavkar and Luebeck, 1992; Hoel and Portier, 1994]. Thus present trends in cellular-level carcinogenesis modeling suggest that the linear model would tend to overestimate carcinogenicity, although supralinear dose-response relationships cannot be totally excluded [Bailar et al., 1988; Sterling et al., 1993].

1.b) In the case of asbestos, some authors contend that non-mutagenicity of asbestos *in vitro* [Mossman et al., 1990], carcinogenesis theory (Cairns' stem cell hypothesis, [Browne, 1991]) and results of *in vivo* experiments [Davis and McDonald, 1988] imply a threshold or sublinear dose-response [Browne, 1986; 1991]. Theoretically, non-threshold carcinogenesis models pertain to genotoxic or initiating carcinogens, yet there is doubt as to whether or not asbestos is

[1] This dose differential resembles that between past asbestos workers and the general population today.

a complete carcinogen at low doses [Van den Hooff, 1986; Mossman and Craighead, 1987; Pearce, 1988: Albert, 1989; Weiss, 1990; Walker et al., 1992; Mossman, 1994; Roggli et al., 1994b; Vainio and Boffetta, 1994], particularly in respect to lung cancer. In a recent "Review of Fibre Toxicology" for the U.K. Health and Safety Executive [Meldrum, for the HSE, 1996], Meldrum concluded that experimental evidence suggests that asbestosis and lung cancer emanate both from the same underlying inflammatory condition, and that a threshold must exist since "exposures which are insufficient to elicit chronic inflammation/cell proliferation" will not increase the risk of lung cancer.

2.a) Upstream from the cellular level, a major uncertainty with the model is the implicit assumption of a linear relation between external exposure level and effective cellular-level dose: the exposure-dose relationship. Generally, toxicants to which the body is exposed are filtered by pathway-specific chains of events before becoming "effective doses" at the cellular level. Mathematical simulations suggest that organ-level or body-level defense mechanism with non-zero efficiency against invaders will transform even linear cellular-level dose-response relations into sublinear exposure-response relations [Holland and Sielken, 1993; Sielken et al., 1994; Stevenson et al., 1994]. Thus, the justification of an epidemiological model on the basis of an uncertain cellular-level or organ-level carcinogenesis model is not manifest. Carcinogenesis models which suggest specific dose-response models are based on effective doses reaching target cells, whereas epidemiological studies and risk assessments observe effects associated with external exposure levels before intake. The relation between these metrics is most likely sublinear, implying that even a linear dose-response relation would likely result in a sublinear exposure-effect relation. The *exposure-dose* relation issue has been obscured by risk assessment terminology which refers to "dose-response" assessment instead of "exposure-effect" assessment while using epidemiological data and targeting human risks.

2.b) With respect to asbestos fibres, airborne fibres must be inhaled, respired, deposited and must then overcome mechanisms such as clearance, dissolution, leaching and asbestos-body coating before they can induce or promote cancer. At low exposure levels, efficiency of such

barriers and defenses must be greater than at high exposure levels, resulting in a sublinear exposure-dose relation, as suggested by macrophage saturation and overloading in animal experiments [Morrow et al., 1991; Davis, 1994; Hext, 1994]. Thus, the asbestos-lung cancer exposure-effect relation would more likely be sublinear than linear [Davis and McDonald, 1988], a hypothesis supported empirically by a review of mineral fibre inhalation experiments on rats that fitted a quadratic relationship between the risk of lung cancer and the concentration of airborne fibres [Lippmann, 1994].

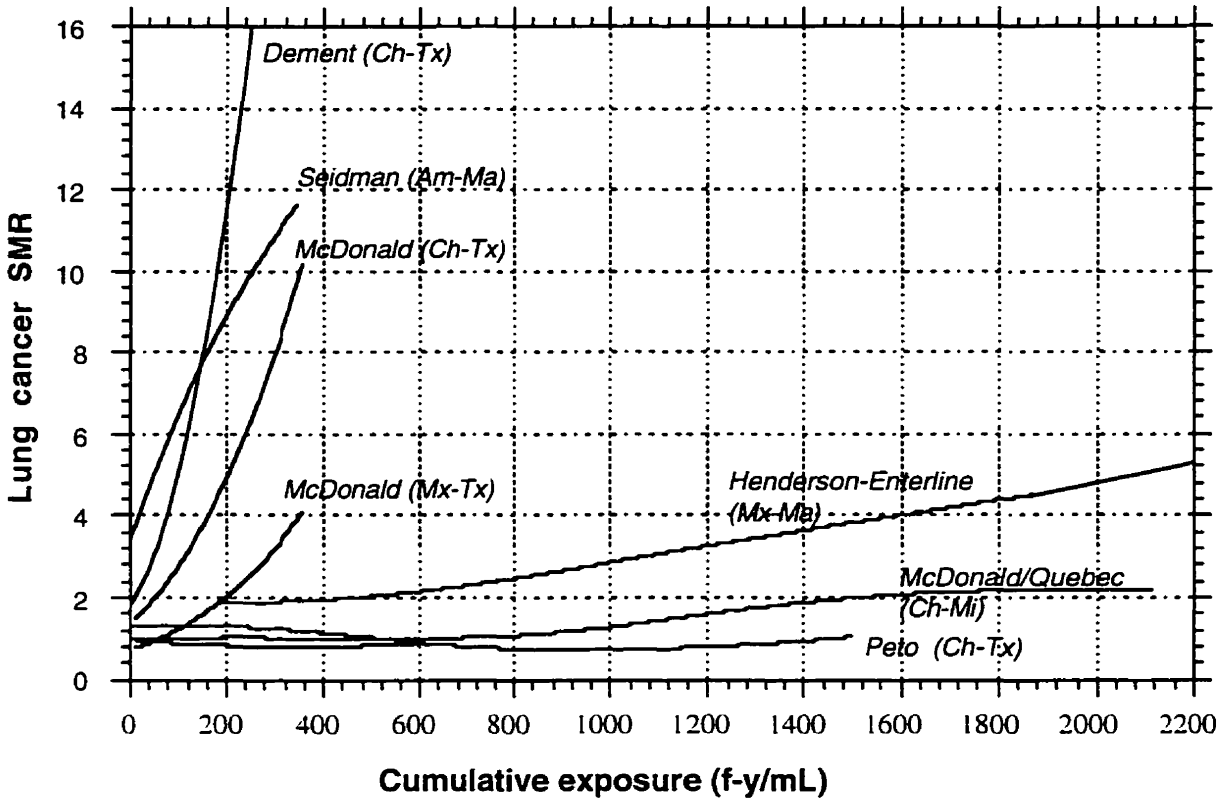
3) Not only is exposure intensity unlikely to translate linearly into an effective dose, but the cumulative exposure metameter used as a proxy for dose weighs different exposure experiences of a subject equally over time even though induction time and biopersistence in the lung imply that past exposures should bear more risk for lung cancer than more recent exposures (see Section B.1.5 on Lung Burden). "Time windows" have been suggested to account for latency [Rothman, 1981] whereas a time-since-exposure weighting has been suggested to account for the retention of toxic substances [Jahr, 1974]. Neither of these approaches has been used in major risk assessments based on occupational asbestos data. Finally, exposure duration and exposure intensity probably should not have the same weight in the metameter [Vacek and McDonald, 1990; Vacek and McDonald, 1991]. Altogether, these flaws of the cumulative exposure metameter would tend to blur or attenuate the exposure-effect relationship but also to hide any threshold or sigmoid (sublinear) exposure-effect relationship. [Armstrong, 1990; McDonald, 1990; Verkerk and Buitendijk, 1992]

4) As to epidemiological evidence, the graph in Figure A-3 shows that, among seven typical occupational asbestos datasets, exposure-effect patterns seem more sublinear than linear but not significantly so. The only supralinear pattern among these seven studies is a study of amosite workers with an indirect and problematic exposure assessment. Overall, risk assessment groups are correct in stating that the linear model is consistent with most occupational datasets. Still, linearity may be more an upper bound inference than suggested by the graph. In epidemiological studies of asbestos workers, measurement error of past exposure

levels has been substantial [Nicholson, 1986], concentrations were most likely non-differentially underestimated [Berry and Lewinsohn, 1979] and errors would probably have been proportional to the mean [Leidel and Busch, 1985]. As a result of these errors and other study design characteristics, the observed patterns would tend to obfuscate sublinear and even threshold exposure-effect relations [Armstrong, 1990; Armstrong et al., 1990; Verkerk and Buitendijk, 1992; Verkerk and Buitendijk, 1993] [Enterline, 1976; McDonald, 1990] [Vacek and McDonald, 1990]. Extensive data on chrysotile miners and chrysotile-cement workers have been said to suggest a threshold for lung cancer [Liddell et al., 1992; 1993; 1994][Hughes and Weill, 1991]. Data on tremolite-contaminated vermiculite miners also suggest a sigmoid relation between lung cancer and exposure intensity [Vacek and McDonald, 1991]. Overall, the exposure-effect relation would more likely be sublinear than linear or supralinear on the basis of epidemiological data and suspected biases.

To conclude, on the basis of available theoretical and empirical considerations, a linear exposure-effect relationship between asbestos and lung cancer seems less likely than a sublinear relationship, but more likely than a supralinear one. Although the linear model may be justified as a conservative combination of public health prudence, scientific parsimony and empirical evidence, the resulting risk estimates should be presented as "conservative", not as "best estimates". This conservativeness might be estimated roughly by fitting different exposure-response models to the different datasets and figuring which model had the best statistical fit with respect to all datasets. Then, the estimates based on this model could be contrasted with linear estimates to determine safety factors.

Figure A-3 Model-Free Fitted Curves on Selected Occupational Asbestos/Lung-Cancer Datasets, Using a Distance-Weighted Least Squares Algorithm



Distance-Weighted Least Squares (DWLS) fitted curves for various occupational asbestos/lung-cancer datasets [Henderson and Enterline, 1979; McDonald et al., 1980; Dement et al., 1983b; McDonald et al., 1983a; McDonald et al., 1983b; Peto et al., 1985; Seidman et al., 1986]. (Ch = chrysotile, Am = amosite, Mx = mixed fibres, Tx = textile, Ma = manufacturing, Mi = mining and milling.)

ii. *Heterogeneity of Exposure-Effect Slopes and "Selection Bias" of the Occupational Data*

Risk assessments on asbestos and lung cancer have recognized the large disparity between exposure-effect gradients 670-fold among occupational cohorts and 200-fold among asbestos industries and this heterogeneity has not withered away since 1985, quite the contrary. Many experts and assessors believe that this heterogeneity appears to be related to industrial process - with the corresponding fibre physical characteristics, exposure circumstances and co-pollutants - and to asbestos species. Still, this explanation would not account for all the variance between cohorts, statistical error could account for a substantial part of the variation, and other unknown factors might be acting. Invoking these reasons, most asbestos risk assessments finally pool the occupational studies and estimate a single point average gradient as if the between-cohort and between-industry variances were attributable to random error. Doing this however contravenes to the basic statistical principle according to which a single point estimate cannot summarize heterogeneous data [Breslow and Day, 1987]; indeed, such an average overfits or is too specific to the data at hand and cannot be generalized to the common denominator category of the data - asbestos exposure.

The EPA's asbestos risk assessment and consequently its HEI-AR and INSERM offsprings may have been most inconsistent in this respect. On the basis of the 14 studies reviewed by the EPA, the precision-weighted average was K_L=0.0065, and only one out of five industrial process groups ("mixed fibre products manufacturing or use") had a 95%CI overlapping with the average K_L. Yet, the EPA decided to exclude only the 3 studies on chrysotile miners and millers from the final K_L estimate because a) the three studies had low K_L estimates, b) the proportion of truly "easily respirable" carcinogenic fibres in samples of nominally respirable fibres could be lower in mining and milling than in asbestos transformation industries, and c) mining and milling is an industrial process remote from the transformed products used by the general population. The 3 exclusions raised the average K_L to 0.01. Still, the 670-fold

cohort-specific and the 200-fold industry-specific disparities remained between the 11 retained studies[1]. In fact, the exclusions seem somewhat arbitrary since a) the three excluded studies are not outliers in a statistical sense, b) the main rationale is based on the very industrial-process heterogeneity disputed by the EPA and denied by making a single point average, and c) with respect to the criterion of the general population's exposures, it could be argued that amosite-products and asbestos-textile[2] workers have been exposed to fibres much less representative of the general population's exposures.

To be consistent about the between-study heterogeneity issue, risk assessments should have chosen one of the following two possible paths. Given the numerous and important uncertainty factors listed by the EPA and the HEI-AR, the differences between asbestos industrial processes might be dismissed as not discernible from random variation, but then studies of chrysotile miners and millers should not have been excluded from the final estimate, which would then be lower. Alternatively, the heterogeneity may be considered as true, implying that fibre dimensions, asbestos species and other unknown factors which characterize different industrial processes may induce different lung cancer risks even though we may not know precisely how. Then, to obtain a "best" estimate, criteria for *weighing* occupational studies in the final pooled analysis should combine statistical precision and *representativeness* [3] of different industrial cohorts with respect to the characteristics and sources[4] of airborne asbestos fibres to which the general population is exposed (>98% chrysotile[5], and >99% short

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[1] The K_L estimate for the friction products industry did not differ significantly from the other industries due to its large confidence interval, but the latter overlapped much more with chrysotile-mining than with the other processes. Moreover, the K_L estimate for friction products is 5 times smaller than for chrysotile-mining. So the argument of separating out purportedly less carcinogenic unprocessed fibres from more carcinogenic processed fibres is not consistent with the evidence.

[2] Asbestos fibres used for textile products are the longest. The average length of respirable airborne fibres in textile plants would have been longer than in other asbestos industries.

[3] A point estimate assumes that the selected occupational cohorts represent adequately the general target population's joint distribution of lung cancer risk factors (including host susceptibility).

[4] The general population has practically no exposure to airborne releases from asbestos textile products. Asbestos in sound insulation boards and general sprayed-on insulation in walls and roofs is mostly chrysotile, although amosite is sometimes incorporated. Amosite has been used particularly intensively on boilers, in boiler rooms and on hot steam pipes [A.Dufresne, McGill U., personal communication, 1996; HEI-AR, 1991, p.4-80]; accordingly, mostly maintenance workers would be exposed to important levels of amosite.

[5] According to 8 studies compiled by Nicholson [1989], mass concentrations of asbestos fibres measured by TEM in the USA (7 studies, 233 samples) comprised 2.4% amphibole, 97.6% chrysotile. Lee et al. [Lee et al., 1992; Corn, 1994] have counted 2.5% of amphiboles among asbestos "structures" in U.S. schools, 97.5%

fibres < 5 μm[1]). Accordingly, cohorts of chrysotile miners/millers should not be excluded in the estimation of K_L. Rather, friction products workers and chrysotile miners and millers should receive more weight, whereas cohorts exposed to amphiboles or mixtures or to longer chrysotile fibres should receive less weight. On the basis of a rough calculation[2], this approach might drive down the K_L estimate to about 0.0023.

To conclude, the heterogeneity of the occupational data is obfuscated by using a single point estimate of K_L. Moreover, this strategy forces assessors to choose a single weighing scheme of the cohort data, an arbitrary one in view of the insufficient knowledge. In fact, since the characteristics of fibres to which asbestos workers in the 14 cohorts have been exposed are not representative of those of the targeted general population, and since heterogeneity of the data suggests that such characteristics induce very different risks, a simple precision-weighted average is akin to opportunistic sampling and is distorted by a "selection bias". Excluding chrysotile miners and millers worsened this selection bias. The studies should either be weighted explicitly by their representativeness of environmental exposures in the general population or should not be pooled in a single point estimate.

iii. Deficient Occupational Exposure Data

In nearly all the studies of historical cohorts of asbestos workers, past exposure measurements were relatively few and were done haphazardously with sampling equipment and techniques which measured aerosols but were not specific to asbestos fibre counts until the mid-1960s. Moreover, the membrane filter method and techniques for counting fibres under a microscope were not standardized until the mid-1970s and continued to improve through the 1980s. Thus the same unit of measurement ("f/cc" or "f/mL") could represent about 10 times more fibres in

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chrysotile. According to Corn [1994] and Nicholson [1989], asbestos in outdoor air is practically all chrysotile. Note: The mass proportion of amphiboles was much higher (40%) in Paris (135 samples) [Sébastien et al., 1976; Sébastien et al., 1980a].

1 Corn [1994] and Lee [1992] reported 99.7% of TEM asbestos structures to be shorter than 5 μm in length.

2 My own estimate was based on weighing chrysotile textile by 4%, friction products by 40%, mining and milling by 40%, amosite manufacturing by 1%, insulation and other mixed fibre products industries by 10%.

the 1960s than in the 1980s [Rickards, 1994]. Conversion factors from thermal precipitators and midget impingers to asbestos fibre counts have proven to be extremely variable [Gibbs and Lachance, 1972; 1974] and the proportion of fibres in the total dust may have been higher in less dusty jobs than in dustier jobs [Dagbert, 1976]. Many industries produced different asbestos products or changed asbestos mixtures over time. One study used exposure data from another plant of the same company in a later era [Seidman et al., 1986], underestimating thereby past exposures. According to Liddell [1991], exposures before the 1960s would have been underestimated by a factor of 2 or 3 in many studies. Most studies have not accounted for the much longer workweeks (48-54 hours) before the middle of this century. Finally, the most polluted areas were undersampled in the past [Peto et al., 1985; HEI-AR, 1991], and measurements exceeding acceptable or standard levels were often downplayed by re-sampling until the standard or an acceptable level were met (these corrective-action samples being added to the dataset of measurements available to epidemiologists).

In view of these exposure measurements problems, the uncertainty intervals estimated in Table A-8 seem too narrow. It is not sure in what direction the exposure-effect gradients would be biased as a result of exposure measurement errors. Non-differential misclassification tends to depress exposure-effect slopes, mostly in the higher exposure range since these errors tend to be proportional to exposure level and exposure duration; this would also obfuscate threshold and sublinear exposure-effect relations. Erroneous dust-to-fibre conversion factors could bias the relation in any direction, depending on industrial process[1], era, production operation[2] [Dement, 1982; Gibbs and Lachance, 1974] and dust level [Dagbert, 1976]. Over-representation of more recent exposures tends to underestimate pre-1970 exposures, increasing spuriously the exposure-effect slopes. Overall, exposure measurement errors can be said to increase the uncertainty of final exposure estimates, and applying a geometric error factor of 2 to the upper and lower 95%CLs of each cohort's K_L estimate as suggested by Nicholson [1986], is not unreasonable but seems like a minimal formalization of this uncertainty. Since

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[1] Textile, mining, etc.

[2] E.g. carding vs. weaving in the textile industry; bagging vs. excavating in the mining/milling industry.

cumulative exposures seem more likely to have been underestimated however, the exposure-effect gradients would tend to be overestimated and thus the gradients' confidence intervals should be skewed toward lower values.

iv. Potential Biases of the Gradient Estimates

All the occupational studies have important deficiencies, particularly in terms of referent population, smoking data, and exposure assessment. In the extensive EPA review [Nicholson, 1986], each dataset was analyzed in numerous ways to attempt to correct for such biases: 1) empirically weighted regression, 2) various adjusted SMRs for different local rates or other factors, 3) forcing the regression line through zero when non-differential misclassification or random error might cause an intercept greater than 1.0, 4) using a "relative slope model"[1] to correct for suspected healthy worker effects which increase the K_L estimates, 5) using internal RR analyses when possible, 6) dividing the overall excess SMR for each cohort by the cohort's average cumulative exposure. After considering possible biases in each study, Nicholson chose the most fitting analysis and, when in doubt, leaned towards higher slope estimates. 10 times out of 14, the EPA's final cohort-specific slope estimate was larger than that estimated with the plain weighted linear regression, and the overall gradient average was more than doubled as a result. In two instances, negative slopes thus became steep positive gradients. The EPA's assessment - as all asbestos risk assessments - was conservative and avoided underestimating the toxicity gradients as possible. Of course, the biases may not have been fully corrected; more likely however, they may have been over-corrected.

A possible underestimation bias mentioned in the ORCA report (1984, p.486) but not accounted for quantitatively by any risk assessment is the competing risk of death from asbestosis or non-specific fibrotic lung disease [Schneiderman, 1981; Schneiderman et al.,

[1] The slope obtained in relative slope regression of SMRs [Acheson and Gardner, 1983; Liddell, 1984; Hanley et al., 1985; Liddell et al., 1985] is similar to that obtained by performing a simple weighted linear regression and then dividing the slope estimate by the estimated intercept. Thus, if an intercept were 0.67 and the slope estimate 2.0, the new adjusted slope would near 3.0. It implies that an intercept $\neq 1.0$ reflects bias due to the non-comparability of the referent population.

1981]. Mortality from fibrosis in these historical cohorts of asbestos workers was about equal to the excess lung cancer mortality. Hence, asbestosis may have masked a significant proportion of lung cancer deaths. The potential extent of this bias has not been assessed.

v. Occupational and Environmental Studies Reporting Either Asbestos-Related Risks or Asbestos Exposures But Not Both Simultaneously

Risk assessments necessarily rely on occupational studies with quantitative exposure data to make an exposure-effect assessment. However, other asbestos-related epidemiological studies should be systematically reviewed for both risk identification and risk characterization purposes. Epidemiological studies without quantitative exposure data may suggest mechanisms, interactions, other risks, qualitative exposure factors, may support or weaken certain assumptions, the presence or absence of risk, etc.

Studies on Lung Cancer Risks in Asbestos Workers

Epidemiological studies have examined lung cancer risks in some 40 cohorts of workers exposed to high levels of airborne asbestos fibres. Asbestos exposure has been quantified in less than half of these studies. However, since the type of asbestos fibre and industrial process are both characterized to some degree in occupational asbestos studies, the studies as a whole can shed light on the plausibility of greater lung cancer risks being associated with amphibole exposures and certain industrial processes. Thus, a risk differential by asbestos species has been proposed for lung cancer in reviews of male and female cohorts of asbestos workers [Ohlson and Hogstedt, 1985; McDonald and McDonald, 1986; Hughes, 1991]. According to Hughes, the excess risk of lung cancer was +26% (95%CI: 16% to 37%)[1] among "chrysotile" workers, +118% (95%CI: 105% to 131%) among "mixed asbestos" workers, and +207% (95%CI: 167% to 251%) among "amphibole" workers. Although one may dispute the inclusion or exclusion of one or two studies in each group, the general picture would not change substantially. Overall, this empirical evidence suggests that there is a significant

[1] I calculated the 95%CIs using Byar's approximation for SMRs [Breslow and Day, 1987, formula 2.13, p.69].

"amphibole effect" for lung cancer following a kind of "dose-response" pattern ("chrysotile" vs. "mixed" vs. "amphibole") which could hardly be explained by biases or different exposure levels and durations, a conclusion similar to that of Meldrum[Meldrum, 1996 #9657]. If the whole of the occupational epidemiological evidence had been used, the apparent exposure-effect differential between asbestos species among the 14 studies reviewed by the EPA would have been corroborated, justifying thereby *ad hoc* risk estimates.

Studies on Lung Cancer in Populations Non-Occupationally Exposed to Asbestos

No risk assessment other than the recent French INSERM report has reviewed systematically the available non-occupational epidemiological data, even though such studies bear on exposure circumstances closer to those of the general population than those of past asbestos workers. About half of the studies summarized in Tables A-2 through A-6 were available when the risk assessments were conducted, and most of the non-occupational studies were available to the HEI-AR panel in 1991. These studies have not been examined seriously partly because they had no quantitative exposure estimates and partly because many were not comparative (comprised no referent group). Regarding lung cancer, the evidence is weak, but nevertheless seems to corroborate the higher risk associated with amphibole exposure. The highest relative risk was observed in a crocidolite-mining area, and the lowest in a chrysotile-mining area[1]. In addition to corroborating the purported amphibole risk gradient, the non-occupational epidemiological data could have been used by the risk assessment groups as a "reality check" to see if the excess risks observed in non-occupational asbestos studies were compatible with the risk assessments' estimated exposure-effect gradients, by applying the gradients to the excess risks observed in non-occupational studies to estimate their cumulative exposure. For example, on the basis of the EPA's gradient K_L=0.01, Botha's crocidolite-mining study's RR[2] of 2.17 would imply an average of 28 f-y/mL (90%CI: 11-51 f-y/mL) cumulative continuous exposure[3], whereas Pampalon and Siemiatycki's chrysotile-mining

[1] The New Jersey neighbourhood study is omitted because neighbourhood exposures must have been negligible.
[2] This RR is estimated by pooling all females together irrespective of race.
[3] This reverse projection of cumulative exposure was obtained by dividing the observed excess relative risk (2.17 - 1.00) by K_L (0.01) and by 4.2 (the continuous to workweek ratio).

study's RR of 1.07 would imply an average cumulative continuous exposure of 1.7 f-y/mL (90%CI: 0-12 f-y/mL). Then the assessors could surmise about the likelihood of such model-based exposure estimates and re-assess their exposure-effect estimate in view of these results. For instance, the exposure estimate for the crocidolite-mining area may seem realistic to assessors while the exposure estimate for the chrysotile-mining area could seem unbelievably low. This would suggest that the model could correctly predict risks associated with non-occupational exposure to crocidolite but would considerably overestimate risks associated with non-occupational exposure to chrysotile. If instead the exposure estimate for the crocidolite-exposed population were deemed high, then the assessors might infer that the model would predict non-occupational risks intermediate between those two types of asbestos exposures. Risk assessments should use available non-occupational data for reality checks.

Non-Etiologic Studies on Asbestos Exposures; the Case of Lung Burden Studies

Lung burden studies (defined in Section B.1.5) might not be used directly for quantification of cumulative dose, but may be used to validate [Sébastien, 1989], complement and qualify existing estimates of cumulative exposures, which have there own limitations. Used in comparison with other occupational cohorts or with the general population, they may indicate significant proportions of unsuspected or underestimated past exposures to amphiboles or provide insight into the fibre-size distribution of biopersistent fibres inhaled in the past. Lung burden studies can also be used to assess the relative importance of past asbestos exposures in different non-occupationally exposed populations for which no ambient measurements are available. Occupational lung burdens can be used with past occupational exposure data to help estimate deposition fractions, biopersistence factors and pulmonary clearance rates of different fibre species and dimensions to validate hypotheses about purported toxicity differentials. Such characterizations of past exposures should be used more thoroughly to increase the specificity of epidemiological studies. Poor characterization of past exposures is a serious limitation of asbestos epidemiology; no data should be ignored.

vi. Single Point Estimates Hide the Real Uncertainty of the Risk Assessments

Most risk assessments conclude that their final point estimates are "best estimates"[1]. Yet, as argued above, many indications suggest that asbestos risk assessments on lung cancer tend to overestimate risks at low exposures. The linear model is justified explicitly in risk assessments in the name of conservatism. Pooling heterogeneous cohorts over-representing exposures to amphiboles and long chrysotile fibres relative to the short-chrysotile exposures (98%) of the general population is conservative. The EPA estimated higher exposure-effect gradients than suggested by the standard weighted SMR regressions. Such decisions show that risk assessments on asbestos are "conservatively inclined" and rightfully so.

In view of the huge uncertainty of the estimates, the use of a single "best estimate" confers the impression that the final estimate is more reliable than it actually is. This simplifies decision making, but it warps scientific and socio-political issues by fusing and confusing them[2]. It is not good science not to quantify uncertainties, and not to show their implications in the final risk assessment[3]. Indeed, risk assessments have been discredited in part for coming up with "bottom-line estimates" [Hattis et al., 1987]. If risk assessments are to both do justice to the data and produce usable estimates for decision-makers, they should produce a few estimates ranked as "minimum", "intermediately lower", "best", "intermediately upper" and "maximum" risk estimates. The estimates would be based on making consequent systematic choices between alternative assumptions and analyses throughout the various steps of the risk

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1 " The risk assessment calculations of this sort are "best estimates" in the sense that we have no direct evidence that they are too high or too low. However, no meaningful upper confidence limits can be assigned to them, due to the many uncertainties in the reliability and representativeness of the exposure data, as well as the scientific uncertainties relating to the model itself (...) " [HEI-AR, 1991, p.8-9]

2 The likelihood of single estimates can be exaggerated by interest groups and decision-makers who base their policies on risk assessments, and the estimates can be deprecated in any direction by opposing interest groups when different assumptions and potential biases are downplayed or left aside in the final estimate. When new considerations, data or analyses arise and do not fit well with preceding data or models, a single-scenario risk assessment can be rejected altogether if it did not provide direction for re-evaluating risk estimates.

3 For instance, in the EPA's final risk estimates [Nicholson, 1986, Tables 6-1, 6-2 and 6-3], we only learn in a footnote that "a 100-fold 95%CI should be applied to risk estimates in undocumented exposure circumstances". As to the point estimate, its 95%CI was estimated as K_L as 0.004-0.027. How these confidence intervals were arrived at is not explained adequately in the report. The implications are neither discussed nor quantified in terms of risks. Rather, only the point estimate of $K_L = 0.01$ is used to characterize the risk for the general population.

assessment. This would avoid a confusing result where 'conservative', 'liberal' and 'most likely' assumptions and analyses made at different steps of the process are finally mixed together into an abstract, puzzling and undetermined estimate which cannot be characterized plainly as being neither 'conservative', 'most likely' nor 'liberal'.

A.3.4. Conclusion of the Literature Review

The excess risk of respiratory cancer due to non-occupational asbestos exposure in the general population has been assessed in two ways: non-occupational epidemiological studies and worker-based risk projections.

According to epidemiological data, household exposure to asbestos entails higher risks of mesothelioma and probably of lung cancer, whereas neighbourhood exposure seems to induce similar but lower risks. The non-occupational evidence is very strong for mesothelioma but it is unclear for lung cancer. The evidence clearly incriminates amphiboles, particularly crocidolite and tremolite. As to chrysotile, excess mesotheliomas have been observed in families of chrysotile workers but not in neighbourhood-only-exposed populations of chrysotile mining areas, but this negative finding has little meaning due to the low statistical power of the evidence. Due to the absence of exposure data, no exposure-effect relationship can be inferred from these data and thus no inference can be drawn for general populations.

Cancer risk assessments of environmental asbestos have been conservative and prudent in their methods and in their estimations of the exposure-effect relations. However, their risk predictions are very unreliable because the strong heterogeneity of exposure-effect slopes and exposure circumstances between occupational studies weighs against summarizing the data in

a single exposure-effect gradient estimate. The sensitivity and the uncertainty of the risk estimates should be assessed and accounted for using different assumptions and scenarios.

There is yet no epidemiological evidence to evaluate the validity of the risk projections for general populations. Still, non-occupational studies should be reviewed even in the absence of exposure data. A "reality check" should be completed by applying the estimated exposure-effect gradients to the observed non-occupational excess lung cancer risks to estimate past non-occupational asbestos exposure levels. These exposure estimates could then be compared among various non-occupational studies, as well as with available occupational and general environmental exposure data to see if they fall in middle of this range, etc.

The best way to check if the risk assessments are realistic is to estimate both the lung cancer incidence and the average cumulative asbestos exposure of a population exposed non-occupationally to asbestos fibres.

A.4. OVERVIEW OF STUDY DESIGN

The study comprised three distinct components: 1) an historical exposure assessment comprising several substudies; 2) a risk assessment to predict the study population's expected relative risk of lung cancer according to the EPA model; 3) an ecological study comparing lung cancer mortality rates of females in the asbestos-mining agglomerations with those of an appropriate referent population to determine the observed relative risk. The following justifies and outlines these different study elements.

a) Ecological Study

Obtaining individual lifetime exposure histories of persons non-occupationally exposed to asbestos who died over the last 20 or 40 years was not practical; consequently, an ecological design was chosen. Several conditions made the ecological study a well-suited and valid design with respect to the research objectives. First, individual asbestos exposures in the exposed population were orders of magnitude above those of the referent population so that the exposure-effect association was not diluted by the ecological analysis. In addition, under the "null hypothesis" of a linear relation between cumulative asbestos exposure and relative risk of lung cancer assumed in asbestos risk assessments, the "ecological fallacy" [Robinson, 1950; Selvin, 1958; Morgenstern, 1995] is controlled with respect to exposure. Indeed, since under a linear model average mortality depends only on average exposure, the manner in which the exposure is distributed among the population is irrelevant [Cohen, 1990]. Finally, the study population has been relatively stable over the study period, with little out-migration. More important for validity, migration was probably not more selective with respect to the risk of lung cancer in the exposed than in the referent populations (details in Section C.5.4).

b) Study Base, Exposed and Referent Bases

The population less than 30 years of age was excluded for efficiency, to focus on the population at risk of developing lung cancer, to make the exposed and referent populations more homogeneous with respect to cofactors and more stable geographically since young persons and families constitute most of inter-regional migrations.

The "agglomeration" was the basic unit of analysis. This geographical unit was defined by the OPDQ (Quebec Office of Planning and Development) as a grouping of municipalities with continuous or adjacent built spaces of at least 4,500 residents in 1976. It involved less misclassification error than the smaller municipal unit, and less ecological dilution of effect than the larger census division unit. It also excluded small isolated municipalities (80% of Quebec's municipalities, 13% of the population) which have very different activities, health services and socio-demographic characteristics from those of the asbestos-mining area.

Five agglomerations were excluded from the study; the largest urban agglomerations (Montreal Metropolitan Area and Quebec Urban Community), the industrial and relatively large agglomeration of Sherbrooke which attracted workers and asbestos-exposed cases from the exposed agglomerations, the ship-building agglomeration of Sorel where asbestos exposures may have caused excess deaths, and the agglomeration of Hull where mortality has been under-reported in the past because of its proximity to the Province of Ontario. The study was thus limited to 62 of the 67 OPDQ agglomerations.

The cases were considered as arising from a study base consisting of the person-years that women of the study population lived in the study agglomerations, at any time between 1970 and 1989, while they were at least 30 years of age.

The exposed study base between 1970 and 1989 comprised 221,400 PY (person-years) lived in the two exposed agglomerations by female residents at least 30 years of age. The agglomeration of Asbestos included the municipalities of Asbestos, Danville, Shipton and

Trois-Lacs and counted 73,160 PY. The Thetford Mines agglomeration included the municipalities of Thetford Mines, Black Lake, Thetford-partie-Sud, and Rivière-Blanche and counted 148,200 PY.

The referent study base comprised 8,629,600 PY lived in the 60 unexposed agglomerations (240 municipalities) between 1970 and 1989 by female residents of at least 30 years of age.

c) Outcome Variable and Measures of Effect

Lung cancer mortality was used as a surrogate for incidence because: a) mortality data were available for a longer period of time with a more constant quality of data; b) lung cancer mortality reflects incidence due to the short survival of lung cancer cases, and c) lung cancer mortality was the outcome used in the asbestos occupational studies on which environmental asbestos risk assessments have been based.

The cases were identified from individual computerized death certificate records of the Quebec mortality Registry for the years 1950-1989. The underlying cause of death coded in the Mortality Registry was used for case ascertainment. The municipality of residence at time of death coded on the death certificate was used to classify the deaths by asbestos exposure region. The denominator data was based on the Canadian quinquennial censuses.

The lung cancer mortality observed from 1970 to 1989 in the exposed study base was compared to that of referent study base, adjusting for age. Cause-specific SMRs, SRRs and SPMRs were computed for each of 42 mutually exclusive and exhaustive causes of death.

These observed lung cancer relative risks were compared with the lung cancer relative risks predicted by the EPA's risk assessment and other risk assessments on asbestos and lung cancer.

d) Exposure Assessment

The purpose of the exposure assessment was to estimate the average cumulative asbestos exposure experienced by the female population of Quebec's asbestos-mining agglomerations. The two components of cumulative exposure, past *exposure intensity* and past *exposure time* , were estimated separately.

Due to lung cancer latency, the relevant exposure period for this study was pre-1970 and mostly pre-1955. Unfortunately, there were no measurements of asbestos or even total dust in the ambient air before 1972. Past ambient asbestos exposures in the asbestos-mining towns of Quebec were therefore estimated using only indirect evidence and expert opinion.

First, diverse qualitative and quantitative approaches were used to estimate past outdoor concentrations.

1. Historical maps were drawn of the location of emission sources relative to the residential areas back to 1920.
2. History of production process and dust controls was documented in detail.
3. The relation between post-1972 ambient asbestos measurements and production levels and dust controls was estimated by regression and then extrapolated on past production levels.
4. Asbestos fibre lung burdens of asbestos workers and local residents were analyzed by non-linear regression with a biokinetic model to derive lifetime average exposure levels.
5. A survey of women documented chronological and spatial patterns in sightings of asbestos depositions.
6. Dusts retained by modern baghouses were characterized to evaluate past asbestos emissions and concentrations using an EPA aerosol dispersion model.

The results were reviewed and synthesized by a panel of five experts in asbestos measurement and exposure assessment. The above-mentioned sources of data were very incomplete, but as a

whole they were complementary and consistent enough for the panel to agree on past neighbourhood exposure estimates for each town for four key years. Values were interpolated for each year from 1900 to 1984 on the basis of production levels estimated by town.

Second, household-contact exposures were estimated using very scarce and remote data, but the panel did not agree on these estimates. Still, the contribution of past household exposures to the global asbestos exposure of the population was estimated under various scenarios to take into account the greater uncertainty of household exposure data.

Finally, residential and household exposure histories of respondents to a representative population survey conducted in 1989 were linked to the past exposure intensity levels estimated by the panel to compute the exposed population's average cumulative exposure by agglomeration over the follow-up period.

e) Exposure-Effect Assessment and Comparisons

The linear relative risk models and toxicity gradients estimated by various risk assessments were applied to the estimated average cumulative asbestos exposure of the exposed population to determine the asbestos-induced lung cancer mortality predicted in the study population. These risk assessments' predictions were compared with the observed lung cancer mortality.

PART B. EXPOSURE ASSESSMENT

The purpose of this thesis is to compare the excess lung cancer mortality of the female population of Quebec's asbestos mining agglomerations to that predicted by the application of the EPA linear exposure-effect model for asbestos. This requires knowledge of the exposed population's average cumulative asbestos exposure. The object of the Exposure Assessment is to estimate the cumulative lifetime exposure of the exposed study base. The two components of cumulative exposure, past *exposure intensity* and past *exposure time* , were estimated separately.

The past asbestos exposures of the female residents of the mining towns comprised a general *"neighbourhood"* component to which all residents of an asbestos mining town were exposed by virtue of the proximity of outdoor asbestos emission sources; thus these sources contaminated the outdoor air which was breathed both outside and inside neighbouring households since the outdoor air would infiltrate the these households. Past asbestos exposures of most female residents of the mining towns also comprised a *"household-contact"* component specific to those residents who shared a household with an asbestos worker who brought home asbestos laden work clothes. Some 70% of the mining towns' female residents have had such household-contact exposure [Siemiatycki, 1982]; this was confirmed in our survey (Chapter B.4.). Past exposure intensities were assessed separately for neighbourhood and household-contact exposures. Both types of exposures were estimated using indirect evidence and expert opinion because no direct exposure measurements were available before 1972 whereas the relevant period exposure for the study base spanned roughly from the beginning of the Quebec asbestos industry in 1876 to 1980. After attempting to inventory all available direct or indirect exposure data, these were collected and analyzed, and then were submitted to a panel of five eminent experts in asbestos measurement. The panel examined the data and analyses during two days and then synthesized it by providing their own estimates of past exposure intensities for different mining towns and years. Finally, a survey of female residents of the asbestos mining areas was conducted in 1989 to estimate the residential history

(years and places) of the study base, and the cumulative exposure of the study base was estimated by combining the exposure intensity estimates with the residential histories.

The Exposure Assessment is thus divided in four chapters. Neighbourhood exposure data and analyses are presented first (B.1), household-contact exposures second (B.2), the panel's assessment of past exposure levels comes third (B.3), and finally the exposure intensity estimates are coupled to the population's residential history to assess its cumulative exposure (B.4).

B.1. PAST NEIGHBOURHOOD OUTDOOR EXPOSURE LEVELS

Residents of the asbestos mining towns have long seen fallouts of dust clouds emitted by tailings piles, dryers' stacks and mills' louvers. Anecdotes about past visible asbestos dusts were very consistent among scores of long-time residents whom we interviewed. Many referred to the common saying: "It snows in Thetford in the middle of July". In the book that he wrote on the history of Black Lake, Dr. Clément Fortier, MD., former director of the Hôpital Général de la Région de l'Amiante, recalls (p. 149) [Fortier, 1983a] the following images of asbestos fallouts in the 1940s:

> "... after a day of westerly winds, dust would literally "fall" on the town. In less than 30 minutes, Brylcream and Wave-Set lost their shine and heads turned white starched by the night. (...) In the morning, the observer could track and identify the footprints of those pious persons that he would soon see in church. Back home, he was greeted by his mother sweeping the front porch to prevent the asbestos dust from spreading over the house's linoleum floors. Dusting the furniture daily was bothersome enough".

Such are the kind of outdoor exposures that the neighbourhood exposure assessment seeks to quantify. Since there was very little direct quantitative evidence to historic concentrations of asbestos fibres, various qualitative and quantitative approaches were used. To understand the exposure setting and circumstances (Section B.1.1), historical maps were drawn of the location of emission sources relative to the residential areas back to 1920, and the history of production and dust controls was documented. Five different methods were used to estimate past exposure intensities. Engineering data (production and dust controls) and EPA emission factors for various emission sources were used to compute the relative dustiness at different eras (Section B.1.2). Dusts retained by modern baghouses[1] provided information about the dust that used to escape into the general environment. Their discharge rate was measured and physical and chemical aspect were characterized to evaluate past asbestos emissions and concentrations using an EPA aerosol dispersion model (Section B.1.3). Consistent anecdotes

[1] A baghouse is an immense room with many huge ventilators which aspire a plant's (dryer or mill) dust-loaded air through thousands of filter bags; the air stream is forced through the bags which retain the dust particles. Then the cleaned air is expelled outside through the dryer's stacks or through the mill's louvers.

and recalls about past visible dustiness can provide some insight into the chronological and spatial patterns and the intensity of heavy asbestos pollution. A survey of long-time residents was carried out to elicit such recall data (Section B.1.4). One of the few traces of past asbestos exposure is the amount of asbestos found in lung tissues at autopsy. Asbestos fibre lung burdens of asbestos workers and local residents had been previously estimated [Case and Sébastien, 1987; Case and Sébastien, 1988; Case and Sébastien, 1989]. We analyzed these data by nonlinear regression with a biokinetic model to derive lifetime average exposure levels (Section B.1.5). The most direct data were actual past environmental dust and asbestos measurements in the towns. However, these only went back to the early 1970s. To project these figures further back in time, we regressed the 1972-1984 air pollution levels by town and year on concurrent asbestos production levels and dust control information, and we applied the estimated regression function to historical yearly production levels by town to project past exposure levels by town and year (Section B.1.6).

B.1.1. Historical, Geographical and Technological Setting

Environmental air pollution by asbestos mines and mills depends on production level, dust emission sources in the production process, efficiency of dust controls to counter these emissions, and diffusion and exposure processes between emission sources and the population. These asbestos pollution factors and processes are described in the following order: a) the spatial relationship between asbestos emission sources and the exposed study population, b) historical trends in production, c) production process and emission points, d) types of dust controls, e) effects of dust controls, and f) characteristics of dusts that used to be emitted in the atmosphere in the past.

a) Spatial Relationship Between Emission Sources and the Populations

i. The Asbestos-Mining Area[1]

The Eastern Townships' serpentine mineral deposits spread over a 120 km long by 16 km wide corridor between Danville (160 km east of Montreal) and East Broughton (230 km east-north-east of Montreal) in the Appalachian plateau, about 70 km north of the Canada-U.S. border. In this area, chrysotile asbestos appeared as mountain masses 200-300 m above the surrounding terrain before these hills were transformed into deep mining pits.

[1] This section on asbestos mineral deposits is largely based on pages 29-33 of a 1952 book on The Asbestos Industry of the U.S.A. Bureau of Mines [Turusov and Montesano, 1983], and on J.G. Ross's Chrysotile Asbestos in Canada [Weinberg et al., 1995].

From 1891 to 1957, Quebec's Eastern Townships supplied most of the world's commercial asbestos and constituted the world's largest asbestos mining and exporting region until 1974. There have been six asbestos production centres in the eastern Townships since the beginnings of commercial production. The relative importance of each centre can be gauged from the estimated cumulative asbestos production volumes from 1900 to 1984[1]: Asbestos and adjacent Norbestos produced 20,000,000 tons (36% of Quebec's cumulative production), Vimy Ridge 5,000,000 tons (9%), Black Lake 13,000,000 tons (23%), Thetford Mines 13,000,000 tons (23%), Robertsonville 3,000,000 tons (5%), and East Broughton and Tring-Jonction 2,000,000 tons (4%). Robertsonville, Vimy Ridge, East Broughton and Tring Jonction were excluded from the study because, as stated above, they were not part of an "agglomeration" defined by the OPDQ (Office de planification et de développement du Québec); they represented 18% of the historical asbestos production and about 15% of the person-years in the study base. However, the asbestos production in Robertsonville and Vimy Ridge affected the air quality of Black Lake and Thetford Mines since Vimy Ridge was adjacent to Black Lake, Robertsonville adjacent to Thetford Mines, and moreover Black Lake and Thetford Mines were adjacent municipalities.

Over the last hundred years, asbestos production was the main economic activity in Thetford Mines and virtually the sole industrial activity in Asbestos and in Black Lake. The economic and urban development of the asbestos mining areas and even the landscape were conditioned by the development of the asbestos industry until the 1980s. Before the commercial exploitation of asbestos, these areas were scarcely inhabited; the population worked on poor farms or for a few wood mills and wood-cutting operations. However, the introduction of an asbestos mining operation would dramatically spur population growth. For instance, Thetford Mines, which was then named Kingsville, grew from about 150 in 1879, to 2,200 in 1891 and 7,000 in 1910. Migratory waves in the region closely followed the economic fluctuations of the asbestos industry with in-migration almost always exceeding out-migration. Every

[1] The municipalities are listed from west to east. The tonnage in parentheses is the cumulative production from 1900 to 1984 estimated for each mining municipality on the basis of individual mines' yearly production capacities and yearly production volumes for the Province of Quebec.

asbestos hill which was mined became an open mining pit sided with man-made hills of overburden, refuse and tailings. All asbestos mines were open pits before 1930; since then, only three mines (Bell, King-Beaver, Johnson) in Thetford Mines ever developed some underground mining. Residential neighbourhoods were periodically relocated to make way for the expansion of the mine pits.

Most of the secondary and tertiary economic activities in these towns revolved around the asbestos industry. Agriculture developed on a small scale in surrounding areas to supply the asbestos towns with food. Forestry evolved independently of asbestos mining towns and at increasing distance from these towns. In the Thetford and Black Lake area, the only industrial activity that could have been a significant source of pollution other than asbestos mining and milling was *chromite mining*. Chromite mining was active between 1890 and 1914 and during the two World Wars, and produced a cumulative output of 250,000 tons. The brown and red colour of pulverized chromite on the lakeside of Caribou Lake still caused the brownish colouring of Black Lake's drinking water supply in the 1980s [Fortier, 1983a]. The only other mining activity has been some production of talc (a serpentine mineral) and steatite (soapstone) in Robertsonville and East Broughton.

Maps representing the town of Asbestos in 1925, 1934, 1953, 1971 and 1985, and the towns of Black Lake and Thetford Mines in 1944, 1954, 1971 and 1983 are presented in Appendix B2. The maps show relief, wind rose, mining pits, tailings piles, residential and commercial areas, and neighbourhoods. The latter were identified from panel interviews of local residents and from our residential history survey.

ii. The Municipality of Asbestos

The Jeffrey Mine started to produce in 1881 and is the only mine ever exploited in Asbestos. The mine soon became the largest asbestos open mining pit in the Western world. This affected the population of Asbestos directly. The town as it was in 1934 had been entirely relocated by 1974, when the pit's diameter reached 1.5 km. Up to 1970, rock debris and

boulders from excavation explosions would occasionally fall on the lawns or roofs of residences located on the edge of the pit; up to 1975, a few landslides of the pit's wall occurred in residential areas.

Table B-1 shows distances between residences and emission sources in different years, as estimated from the maps in Appendix B2. The residential area has expanded away from the mining/milling operations since 1925. This expansion probably attenuated the asbestos pollution for most of the population since fallouts decrease as distance from the source increases. Moreover, as the pit got deeper, less emissions would surmount the pit's walls; this must have further reduced the influence of the pit on air pollution relative to the other emission sources. The tailings piles from the Asbestos operation were located at a distance from the residential areas and would have had less polluting effects than the mill and dryer for that reason. Due to the orientation and expansion of the town, the influence of the dominant westerly winds was reduced over time. Throughout the history of the town of Asbestos, it seems that more than half of the town resided downwind from the asbestos emission sources relative to the dominant winds. However, residential areas expanded north-eastward while the mill and dryer were repeatedly relocated southward, the town becoming less and less directly downwind from the mill and dryer. Relief apparently did not have a substantial effect on the population's exposure. (Maps in Appendix B2.)

Table B-1 Distances (km) Between Residences and Asbestos Emission Sources in the Town of Asbestos, 1925-1985

Residential Perimeter [1] and Emission Source	Year				
	1925	1934	1953	1971	1985
Most [2] Dwellings vs. Mining Pit	0.75	1.25	1.80	2.75	3.40
Most Dwellings vs. Mill and Dryer	0.60	0.75	1.00	1.50	2.40
Most Dwellings vs. Tailings piles	1.50	2.20	3.00	3.00	5.00

Table B-2 Distance (km) Between Residences and Asbestos Emission Sources in the Towns of Thetford Mines and Black Lake, 1944-1983

Residential Perimeter [3] and Emission Source	Thetford Mines		Black Lake	
	1944	1983	1944	1983
Most Dwellings vs. Mining Pit	2.00	3.00	2.00	2.00
Most Dwellings vs. Mill and Dryer	1.00	2.00	1.00	1.30
Most Dwellings vs. Tailings piles	0.80	1.50	0.80	1.20

1 The area of the town of Asbestos was 13 km^2 with a residential area of roughly 6 km^2 in 1974.

2 "Most" was defined by the radius from the source - as measured with a ruler on a map - comprising roughly 90% of the residential area. The distance from tailings was based on our estimation of the point of discharge of floats (fresh dust readily airborne) on the piles.

3 The area of the town of Thetford Mines was 23 km^2 with a residential area of roughly 16 km^2 in 1974. The area of the town of Black Lake was 41 km^2 with a residential area of roughly 5 km^2.

The successive mills and dryers were built much higher over time, and so the louvers' and stacks' emission points became higher as well, reducing the fallouts on the town.

The town of Asbestos also had an asbestos products manufacturing plant from 1924 to the 1970s. The plant produced asbestos textile products, felts, protection clothes, pipe and heating insulation, tiles, brake pads and linings, roof shingles, gaskets, etc. Its contribution to the town's dustiness was deemed negligible because its production process was not air intensive; rather, asbestos fibres were not air driven but glued, compressed, wetted, and locked in other materials. For the same reason, the imported crocidolite incorporated in some products could likely have caused significant exposure to the workers but most unlikely to the non-occupationally exposed population[1].

To summarize, in the past, residents of the town of Asbestos lived closer to and more downwind from the asbestos emission sources. Louvers of asbestos mills and dryer stacks were not as high as today. Overall, for the same amount of dust emissions, the population of Asbestos must have been more directly exposed in the past than today.

iii. Thetford Mines and Black Lake

As indicated by the approximate distances in Table B-2, the population shifted away from asbestos emission sources in Thetford Mines between 1944 and 1983. On the contrary, the urban expansion westward and northward development of Black Lake was offset by the development of new and existing mines and by the mushrooming of gigantic tailings piles all around the town and in its very centre.

The main characteristics of this area can be summarized as follows. 1) Thetford Mines, Black Lake, Robertsonville and Vimy-Ridge were all located in the Bécancour River valley, in a corridor 100-200 m deep whose orientation along the dominant southwesterly and

[1] In 46 autopsies from the population of Asbestos, crocidolite fiber was identified in lung tissue from 15 of 23 miners and millers but in only 1 of 23 environmentally exposed persons [Case and Sébastien, 1987].

northeasterly winds must have favoured a back and forth dust exchange between the asbestos mining towns in the corridor. 2) The greater number and spread of emission sources should tend to make emissions more pervasive and exposure more ubiquitous, uniform and constant, and also more difficult to control. 3) contrary to the population of Asbestos, those of Thetford Mines and Black Lake lived much closer to one or other of the numerous high tailings piles. The best known example is Smith Street which was located in a 200 m "valley" between two immense tailings piles (Photograph in Appendix B9, Figure B9-2). 4) Black Lake is the only asbestos mining town where the distance between emission sources and populations did not increase over time. 5) The older, smaller and more numerous asbestos operations in the Thetford-Black Lake area were slower to implement baghouses on mills and dryers than was Johns-Manville in Asbestos.

b) Production Trends

Asbestos production level in the asbestos mining regions is the driving force behind asbestos pollution in these areas. Its time trend graphed in Fig. 1.1.b should give some indication of asbestos exposure trends in the asbestos-mining towns.

Annual untransformed production of asbestos fibres increased from 27,000 tons in 1900 to 278,000 tons in 1929 with an average growth of 8.4 % per year over those 29 years. After the Great Depression, the industry grew at a rate of 9.7% per year from a low of 112,000 tons in 1932 to 859,000 tons in 1951. It remained at that level until 1959, and then grew at a slower pace of 3 % per year until 1974, when it reached the peak level of 1,405,000 tons. A 6-month labor strike in 1975 cut the yearly output by 35%. Thereafter, the asbestos market fell 60% to 530,000 tons in 1988. This dramatic decline was largely due to political, social and health issues, as several industrialized countries introduced measures to regulate and restrict the uses and importation of asbestos. In the past decade however, growing demand for asbestos-cement products in developing countries has spurred a renewed growth in industrial asbestos output.

Trends in usage and uses of asbestos products in industrialized areas of the world roughly paralleled this production curve; for instance, uses of asbestos fibres increased from approximately 400 applications in 1950 to 4000 in 1970. Ambient air pollution by asbestos fibres in most non-asbestos-producing areas must have followed roughly the same trend, albeit with a certain lag due to the extant asbestos products. However, in Quebec's asbestos mining areas, airborne asbestos levels must have followed the production curve very closely until the introduction of significant emission controls in the 1950s.

Table B-3 shows the asbestos production volumes by asbestos mining town over time. Because production volume data were available by company only and because an individual company could comprise mines and mills in different towns, some town-year-specific data were based on partitioning individual companies' production volumes according to the known production capacity of its individual mills.

Figure B-1 Trend in Asbestos Fibre Production in Quebec, 1900-1986

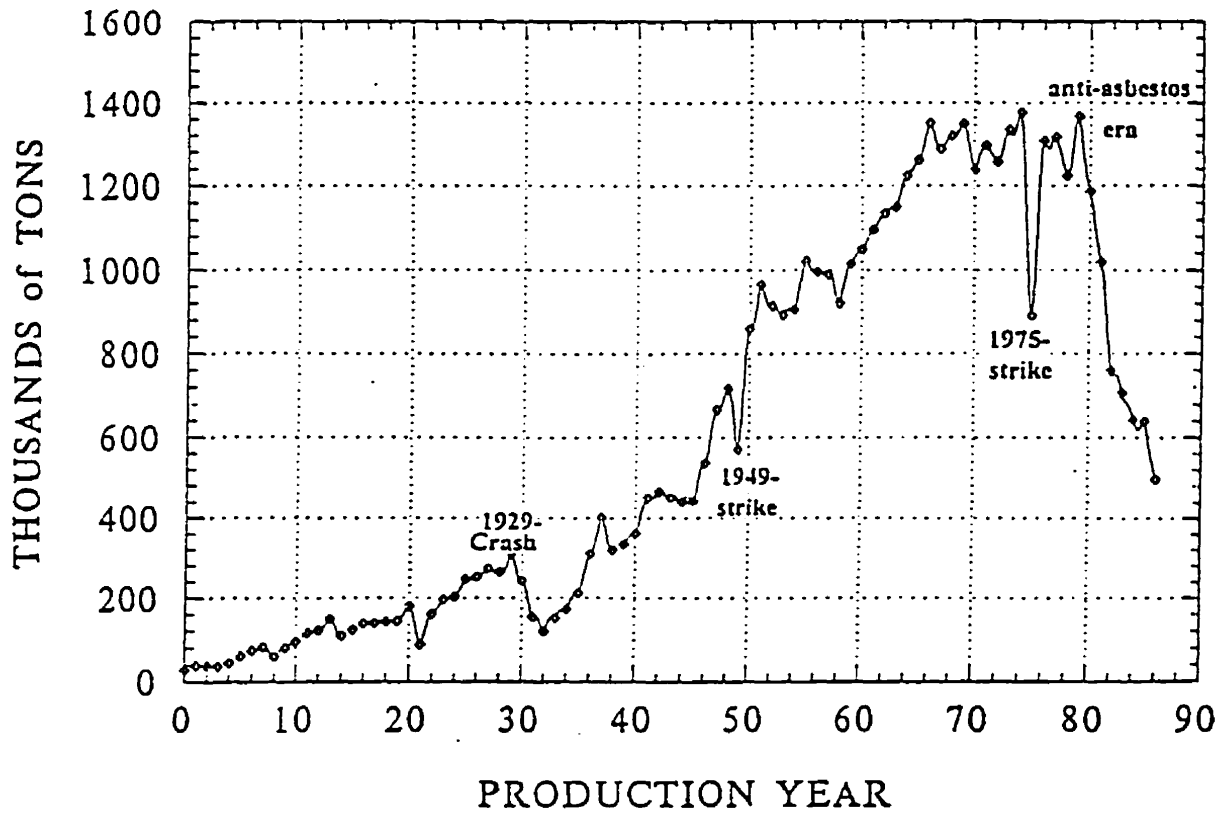


Table B-3 Asbestos Fibre Production in Certain key Years, by Town

	('000 tons/year)		
YEAR	Asbestos	Thetford[1]	Black Lake[2]
1920	10	**50**	20
1930	50	**120**	60
1945	120	**220**	110
1960	**410**	260	330
1974	**600**	300	480
1984	**300**	110	230

1. Includes adjacent Robertsonville, north-east of Thetford Mines
2. Includes ex-Vimy Ridge, south-west of Black Lake

Before 1950, Thetford Mines was the largest asbestos mining/milling centre. After 1950, Asbestos became the most important centre, followed by Black Lake. From 1945 to 1974, the asbestos fibre production output in Asbestos increased 5-fold and Black Lake's 4-fold, but Thetford Mines' increased by less than 40%. At the end of the 1970s, the asbestos slump hit Thetford Mines hardest. (For annual production by town, see Appendix B3)

In Quebec's asbestos mining areas, mines supply the ore to fiberizing mills which extract and classify the fibres into several grades. The fibres are then shipped to consumer industries for use in manufacturing and in construction. Very little secondary transformation of asbestos has ever been done, except for a manufacturing plant which operated in Asbestos. Some asbestos was transformed in the Montreal area (e.g. textile, friction materials, sealants, asbestos cement products, floor tile, etc.) and in Kingsley Falls. Some asbestos was used for shipbuilding in the towns of Sorel and Tracy, yet about 95% of the production was exported abroad to head offices (e.g. Johns-Manville in the U.S.A.) of Canadian operations and to foreign consumer industries.

c) Production Process and Emission Sources

The flowchart in Figure B-2 represents a typical asbestos mining-milling operation. The chart and most of the following information were taken from a 1973 report to the Air Pollution Control Directorate of Environment Canada [J.F. MacLaren Ltd., 1973], from a 1977 document and survey produced by the Air Pollution Control Directorate [Gagan, 1977] , and from a technical course manual produced in 1986 by the Asbestos Region College [Nadeau, 1986].

Figure B-2 Flowchart of a Typical Asbestos Mine and Mill - Part 1: Schematic of Operations

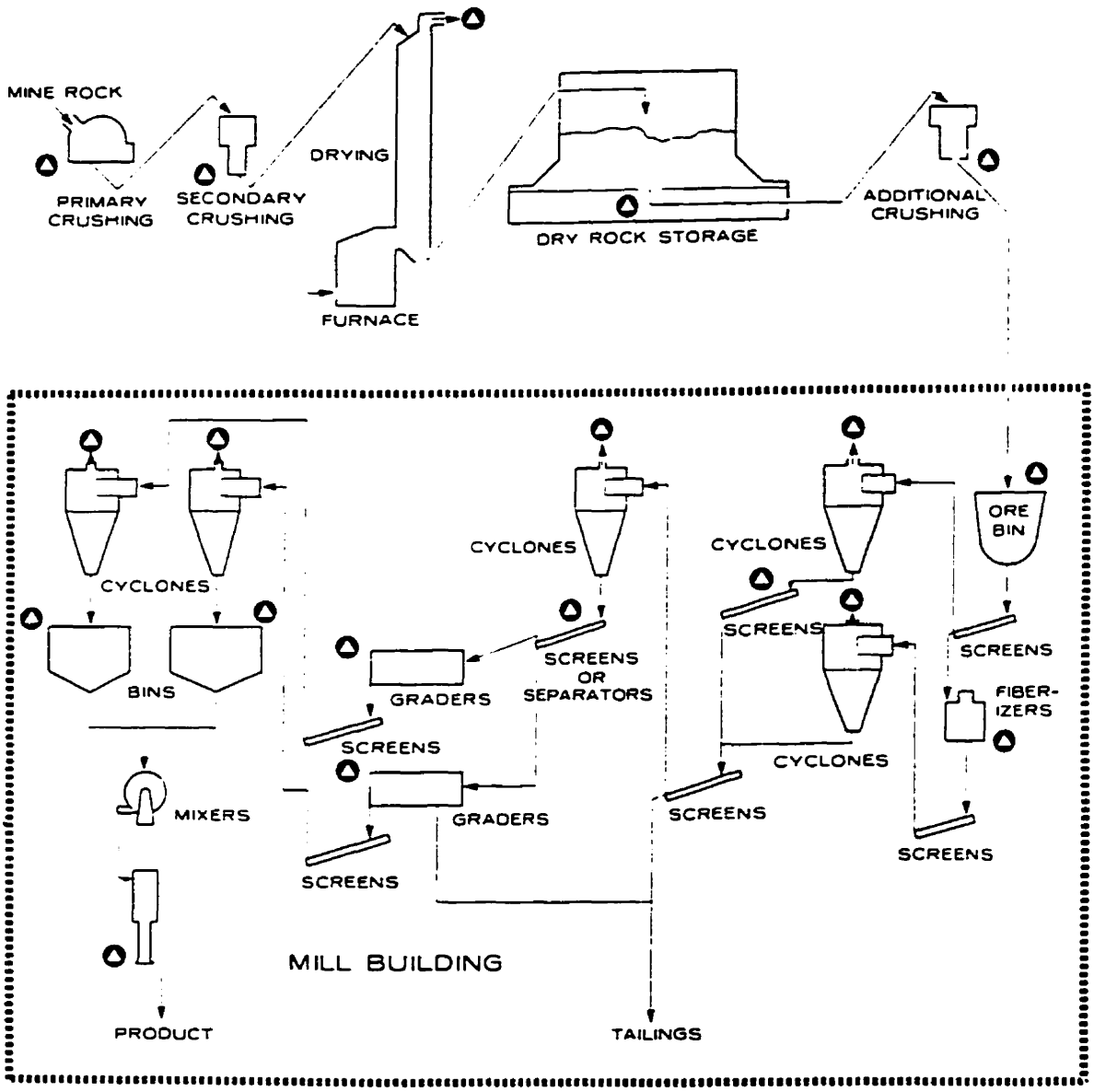
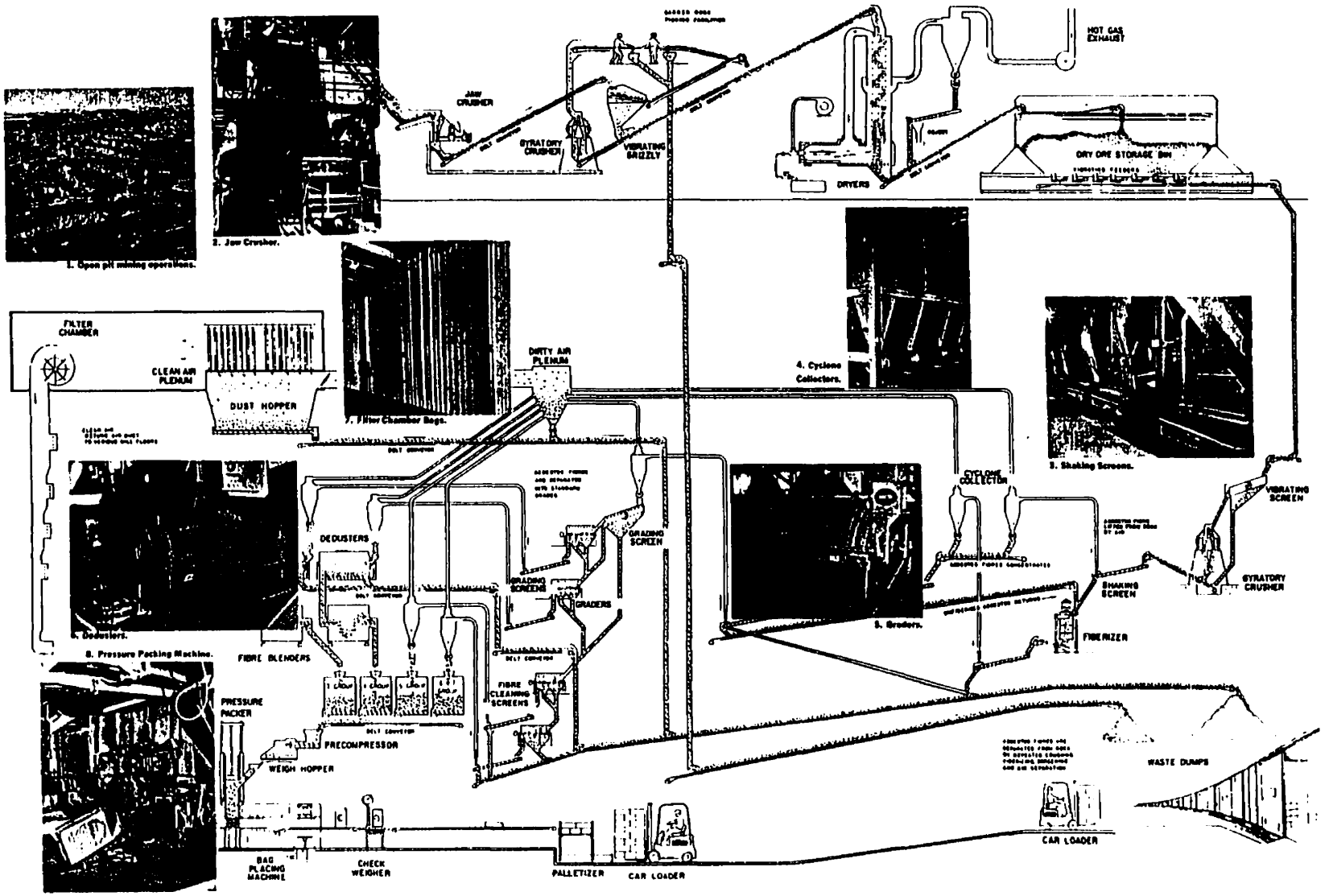


Figure B-3 Flowchart of a Typical Asbestos Mine and Mill - Part 2: Illustrative Diagram



Step #1: 95% of the asbestos mining in Quebec is carried out in open-pit operations. Asbestos ore is removed from the pit's wall by blasting and drilling. Overburden and waste rock is directly hauled to waste dumps. Mining dust emissions have a low asbestos content, they are coarse and fall at very short distance from the source, and their diffusion is reduced by the depth of the mines. This emission source was considered negligible relative to other sources.

Step #2: The broken ore is loaded by large shovels into giant trucks - rail-wagons in the past - to be hauled to crushers where the ore is reduced to about 7 cm in size. Although this operation is executed on the main ground level and may generate somewhat more airborne dust than mining, this source of asbestos emissions was considered negligible for about the same reasons as for mining.

Step #3: A conveyor hauls the reduced ore to the dryer. The ore is dried in fluid-bed, vertical or more commonly rotary dryers. The fuel used is coal, oil or natural gas. Dryers are equipped with stacks to evacuate the hot gases; these gases transport small particles and fibres released from the ore. Due to the height of stacks and the hot temperature of emitted gases, dust emissions fall relatively far from the source. Stack emissions were a major cause of concern for the citizens of the asbestos mining towns, in part due to the visible cloud that they shed over the towns. The most important pollution factor is that dryers use some 10-20 tons of air per ton of final asbestos fibre produced by the mill. These emissions and their fallouts were evaluated in the exposure assessment.

Step #4: The dried ore is delivered to the dry-rock storage building where it finishes drying. Past emissions from this building were due to the air convection through louvers and roof vents caused by the warm ore and were visible. Still, the emission rate and volume of this source were extremely small in comparison to those of dryers and mills. These trivial emissions were not included in the exposure assessment.

Step #5: The dried ore is conveyed to the mill where it enters the "rock circuit" to be repeatedly crushed and screened, releasing fibres at each stage. When the rock has been reduced to less

than 0.5 cm, it is sent to the tailings piles. The asbestos fibres freed at various steps of the process are lifted by air suction and aspirated through hoods to cyclone collectors[1] which direct the fibre into a "fibre circuit". After removal from the air stream, the fibres are passed over cleaning screens and are aspirated again. The undersized dusts called "floats" are discarded to the tailings. After running through fiberizers, rotary trommel dusters and cleaning screens, the fibre is separated by different grading methods, aspirated and collected for bagging.

According to the engineers[2] and ventilation specialist[3] interviewed, asbestos *mills* use about 100 tons of air[4] per ton of asbestos fibre produced [Lebel, 1984]. 90% of this air is used to separate, collect and grade the fibres. At the end of the fibre separation process, the air exhaust and the small floats which bypassed the cyclones are discharged in a large chamber on the last story of the mill. In the past, this chamber was a "float shed" where aerosol floats decanted; it was divided into several compartments at varying distances from the air inlet so as to capture floats of different size ranges [Ross, 1972, p.78]. In the 1950s and 1960s, some plants completed the float chamber with electric precipitators or with scrubbers. Today, float sheds, precipitators and scrubbers have been replaced by extremely efficient baghouses.

Milling was considered as the main source of asbestos pollution in asbestos-mining towns in the exposure assessment. Indeed, the air volume, the asbestos content of emissions and the proportion of smaller respirable fibres are much higher in milling than in any other process. In addition, mills' emissions exit at a much lower temperature and at a lower height (1/2-2/3) than dryers' emissions, so that mills tend to pollute closer neighbourhoods whereas dryers tend to

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1 A cyclone is an open-ended conic cylinder using centrifugal force to separate materials according to their mass or aerodynamic equivalent diameter. A fast dust loaded air stream enters tangentially near the top of the cyclone, centrifugal force precipitates heavier aerosols against the surface and down through a hole at the bottom of the cyclone while the air stream and the lightest particles continue their path through the top of the cyclone. Cyclones are the most widespread dust separators used in industrial plants.

2 E. Thibodeau from J-M Asbestos, T. Coleman, L. Michel and G. Dufresne from LAB-Chrysotile, and J. Lebel from the Asbestos Institute.

3 R. Vaillancourt from LAB-Chrysotile.

4 To save on heating fuel or electricity, part of the air (50-90%) is recirculated during very cold days (about 50 days per year) so that the air output is about 2-10 times smaller than usual on these days [Vaillancourt, Thibodeau, 1991, personal communications]. The recirculation rate depends on the outdoor temperature. Recirculation rates might have been higher before the industry started to reduce dust levels in the workplace; however, the incentive to recirculate was probably lower when energy was relatively cheap, and the absence of baghouses probably prevented air recirculation before the 1950s.

pollute farther neighbourhoods. Finally, since mining towns were much smaller and closer to the mills in the past than today, the impact of mills on air pollution would have been even greater than today relative to the impact of the dryer.[1]

Step #6: Refuse from various points in the milling circuit is transported by belt conveyors to 100 to 180-meter high piles[2] . Before 1975, dusts were blown right off the conveyors which were not enclosed at the time. But the most important pollution problem was the 2400 rpm high-speed flinger at the top of the piles. During dry weather, strong winds could generate dust clouds 500 to 3000-meter long. Even though emissions from tailings piles were visible and substantial[3] before 1975, they were not usually as dramatic, and their volume and the distance of their fallouts was irregular in comparison to the constant huge air flows carrying out drying and particularly milling emissions. Hardly any data was available to quantify these emissions, so they were only roughly assessed in Section B.1.2.

i. Past Production Equipment and Processes May Have Polluted More

Although the basic asbestos mining process and the air-intensive fibre extraction process have remained fundamentally unchanged over the last 80 years in Quebec, the production process *per se* might have generated more dust emissions in the past, independently of dust controls. In the past, cyclone collectors used to clog and break frequently, sending huge amounts of dust in the ambient air, inside and outside the plant. Since the 1950s, better maintenance,

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[1] The greater importance of the milling emissions may not be fully in accord with recollections by many residents and visitors who have been impressed by visible emissions from dryers and from tailings piles up until the early 1970s. Dryer and tailings emissions probably appeared more dense than they actually were: hot gas made dryer emissions more visible due to condensation in colder weather, and tailings piles emitted a large proportion of aerosols which were either not respirable or not asbestos fibres. Visible dustiness must have correlated better with total dust emissions than with respirable asbestos emissions.

[2] These piles were high enough to attract gliders and even skiers in winter. Over the last decades, despite warnings and watch guards, avalanches have taken the lives of about 10 young trespassing skiers or gliders between 1970 and 1995 [Findley et al., 1984]. In the past, even in the summer season, the tailings piles were tempting playgrounds, mostly for kids whose backyards were invaded by the bottom of the piles such as on Smith Street

[3] An industrial hygienist who used to work for the asbestos industry (J. Lebel) witnessed in the spring of 1974, the upset of a sugar-loaf party 2 km downwind from Carey's active tailings pile. Strong winds had brought down enough asbestos dust, maybe 15 f/cc, from the pile to force the guests to leave. This hygienist told us that visible dust clouds from tailings piles were a normal thing and occurred about every week.

introduction of automatic bypass and shutdown systems, the addition of an internal ceramic coating on cyclones to reduce breakages[1], and other improvements likely reduced industrial asbestos emissions materially. In addition, in 1939, asbestos mills[2] began to recuperate for new markets "short" and "float" grades of asbestos fibres which until then were sent directly to the dump or in the atmosphere through the mills' louvers. The effect of these measures could not be estimated however and were not included in the exposure assessment.

Estimations (see Appendix B4) of asbestos emissions computed by engineers [J.F. MacLaren Ltd., 1973; Gagan, 1975; Gagan, 1977] for the years 1972-74 corroborate quantitatively the above ranking of different emission sources as to their environmental impact.

d) Dust Emission Controls

Since 1912, citizens have complained officially of many annoyances [Cinq-Mars et al., 1994, p.201] due to asbestos emissions from the mills and tailings piles: "snow falls" at dusk, the need to continuously broom houses' entrances and balconies, daily dedusting of furniture, dust on the laundry on the hanging on outdoor clothes lines, dirty or even clogged mosquito screens, gray lawns, etc. To respond to these complaints, a few dust emission controls were implemented in asbestos mills during the first half of the century: e.g. a system developed by T. Lafrance in 1922 [Fortier, 1983a, p.108] and jute bag curtains installed in the 1940s [L. Piuze, 1989, personal communication]. Some primitive "electric magnet system" [Cinq-Mars et al., 1994, p.202] was tried; this may have been some kind of electrostatic precipitators. While these attempts may have stopped some of the heaviest floats (>50 µm

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[1] Before the 1950s, cyclone collectors used to clog and to break very frequently, sending huge amounts of dust in the ambient air, inside and outside the plant, through louvers, vents and open windows.

[2] Bell Asbestos in Thetford Mines was the first operation to market asbestos floats.

D_{ae})[1], the small respirable floats (<10 µm D_{ae}) would not have been abated by such gross techniques [Ross, 1972; Bisson, 1986].

From 1948 to 1975, milling emissions were greatly abated with the installation of baghouses[2] which could filter out dust particles of less than 50 µm D_{ae} [Ross, 1972]. Baghouses reduce the mass concentration of asbestos bypassing the last cyclone by at least 99.9% [Ross, 1972; R. Vaillancourt, LAB-Chrysotile, 1991, personal communication], although their efficiency for the small respirable fraction of the dust is less certain, depending not their maintenance. The improvement was remarkable, as shown in Figure B-4 which contrasts the visibility of emissions "before" and "after" installation of a baghouse at British Canadian's operations in 1948. In 1974, many baghouses did not have sufficient capacity nor appropriate maintenance until the 1974-1984 period, when government imposed emission standards, measured the emissions and recommended or imposed solutions to delinquent operators.

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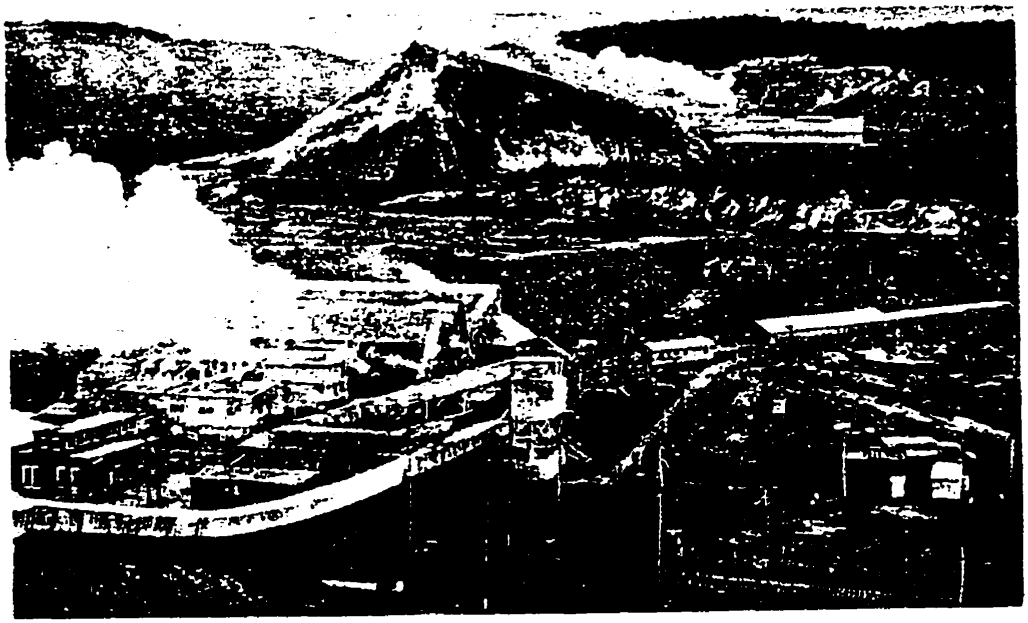
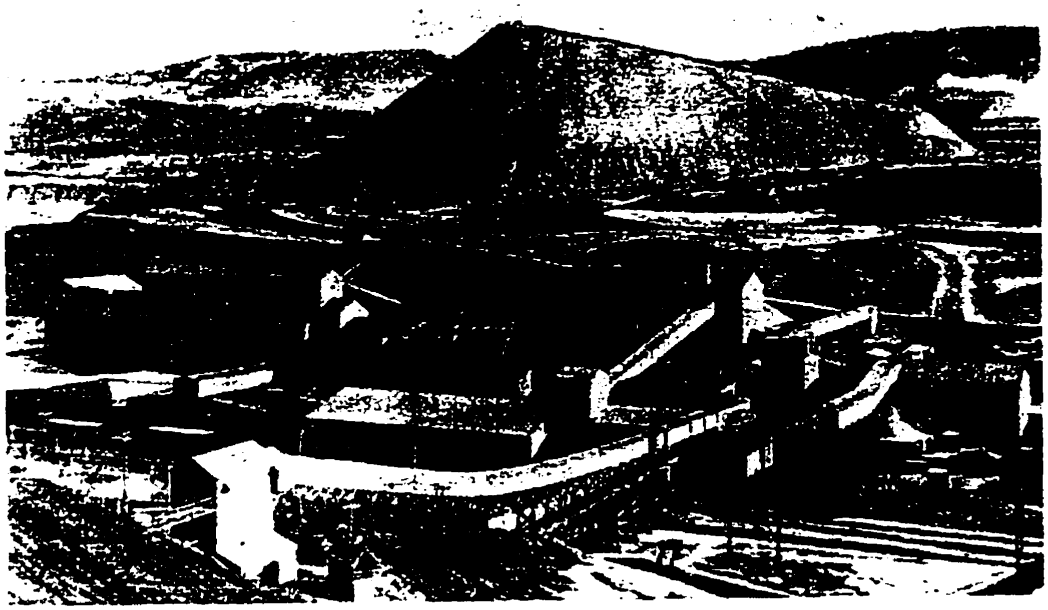
[1] D_{ae} is the aerodynamic equivalent Diameter of a particle and depends on the particle's size, density and shape. The µm value stated for filtration systems in the present and the next paragraphs represent the minimum particle size for which at least 90% of the particles will be removed by filtration.

[2] Dust-loaded air enters bag filters and is aspirated through the bags to a negative pressure aspirating room. Filters have pores as large as 100 µm, but a "cake" of dust covers the interior of the bags and clogs the pores, letting only particles smaller than 10 µm pass through. Today, baghouses reduce the concentration of respirable fibres longer than 5 µm to about 0.1-0.2 f/mL in the baghouse.

Figure B-4 Pollution, Depollution: British Canadian Asbestos (Black Lake) Before and After 1948[1]



Pollution
British Canadian Asbestos
Dépollution



[1] Reproduced with the permission of the author and editor[Fortier, 1983a].

For technological reasons, the implementation of baghouses on dryers lagged that on mills by about two decades. Dryer emissions were filtered by cyclone collectors to recuperate the larger useful airborne material; they had little efficiency in retaining particles smaller than 25 μm D_{ae}. In Asbestos, the dryer was equipped with a baghouse in the 1950s. In the other asbestos-mining towns, electrostatic precipitators and scrubbers - which proved to be inefficient with asbestos dust - were added to the cyclone collectors on most dryers in the 1960s and 1970s; only a few dryers were equipped with baghouses. In the 1970s however, filtration bags were developed which could sustain the high temperatures and acid conditions of dryers and the Provincial and Federal governments pushed for efficient asbestos dust controls. Therefore most dryers were equipped with baghouses in the 1970s, particularly in 1974 when a strike gave the opportunity to operators to change their equipment without stopping production.

Even after baghouses were implemented on mills or dryers, dust controls continued to improve. According to the above mentioned engineers and ventilation specialist (personal communication), baghouse technology, efficiency and particularly maintenance improved significantly over the 1950-1985 period, particularly after 1974.

Wet mining, closed conveyor belts and appropriate dust controls on tailings disposal processes came into widespread use only in the late 1970s and had some impact on dust reduction, although this impact has not been evaluated.

Table B-4 summarizes the history of implementation of dust emission controls. All these improvements in dust emission controls contributed to abate pollution levels, albeit in different proportions according to the production process involved.

Table B-4 Outline of Principal Methods of Emission Control for Different Emission Sources, in Different Eras, in Quebec's Mining Industry

Emission Source	Era		
	Before 1945	1950s and 1960s	1970s and 1980s
Mining	• Nothing	• Wetting rocks and roads • First baghouses on drills	• Wet excavating • Baghouses on machinery
Drying	• Cyclone collectors	• Better cyclones • First baghouses • Precipitators or scrubbers	• Baghouses • Improved maintenance of baghouses
Storage	• Nothing	?	• Baghouses
Milling	• Cyclone collectors and settling room	• Better and more cyclones • Baghouses	• Larger baghouses • Improved maintenance of baghouses
Tailings	• Nothing	• Some wetting • Some baghouses	• Pugging • Low speed conveyors replace flingers • Enclosure of conveyors • Baghouses

ii. Maintenance and Efficiency of Emission Controls

Baghouses were essentially implemented on mills between 1948 and 1974 whereas they were implemented on dryers between 1974 and 1980. The question arises as to whether or not the resulting abatements in dust emissions were proportional to the importance of the emission sources and to the nominal efficiency of the dust control systems. Likewise, if uncontrolled dust emissions were 'X' times larger from mills than from dryers, would the dust abatement on mills before 1974 be 'X' times larger than the measurable (available dust concentration measurements from 1974 on) post-1974 dust abatement largely attributable to dryer emission controls?

The more or less continuous improvement in dust abatement even after the installation of baghouses tells of the importance of *maintenance* on the efficiency of emission controls. Internal reports [Brulotte, 1976; Brulotte, 1980; Boisjoly, 1988] of the Quebec Ministry of Environment on the regulatory compliance of asbestos producers indicated a continuous improvement in regulatory compliance from 81%[1] in 1979 to 95% in 1984 and 100% in the late 1980s[2]. Most infractions were explained by poor maintenance of baghouses such as unreplaced torn bags. Dust control maintenance deficiencies must have been even more important in the 1950s and 1960s in the absence of regulations. Thus the real efficiency of baghouses in the 1950s and 1960s was much lower than their 95-99.5% efficiency ratings. (Some details and pictures from the reports are available in Appendix B5.)

The effect of maintenance failures on dust concentrations at the output of baghouses was corroborated by anecdotes concerning measurements taken before and after repairing torn bags. In 1979, at the National Mine, the perforation of two out of the thousands of bags in the baghouse caused a doubling of the emission rate from 2 f/mL to 4.2 f/mL. In 1988, at another

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[1] Proportion of mandatory samples not exceeding the standard values.

[2] The compliance rates reported here are slightly lower than the official statistics. The latter comprise the extra samples required to correct the detected problems.

operation (confidential information), an emission concentration of 5.5 f/mL measured in one stack was brought down to 0.12 f/mL by replacing the single torn bag. At the same location in 1989, a stack sample measurement of 1.9 f/mL was brought down to 0.2 f/mL only four hours after replacing the single torn bag.

Maintenance appears to be crucial for the efficiency of dust controls and must be taken into account to estimate the impact of the implementation of dust controls. Thus, although baghouses introduced over the 1948-1974 period had a rated efficiency of 98% and would theoretically have reduced asbestos dust emissions from mills by a factor of 50, their real impact must have been much lower due to maintenance insufficiencies until the late 1970s.

e) Conclusion

Production volume increased steeply and almost continuously after the Great Depression up until the late 1970s, when the asbestos industry slumped, falling to pre-1950 production levels in the 1980s. Other determinants of asbestos emissions have modulated significantly the relation between production, emissions and air pollution. Further back in time, the population of the asbestos-mining towns lived closer to and more directly downwind from the asbestos mills, mines and tailings piles; the main exception was the rapid multiplication and rise of gigantic tailings piles in the midst of Black Lake in the 1950s and 1960s. Among the three main asbestos-mining towns, the citizens of Asbestos have always been much less exposed to emissions from tailings piles, whereas the citizens of Black Lake lived closest to all types of emission sources after the 1950s.

Asbestos emissions due to asbestos production were abated dramatically throughout the 1950-1980 period, during which time baghouses and various dust controls were implemented. However, the baghouses approached their nominal efficiency only in the early 1980s, after the implementation of governmental controls and regulations and better maintenance procedures.

Throughout the century, asbestos milling was the determining factor of asbestos pollution in asbestos mining towns, due to the strong concentration of fine asbestos dusts at the output of the process, the extremely large volume of air used and emitted by the process, the nearly ambient air temperature and the intermediate height of the mills' emissions. Accordingly, the installation of baghouses on mills from 1945 to 1975 must have abated asbestos emissions more radically than the installation of baghouses on dryers.

B.1.2. Engineering Estimations Based on Emission Factors

a) Introduction

The evaluation of airborne asbestos concentrations 50 years ago involves many indeterminable interacting factors. This estimation problem can be simplified by making the approximation that outdoor respirable asbestos levels (f/mL) are primarily determined by and should therefore be proportional to the amount of industrial asbestos emissions. Although asbestos concentrations cannot be quantified directly from emission volumes, the proportionality assumption allows using the ratio of past to present emissions in lieu of the ratio of past to present concentrations. This chronological emission ratio could then be applied to documented recent asbestos concentration levels to estimate past concentration levels.

Emission volume can be conceived as the product of production volume by an *emission factor,* this factor being the amount of dust emitted in the atmosphere by unit of throughput of a given activity, process, plant or industry. Emission factors can be estimated algebraically from available engineering data. Since past annual production volumes are known, past emission volumes can be assessed by estimating an emission factor for different eras for the Quebec asbestos mining/milling industry, and by applying these factors to past production volumes.

Two independent estimations of asbestos emissions by Quebec asbestos mining and milling operations were already available, one for the year 1974 and another for the year 1984. As to emissions before the introduction of dust controls, we estimated the emission factor of the mills and dryers before the installation of baghouses and applied this factor to the production

volume in the year 1945. Indeed, fibres that in the past would have been expelled from the mills and dryers into the town environment are today captured in the baghouses. The volume of these fibres and their respirable proportion were determined to estimate emission factors before the introduction of baghouses.

Emission ratios between key years were derived from the asbestos emission estimates for 1945, 1974 and 1984.

b) Materials and Methods

The emission factors used and estimated below are "production-based" in that they represent the number of grams of asbestos dust emitted in outdoor air during a given time interval divided by the tonnage of asbestos fibre produced during the same time interval. They are expressed in units of g/t, i.e. grams per ton.

Gagan [Gagan, 1977] evaluated the volume of asbestos dust emissions for the years 1973 and 1974, applying "process-based"[1] emission factors estimated by the U.S. EPA to process-specific data gathered in a survey of Canadian asbestos mining companies by the Canadian Ministry of Environment.

The emission factors for 1984 were estimated more empirically and directly by Lebel [Lebel, 1984]. The mass of respirable dust emitted by each source was sampled and measured with either *ad hoc* or continuous sampling in every mining/milling operation in Quebec in 1984, when all modern dust controls and maintenance systems were already in operation. Lebel's estimates of emission volumes and emission factors were available by process and by mining operator.

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[1] The U.S. EPA's emission factors were process-specific and pertained to the volume of material handled in a given step of the production process. E.g. the amount of dust airborne as the result of loading and unloading trucks transporting ore material to the crushers.

Pre-baghouse mill and dryer emission factors were estimated on the basis of production data collected over a two-week period in November-December 1990 by the plant managers of three mills and two dryers[1] in the asbestos mining towns. (This evaluation is described in detail in Appendix B6; the Appendix comprises the gravimetric size distribution of the dusts filtered out in the baghouses.) For each plant, the mass of dust retained by the baghouse and sent to the tailings over the two-week period was computed by the managers as was each plant's output volume. The plant managers believed that the ratio of filtered dust to production throughput was a characteristic of each plant that was independent of the sampling period or time of year. Since the amount of dust escaping the baghouses estimated by Lebel in 1984 would be 4 or 5 orders of magnitude smaller, it was deemed negligible; this approximation was also justified by the >99.95% rated efficiency of modern filtration bags. The total volume of dust filtered out by each baghouse was multiplied by the asbestos content proportion estimated by Gagan (31% for mills and 5% for dryers in 1974) to estimate the volume of *asbestos* would-be emissions. The latter quantity was then divided by the tonnage of asbestos fibre produced over the period to compute asbestos emission factors. The average mill and dryer emission factors were applied to the 1945 annual production volume to estimate emissions by mills and dryers before the installation of baghouses.

The *respirable* proportion of these emissions was estimated by analyzing dust samples taken from the baghouses of all four mills and three dryers[2] still operating in Quebec's asbestos mining towns in 1990. A one-kilogram "grab sample" was taken from at the beginning of each shift over the two weeks mentioned above, for an approximate total of 72 kilos per sampling site. A 500 g sub-sample from each sampling site was then separated into six D_{ae} particle size strata by the research division of the Société Nationale de l'Amiante (SNA), using a standard Ro-Tap sifter. For 1984, it was assumed that 100% of the emitted asbestos aerosols were respirable, whereas Gagan's emission estimates for year 1974 were multiplied by a respirable

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[1] One manager responsible for one mill and one dryer never sent us the requested information.

[2] The plants involved were: J-M Asbestos, Bell (no dryer), B-C and Lac d'Amiante, the last three belonging to LAB-Chrysotile.

proportion interpolated linearly between the 1945 ("x"%) and 1984 (100%) estimated proportions as: $100\% - (100\% - x\%) \bullet \frac{(1984 - 1974)}{(1984 - 1945)}$.

The three estimated asbestos emission volumes were multiplied by their respective respirable proportion to deduce the corresponding volumes of respirable asbestos emissions. Past to present *gravimetric*[1] respirable asbestos emission ratios were obtained by dividing the estimated *mass* of emissions in an earlier year by the estimated mass of emissions in a later year: 1945÷1974, 1945÷1984, 1974÷1984. Inference about chronological ratios of asbestos concentrations expressed as *numbers* of fibres per milliliter of air should be based on *numeric*[2] emission ratios. However, the latter were assumed to be reasonably approximated by the gravimetric ratios because the chronological comparisons were restricted to extremely small *respirable* particles whose size distribution would not vary much within the respirable size range. The potential impact of an error in this assumption was estimated theoretically. Assuming a material differential size distribution within the respirable range did not affect the ratio estimates importantly. This simulation is explained and quantified in Appendix B7.

Finally, to determine whether respirable particles emitted before the installation of baghouses were different from those emitted today, the physical aspect and chemical nature of the respirable dust sampled from the baghouses were analyzed by transmission electron microscopy (TEM) and by x-ray energy dispersion (EDXA) by Dr. A. Dufresne (McGill University's Occupational Health Microscopy Laboratory).

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1 Dust quantities measured with respect to their mass.

2 Dust quantities measured with respect to the number of particles.

Table B-5 Gravimetric Asbestos Emissions in Quebec's Asbestos Mining Industry for Years 1984, 1974 and 1945

	Estimated Asbestos Emission Factors (g/t) [1]		
Emission Source	year 1984 [2]	year 1974 [3]	year 1945 [4]
Milling	3	3,500	140,000
Drying	1	380	4,000
Tailings	4 [6]	500	5,000 [5]
Crushing	1	100	2,000 [5]
Storage	> 0 [6]	3	500 [5]
Mining	1	> 3	1,000 [5]
Global Asbestos Emission Factor	> 10	≈ 4,500	≈ 155,000
	Estimated Annual Emission Volumes (tons)		
Global Emissions [7]	> 6.5	≈ 6,000	≈ 69,000
Respirable Asbestos Emissions [8]	> 6.5	≈ 5,400	≈ 42,000

1 g/t: grams of asbestos emitted per ton of fibre produced.

2 Estimated by J. Lebel [Lebel, 1984].

3 Derived from asbestos emission volumes estimated by E.W. Gagan[49]. (Details in Appendix B4)

4 Estimated in 1990 by M. Camus as explained in the Methods. Plant #1 produced 25 t/hr of asbestos fibre and rejected 6 t/hr from the mill and 3.3. t/hr from the dryer; plant #2 produced 15.3 t/hr and rejected 5 t/hr from the mill and 1 t/hr from the dryer; plant #3 produced 13.5 t/hr and rejected 13 t/hr from the mill which used dried ore from another plant. The sum of rejects was divided by the production total to compute *pre-controls emission factors*: *530 kg/t* total, *450 kg/t* from mills, *80 kg/t* from dryers. The mill and dryer *dust* emission factors were multiplied respectively by 30.9% and 5%, the emissions' asbestos content estimated by Gagan in 1974, to obtain *asbestos* emission factors.

5 "Guesstimates" for 1945: no data for tailings, mining, crushing or storage before the introduction of controls. Based on verbal accounts of hygienists and residents, these figures were thought up as upper estimates, 10-300 times larger than the 1974 figures. The estimated global emissions was not sensitive to these sources which were small (≤ 5%) relative to milling+drying.

6 Guesstimates for 1984: tailings' emissions were not measured by Lebel, but complaints by citizens still signaled fugitive visible emissions in the early 1980s, so this source was not negligible.

7 Yearly emission volumes are the product of yearly production volumes by the emission factors.

8 The gravimetric respirable fraction of emitted dusts was estimated as ≈60% for 1945 (Appendix B6), 100% for 1984 with modern baghouses, and an interpolated 90% for 1974. These fractions were applied to Total emission volumes to estimate Respirable emission volumes.

Table B-6 Estimated Asbestos Emission Ratios in Mining Towns

	3 Chronological Comparisons		
Emission Ratio	1945 vs. 1974	1974 vs. 1984	1945 vs. 1984
Gravimetric Emission *Factor* Ratio [1]	≈ 35	< 450	< 16,000
Gravimetric Emission *Volume* Ratio [2]	≈ 12	< 900	< 10,000
Gravimetric *Respirable* Emission Volume Ratio [3]	≈ 8	< 800	< 6,000

1 Between-year ratio of gravimetric emission *factors* from Table B-5.

2 Between-year ratio of gravimetric emission *volumes* from Table B-5.

3 Between-year ratio of gravimetric *respirable* emission volumes from Table B-5.

c) Results

The estimated emission factors and volumes for the years 1945, 1974 and 1984 are presented in Table B-5. The emission factor estimates by emission source indicate that milling was by far the main source of asbestos pollution and that the reductions of mills' emissions essentially determined the improvement of air salubrity in the mining towns. Indeed, from 1945 to 1974 and from 1974 to 1984, reductions in aerosol emissions by the mills would have constituted at least 80% of the gravimetric reductions in asbestos emissions by all sources.

Table B-6 contrasts the three years to one another on a ratio scale. The estimated ratios imply that each ton of asbestos fibre produced before 1945 generated about 35 times more asbestos dust than in 1974, and up to 16,000 times more than in 1984. The estimated mass of asbestos emissions before 1945 was about 12 times higher than in 1974, and up to 10,000 times higher than in 1984. However, the mass of respirable asbestos dust emitted in 1945 was estimated to have been 8 times higher than in 1974, and up to 6,000 times higher than in 1984.

d) Discussion

The asbestos emission volumes and factors estimated for 1945, 1974 and 1984 were obtained in three different studies using three different approaches. It is unlikely that methodological differences could fully account for the large differences in estimated emission factors between key years. However, the estimated reduction in emission factors between 1945 and 1974 was about 13 times smaller than that between 1974 and 1984. Yet the reduction in emissions should have been larger before 1974, when controls centered on mills, than the reduction after 1974, when controls centered on dryers and other sources, since mills emitted 10-30 times

more dust than dryers before the complete implementation of modern baghouses. This paradox may result from non-comparable estimation methods for different years.

The 1945 estimate was based on a reliable direct quantification of dusts retained by baghouses and a straightforward extrapolation to a simple pre-baghouse situation; it could hardly be off by more than a factor of 2 or 3 either way. The 1984 estimate was based on precise and immediately relevant data, but emissions were so small that the slightest error would have been proportionately important. In particular, the slightest oversight would have underestimated emissions significantly. Such a bias was most likely. Indeed, sampling in that study was done under "normal" conditions, which excluded the then frequent intermittent dryer-baghouse bypasses[1] and the still significant occasional bag breakages. Overall, the 1984 estimate could easily underestimate true time-weighted average levels by two- to ten-fold; if this bias were true then the adjusted pre-1974 and post-1974 proportional improvements in emission factors would have been about equal in importance. As for the 1974 estimates, it is difficult to assess the reliability of Gagan's assessment because he did not explain his calculations and assumptions in detail; his estimates were based on an industrial survey and on process-specific emission factors. The uncertainty of the 1974 estimate is probably intermediate between those for 1945 and 1984. Given the large investments in dust controls between 1945 and 1974 and the established visible improvement in outdoor dustiness, I thought that the emission factor must have been at least 12 times lower in 1974 than in 1945.

Adjusting for these likely methodological errors and differences, I would guess that the emission factor in 1945 was more likely 12-100 times higher than in 1974 and 100-2,000 times higher than in 1984, and about 8-20 times higher in 1974 than in 1984. Accordingly, accounting for the different production volumes, respirable asbestos emissions in 1945 would have been 4-30 times higher than in 1974 and those in 1974 40-200 times higher than in 1984.

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[1] Some drying operations used baghouse bypasses to continue production during short breakages, repairs or maintenance operations, but they were mostly automatic preventive bypasses during heating peaks to avoid burning the filtration bags. Dryers' emissions would then go directly to the atmosphere; no stack sample has ever been taken during a bypass, when emission rates might could be 3-4 orders of magnitude larger than normal. In 1984, bypasses were limited by Environment Quebec to 1 hour per month. This could still have been enough to multiply monthly emissions roughly threefold.

The ratio scale is multiplicative and appropriate for asbestos *emissions* from the asbestos industry due to the characteristic proportional or ratio action of dust controls. Indeed, dust controls do not remove a fixed absolute quantity of dust but rather a proportion of the dust load of the controls. Accordingly, ratios can represent how much more asbestos dust was emitted in the past. However, the assumption that asbestos emission ratios can represent ratios of airborne asbestos levels at different time periods does not apply validly to years as recent as 1984. When industrial emissions approach zero, as in 1984, the ambient "background" pollution level due to non industrial sources of asbestos becomes material relative to the contribution by industrial emissions, while it was negligible in the years when industrial emissions were not filtered adequately. This background level must have remained roughly constant throughout the 1945-1984 period.

Hence, emission ratios intrinsically overestimate ratio changes in airborne asbestos concentrations over time. In practice, the overestimation should be negligible for changes between 1945 and 1974 because emissions by the asbestos industry constituted the largest part of outdoor asbestos concentration levels over that period. However, by 1984 the asbestos industry's emissions were so low that their contribution to the global outdoor concentrations may have been lower than background asbestos pollution. Thus emission ratios relative to 1984 may considerably overestimate between-year differences in ambient asbestos concentration levels and should be discarded.

e) Conclusion

Engineering data and calculations obtained from different sources indicate that, in the asbestos mining towns, the number of respirable asbestos aerosols emitted by the asbestos industry in 1945 must have been at least 4 times, most likely 8 times and at the most 30 times higher than in 1974. Due to background asbestos levels, only the 1945-1974 ratio can be extrapolated validly to outdoor asbestos concentrations (f/mL) in the asbestos towns' ambient air.

B.1.3. Aerosol Dispersion Modeling

a) Introduction

Mathematical models have been developed which simulate the dispersion and movement of aerosols emitted into the environment. The U.S. Environmental Protection Agency (EPA) and engineers have been using the "ISC-LT"[1] aerosol dispersion simulation program for many years to evaluate the impacts of stacks, louvers and other point sources of emissions, and even the impact of buildings and other diffuse sources, on long term particulate or gas concentrations in the neighbourhood of these sources. This model is commonly used along with short term simulation models by engineers to determine the height and overall size characteristics of new stacks and emission controls so as to respect the Clean Air Act. The Quebec Ministry of Environment uses these models also to establish and quantify the responsibility of polluters, to plan and improve their air surveillance network and system, and to prevent excessive pollution.

In the present exposure assessment, the ISC-LT simulation model was used to directly estimate asbestos dust concentration levels in the outdoor air of the town of Asbestos around 1945, before the installation of baghouses and other dust controls in any mining town. A second objective was to see what concentration levels the model would predict at various distances and directions from the mill and dryer. A third objective was to estimate the relative contributions of mills and of dryers to asbestos pollution levels in the air of the mining towns.

[1] ISC-LT stands for "Industrial System Complex - Long Term".

The validity of the model was tested by comparing predicted dust levels with dust concentration measurements taken by the Quebec Ministry of Environment in 1972.

The model was not applied to Thetford Mines and Black Lake. Indeed, the model requires over 20 different pieces of information concerning each emission source. However, many mills and dryers that existed before in the past are no longer in service, and little relevant data could be found. Further, the dryers and mills in Thetford Mines and Black Lake were surrounded by immense tailings piles which complicated the dispersion modeling and limited its validity.

Therefore, the ISC-LT model was only applied to Asbestos, where there was a single asbestos producer with easily defined emission points, and tailings piles were distant from the population and did not lie between the dryer, the mill and the population.

b) Materials, Methods and Assumptions

The ISC-LT model is based on mathematical formulae of physical phenomena of buoyancy and various gas properties, settling velocities and a gaussian atmospheric aerosol dispersion model whereby the dispersion and density of the cloud are expressed as a function of distance from the source, vertical and horizontal standard deviations from the centre of the dust plume, and Pasquill-Gifford classes of atmospheric stability. The model can also take into account some wake effects or turbulences caused by adjacent buildings and obstacles.

Aerosol dispersion simulation models like the ISC-LT are based on assumptions which limit their validity in the case of extremely high emissions of asbestos fibres. These models apply in principle to gases or to very small round particles, and are often extended to isometric[1] particles; but their application to asbestos fibres must be less valid because these are very thin and long particulates which do not behave aerodynamically like isometric aerosols, as their

[1] Isometric: symmetrically and regularly shaped particles with similar dimensions in every axis.

movement is determined by their alignment, interception and other idiosyncrasies. The massive concentrated dust emissions of the past also hinder the validity of aerosol dispersion modeling; models cannot account for "coagulation" or flocculation which occurs when particle number concentration in a cloud is extremely high, a phenomenon likely to be even more severe with fibrous particles; the effect of coagulation is complex as it inhibits cloud settling on one hand but also affects the entrainment of smaller particles by larger ones which deposit more rapidly. On the other hand, the particle size distribution inputted into the ISC-LT simulation may partly account beforehand for the non-isometric, coagulation and entrainment characteristics of past asbestos emissions irrespective of the model. Indeed, the aerodynamic size distribution of dusts stopped by today's baghouses and presumably emitted into the environment before the installation of baghouses was characterized by a Ro-Tap sifter, a dry classification system which does not suspend the dust into a fluid medium and thus cannot completely separate fibres of different sizes. Instead, the dust particles are deposited on shaking plates and are constantly hitting or rubbing against each other, so that many short fibres stick to or are entrained by longer fibres, ending up in the larger D_{ae} strata.

The models also assume that atmospheric stability (Pasquill-Turner index) is even and that wind is homogeneous in the whole diffusion layer. Another important limitation with the simulations was that the meteorological data used as input in the model was not specific enough; the Environment Canada daily meteorological data came from Sherbrooke, and the mixing height (1150 m) was estimated from measurements taken at Maniwaki and Sept-Îles, hundreds of kilometers away from Asbestos. Finally, the air volumes per ton produced, the grades of fibres produced, the gas temperature of the dryer and the proportion of recirculated air in the plant were somewhat different in the past from what they were in recent years and were not documented. Nevertheless these approximations were not greater than in other approaches used in this exposure assessment, and the model was applied to the town of Asbestos.

The US-EPA's ISC-LT aerosol dispersion simulation program was used under the supervision of R. Leduc, Ph.D., meteorologist in the Quebec Ministry of Environment. The basic dispersion formula and the data requirements of the model are listed in Appendix B8, along with explanatory figures. Dr. Leduc lent us a complete meteorological data file in the standard "STAR" format required by the simulation program; the data comprised the standard meteorological measurements taken by Environment Canada over the 1985-1990 period in Sherbrooke, 40 km south of Asbestos. A large-scale topographic map of Asbestos was used to identify the geographical coordinates of the mill and dryer in 1949-1950 as well as their altitudes and those of the surrounding area. The 1950 yearly average asbestos production volume was also input in the model, as well as the 1950 heights and today's (in the absence of 1950 data) dimensions of stacks and louvers, and today's engineering data on air and dust volumes emitted by the J-M Asbestos mill and dryer. The aerodynamic equivalent diameter (D_{ae}) size distribution of the dust retained by the filter bags today and characterized in 1.1.f was also input in the model.

For the 1972 aerosol dispersion simulations, the 1972 production volume and geographical coordinates (new plants had been built and the old ones had been destroyed) were input in the model. The projections were compared with the 1972 environmental measurements by the Quebec Ministry of Environment used in Section B.1.6.c on "Using the Pollution-Production Model to Infer Past Concentration".

The simulations estimated annual average outdoor dust concentrations ($\mu g/m^3$) of airborne dusts and annual deposition rates (g/m^2) in Asbestos for the years 1946-1950; these levels were represented by isolines of same dust concentration levels on a topographic map. The same was done for 1972.

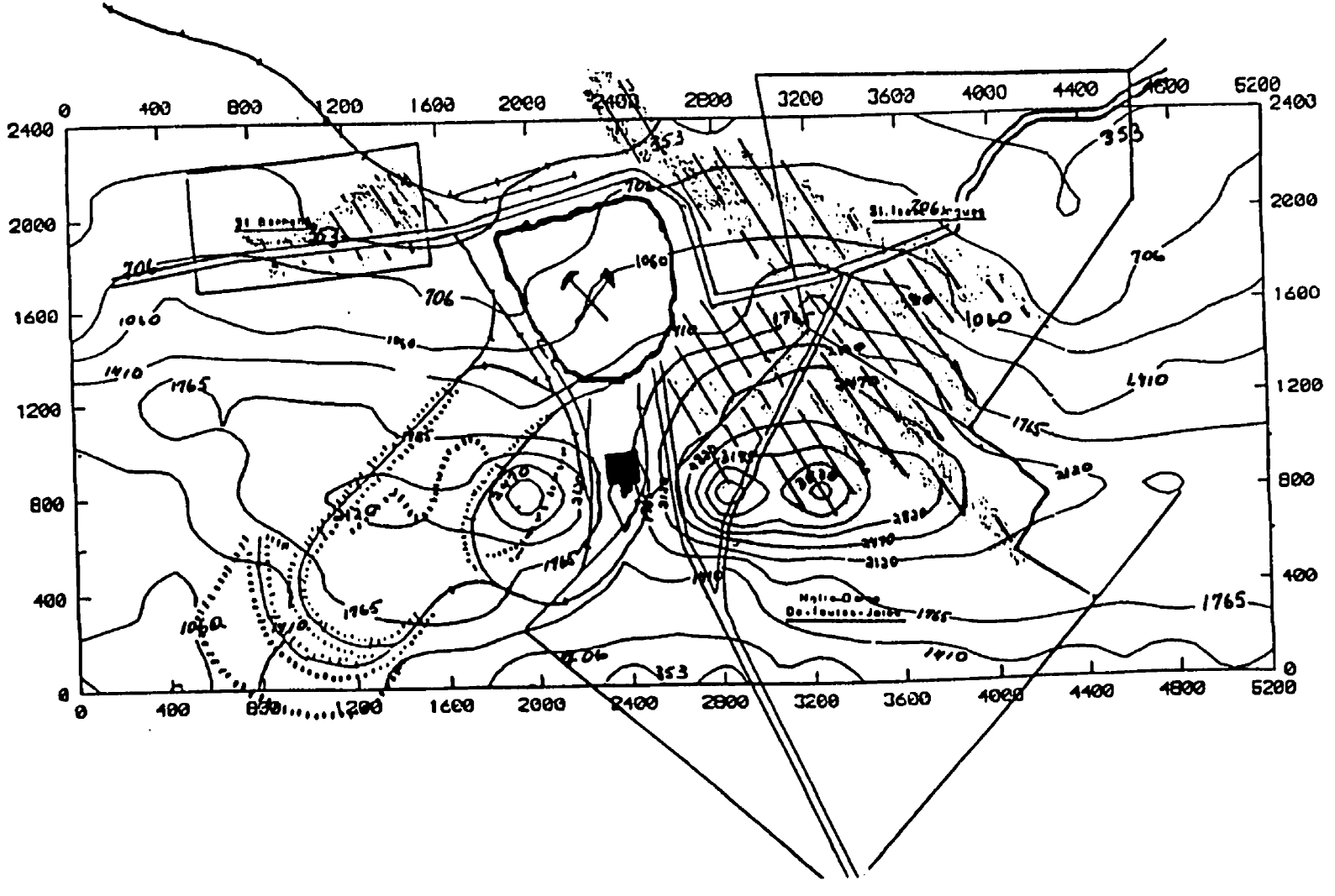
c) Results

i. Projections for 1945, Before the Introduction of Controls

Figure B-5 shows a map of Asbestos with superimposed "mill + dryer" dust concentration ($\mu g/m^3$) isolines 1.5 meters (breathing height) above the ground. The concentration level "isolines" stretched from west to east mostly, reflecting the dominant westerly and subdominant south-westerly and north-westerly winds; the more frequent and the stronger the winds, the further the dusts would be carried away from the emission source before reaching the breathing space of the population.

Under the assumptions of the present simulation, the town of Asbestos in 1945 would have been exposed to annual dust levels attributable to the asbestos industry between 300 $\mu g/m^3$ and 4,000 $\mu g/m^3$, with a town average of roughly 2,000 $\mu g/m^3$. Saint-Barnabé, the neighbourhood north-west of the pit would have been the least exposed, with some 300-700 $\mu g/m^3$; followed by St-Isaac-Jogues, east and farthest from the sources in 1945, with 700-2,100 $\mu g/m^3$. The most exposed neighbourhoods would have been Notre-Dame-de-Toutes-Joies, east from and very close to the mill and dryer in 1945, with some 2,100-4,000 $\mu g/m^3$, and a neighbourhood north-east and very close to the mill and dryer in 1945 (an area west of St-Isaac-Jogues and north-west of Notre-Dame-de-Toutes-Joies, that has since disappeared in the eastward enlargement the pit), with 300-2,500 $\mu g/m^3$. Using the conversion factor (1 $\mu g/m^3$ = 1 f/mL) estimated by regression in Section B.1.6 on "Projections From Recent Measurements", the annual exposure levels in Asbestos in 1945 would translate to a range of 0.3-4.0 f/mL, with a town average of about 2.0 f/mL. Towards the north-east, concentration levels decreased by roughly 150 $\mu g/m^3$ or 0.15 f/mL per 100 meters over most of the residential area. In fact, the decrease was necessarily steeper near the sources and flatter away from the sources, it was also steeper along the north-south axis than along the west-east axis.

Figure B-5 **Dust Concentration Levels (Isolines) Estimated by the ISC-LT Model for Asbestos, Year 1945**



ii. The Relative Impact of Mining and Milling on Outdoor Pollution

The respective contributions of mill and dryer emissions could be assessed with the model since projections were made for these two sources separately taking into account the characteristics of the emission sources. The emission factors[1] used for this dispersion simulation were 450 kg/ton for milling and 80 kg/ton for drying, for a mill/dryer emission ratio of 5.6. However, the aerosol dispersion simulation confirmed that the physical emission characteristics of the two processes[2] substantially increased the influence of mills relative to that of dryers in the ambient concentrations of aerosols in residential areas. Thus mills would have contributed 15 times more than dryers to outdoor air pollution before the introduction of baghouses, a mill/dryer pollution ratio nearly 3 times greater than the mill/dryer emission ratio.

iii. Measurements vs. Projections for 1972

To assess the reliability of the preceding ISC-LT estimates, the model was applied to 1972, a year for which all input data required by the model were available and some empirical measurements were available for comparison with the model's projections. According to the mapped projections in Figure B8-4 of Appendix B8, the town of Asbestos in 1972 would have been exposed to annual levels between 10 $\mu g/m^3$ and 150 $\mu g/m^3$, with a town average of roughly 100 $\mu g/m^3$ (about 0.1 f/mL). Saint-Barnabé and Saint-Aimé would have been exposed to some 10 $\mu g/m^3$, St-Isaac-Jogues to 10-80 $\mu g/m^3$, a small area that has now been engulfed by the pit's expansion to 10-120 $\mu g/m^3$, and Notre-Dame-de-Toutes-Joies to 80-170 $\mu g/m^3$. The simulated concentrations for 1972 seemed about 14 times lower than those for 1945.

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1 Emitted dusts per ton of fibre produced (see Section B.1.2).

2 In particular, relative to mills, dryers have smaller air/gas flows, higher gas temperatures and higher emission points (stacks rather than louvers).

Table B-7 Comparison of ISC-LT Projections With Concentration Measurements in 1972, by Sampling Station

Sampling Area	ISC-LT Projections $\mu g/m^3$	Q. M. Envir. Annual Data $\mu g/m^3$	Absolute Difference $\mu g/m^3$
West of N-D-de-Toutes-Joies	125	97	|+28|
Area engulfed by the pit	60	102	| -42|
West of St-Isaac-Jogues	45	73	| -28|
East of N-D-de-Toutes-Joies	75	53	|+22|
Saint-Aimé	(20)	not measured	---
Saint-Barnabé	(20)	not measured	---
Arithmetic averages and mean absolute difference	76	81	| 30 |

Table B-7 compares the available annual average measurements taken by the Quebec Ministry of Environment in 1972 with the ISC-LT projections. On average, the measurements were merely 5 $\mu g/m^3$ lower than the aerosol dispersion model's projections. However, the absolute difference between observed and projected concentrations at each station was on average about 30 $\mu g/m^3$ or 40%. Given the number of assumptions and approximations involved in the dispersion modeling, and the error in the measurements, this is a remarkably close agreement.

d) Discussion

The comparison of ISC-LT projections with outdoor air measurements was not completely valid because the dust emission data input in the model and thus the dispersion simulation mistakenly comprised non-respirable dusts in a gravimetric proportion of 28%. On the contrary, the Quebec Ministry of Environment sampled only respirable dusts. If the comparison had been restricted to respirable dust emissions and measurements, the simulations would have underestimated the measured gravimetric concentrations by about 25 $\mu g/m^3$, a sizable underestimation error of about -30%. Yet, the uncorrected projections for 1972 approximated well the measurements, and thus the uncorrected projections for 1945 should be a valid estimation of the true past respirable concentration levels.

Since the ISC-LT projections were partly based on annual production volumes and on estimated emission factors for 1945 (dusts retained by modern baghouses) and for 1972 (Gagan's estimates for 1974), the relative concordance between the model's projections and actual measurements in 1972 also enhances the credibility to the other substudies of the present exposure assessment where past exposure levels have been also estimated on the basis of annual production volumes and emission factor estimates.

e) Conclusion

The global results of the aerosol dispersion modeling for the whole town of Asbestos agreed closely in 1972 (about 76 $\mu g/m^3$) with the yearly average measurements by Environment Quebec (about 80 $\mu g/m^3$). While this does not demonstrate persuasively the validity of the ISC-LT projection of about 1700 $\mu g/m^3$ or 1.7 f/mL for Asbestos in 1945, it lends credibility to that estimate.

B.1.4. Past Visible Asbestos Pollution Recalled by Residents

a) Introduction

Residents of asbestos mining towns have long seen fallouts from dust plumes over their towns. As far back as 1912, the municipal authorities of Thetford Mines made repeated representations to the asbestos companies to limit their dust emissions [Cinq-Mars et al., 1994, pp.201-2]. Common sayings, writings and movies referring to the town's dustiness and similar accounts by local residents and leaders are abundant (Appendix B9). Photographs such as Figure B-6, and others in Appendix B9, also testify to the visibility and importance of past dust pollution in the asbestos mining towns.

Figure B-6 Looking West[1] of Thetford Mines in 1905: Dust Emissions From Mining Operations West of the Municipality[2]

Vue de la ville vers le sud à partir du clocher de l'église de Saint-Alphonse en 1905. À l'arrière-plan le viaduc traverse la rue Notre-Dame Sud.
SAHRA. Collection Galerie de nos ancêtres de l'or blanc.

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1 The French inscription under the picture says that picture was taken looking "south" and mentions the street "Notre-Dame South"; however, due to an original error in the orientation of the municipality and its streets, the official "south" of the city is really west, "north" is really east, and so forth. [Cinq-Mars et al., 1994, p.186]

2 Reproduced from Cinq-Mars et al. [1994, p.27] with the permission of the municipality of Thetford Mines [Mr. Y. Faucher].

There must be some correlation between actual asbestos exposure levels in an asbestos-polluted environment and the visibility of airborne or deposited dust. Over the past 15 years, occupational asbestos dust in Quebec's asbestos industry have generally not been visible and the corresponding measured concentrations usually varied between 0.1 and 0.75 f/cc (1 f/cc = 1 f/mL). In 1974, even when asbestos levels as high as 15 f/mL were frequently measured [Nicholson et al., 1979], dust aerosols or deposits were rarely visible in those workplaces. However, before 1950, dustiness was usually apparent in those workplaces and dust levels averaged around 15-25 mpcf or 50-75 f/mL [Liddell, 1991]. In 1989, during my visit to an asbestos mill, an engineer, an industrial hygienist, two fibre quality verifiers and a plant manager mentioned an area of the plant where respiratory masks are mandatory and where dusts accumulate on the floor. It was mentioned that the visible depositions there were similar to those they used to see on the porch of their homes in their childhood in the early 1950s. The hygienist mentioned that he had measured some 2 f/mL in that area. Such experiences and anecdotes corroborate the notion that dust visibility in an asbestos-polluted environment is correlated with asbestos fibre concentration. The correlation is imperfect however, since the proportion of respirable asbestos fibres in airborne dust varies throughout an asbestos mining and milling operation[1].

Visible dustiness may give some general indication of airborne asbestos levels in areas of extreme asbestos pollution, i.e. where asbestos dusts constitute an important proportion of total particulate pollution. Even then however, visible signs of dustiness would not all be equally feasible and representative of actual asbestos exposures. For instance, visible *emissions* might have been seen in any particular instance by every resident of a mining town without affecting everyone equally; depending on the direction, and on dispersion and falling speeds of dusts in such plumes, large parts of the town would not be affected directly. Moreover, visibility depends partly on extraneous factors such as humidity and temperature of the plume, climatic

[1] Indeed, even parallel industrial hygiene samples of respirable dust particles and airborne respirable asbestos fibres show only a weak (0.5) albeit positive log-log correlation [Dagbert, 1976], reflecting different proportions of asbestos fibre in respirable dust aerosols.

conditions, and the proportion of heavy non-respirable dusts in the plume. Thus visible emissions would not be a reliable or valid indicator of the population's asbestos exposure. On the other hand, visible airborne dustiness near residents' homes might reflect more truly the population's actual exposure levels, but no resident interviewed in the feasibility study mentioned such visible dustiness as a common occurrence. However, one dustiness indicator seemed more representative of individual exposures and was reported consistently and vividly by residents interviewed and in writings on Quebec's asbestos mining towns and areas: visible dust deposits in or near the home. The visibility of deposited asbestos dust fallouts could be graduated in terms of intensity and frequency, and its distribution in space and time could be documented by a population survey.

A survey was thus conducted to determine where and when female residents had seen visible asbestos dust deposits during their lifetime, and to characterize the intensity and frequency of such sightings by town, neighbourhood and year.

The main objectives of the study of visible dust exposure recalls were to determine 1) when were asbestos deposits most visible, 2) what mining town(s) and neighbourhoods were dustiest, 3) what were the major determinants (era, town, neighbourhood, distance) of visible dust exposure, and 4) how much more visible dustiness there was in the past relative to the early 1970s, the earliest period for which airborne asbestos concentration data are available.

b) Materials and Methods

The survey was designed 1) to obtain lifetime residential histories for the Cumulative Exposure part of the assessment, 2) to collect health and socio-demographic data on the exposed population for the Mortality Study, and 3) to assess recall of past visible dust deposits.

The survey was conducted in the Spring of 1989 on a random systematic stratified sample of 1096 women at least 50 years of age, 20% in the "50-59" age stratum, 40% in the "60-69" age

stratum, and 40% in the "70 years and over" stratum. The sample was geographically stratified as follows: Thetford Mines agglomeration - 42%, Asbestos agglomeration - 33%, surrounding areas - 25%. The sample was selected from the 1985 Provincial electoral lists. 817 women answered a lengthy questionnaire either by mail or in a personal interview (response rate 74.5%). Response rates hardly differed by age or geographic stratum.

Regarding past neighbourhood exposures, respondents were asked to identify all their residences since birth, their addresses (street, parish or neighbourhood) and their age at arrival to and departure from each of these addresses. Addresses were recoded in terms of neighbourhoods familiar to the local population. In a separate question, the respondents were also asked if they recalled seeing any of six indicators of visible asbestos dust deposits, and if so from what age to what age. The age data were recoded as calendar years, using the respondent's year of birth.

The following is an abridged version[1] of the questions bearing on visible dust deposition. A copy of the full questionnaire can be found in Appendix B10.

1. Have you *ever* seen asbestos dust deposits near your home? _____
   From age __ to age __.
2. Was there a period of your life when you saw deposits *nearly every week* ?
   _____ From age __ to age __.
3. Was there a period of your life when you saw deposits *nearly every day* ?
   _____ From age __ to age __.
4. Was there a period of your life when you could see *footprints* in the dust deposits near your house? _____ From age __ to age __.
5. Was there a period of your life when you would sometimes return from a walk with asbestos dust *on your head, shoulders or clothes* ? _____
   From age __ to age __.

[1] Another question bore on the visibility of dusts on lawns after thawing in Spring. It was dropped in the analysis because there was no difference between the answers to this question and to the question on "ever seeing" dust deposits.

For each neighbourhood and town, for each of four "eras"[1] from 1920 to the present and for each of three distance categories from a mining operations, the person-years proportion of respondents reporting each visible exposure indicator was computed.

Example #1:

If 80 respondents in 1989 reported living at least 1 year in Asbestos between 1920 and 1949, on average 20 years each, a denominator of 1,600 person-years was available for that town-era stratum. If 75 of these 80 respondents reported seeing *some* deposits near their home in that era, and if they did so on average 17.5 years each, totaling 1,312 person-years, then the average proportion of such sightings over that period in Asbestos was computed as: 1,312 ÷ 1,600 = 82%.

Example #2:

- A 70-year old respondent in 1989 who lived in Asbestos from 1925 to 1968 contributed 25 person-years from 1925 to 1949 to the denominator of the "Asbestos:1920-1949" cell, and 19 person-years from 1950 to 1968 to the "Asbestos:1950-1969" cell. If she recalled seeing weekly deposits from age 20 to 40, i.e. from 1939 to 1959, she contributed 11 person-years from 1939 to 1949 to the "weekly sightings" numerator of the "Asbestos:1920-1949 cell", and 10 person-years from 1950 to 1959 to "weekly sightings" numerator of the "Asbestos:1950-1969" cell.

The number of the 817 respondents in 1989 who where living in a given town in a given year in the past is represented in Figure B-7. The surface under a town's curve between two years represents a person-years denominator over that time period in that town. 15 respondents were excluded after answering the survey because their answers seemed incongruous[2].

To make a synthetic analysis and graphical representation of the recalled indicators of deposition sightings, an index score[3] was developed as the square root of the proportion of the

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[1] "Eras" refer to relatively homogeneous periods with respect to reported dustiness trends and dust emission controls.

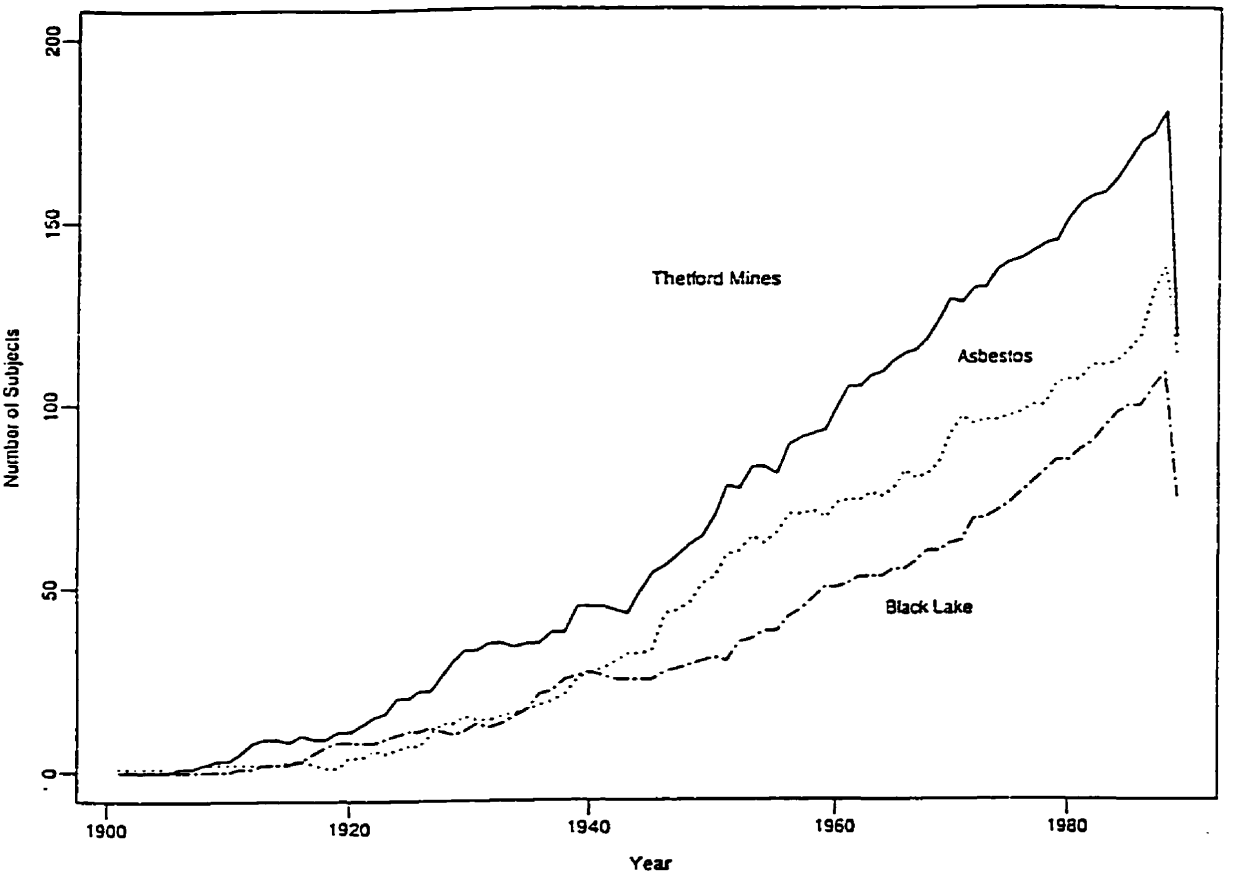
[2] 15 respondents were entirely excluded from the results because they reported seeing weekly dust deposits and even footprints in the 1980s when visible deposits were rare and slight: they probably misunderstood the questions. Their answers to the questionnaire were considered invalid.

[3] Out of seven tentative synthetic indices, this composite exposure index was the one that obtained the highest squared correlation when regressed on three variables: town, era and distance of residence from the closest mill.

5 visible deposits indicators (I_n) reported for a given year: $\sqrt{\sum_{n=5}^{5} I_n / 5}$. This score was computed separately for each year of life of each respondent and was imputed to the specific town where the respondent lived in any given year. The score had a minimum value of 0.00 and a maximum of 1.00. The graphs of PY frequency curves were smoothed by a distance weighted least squares algorithm within the S-Plus® statistical software package.

Figure B-7 Sample Size for Each Town by Calendar Year

Number of Women Who Were Available to Respond for Each Town and Each Calendar Year

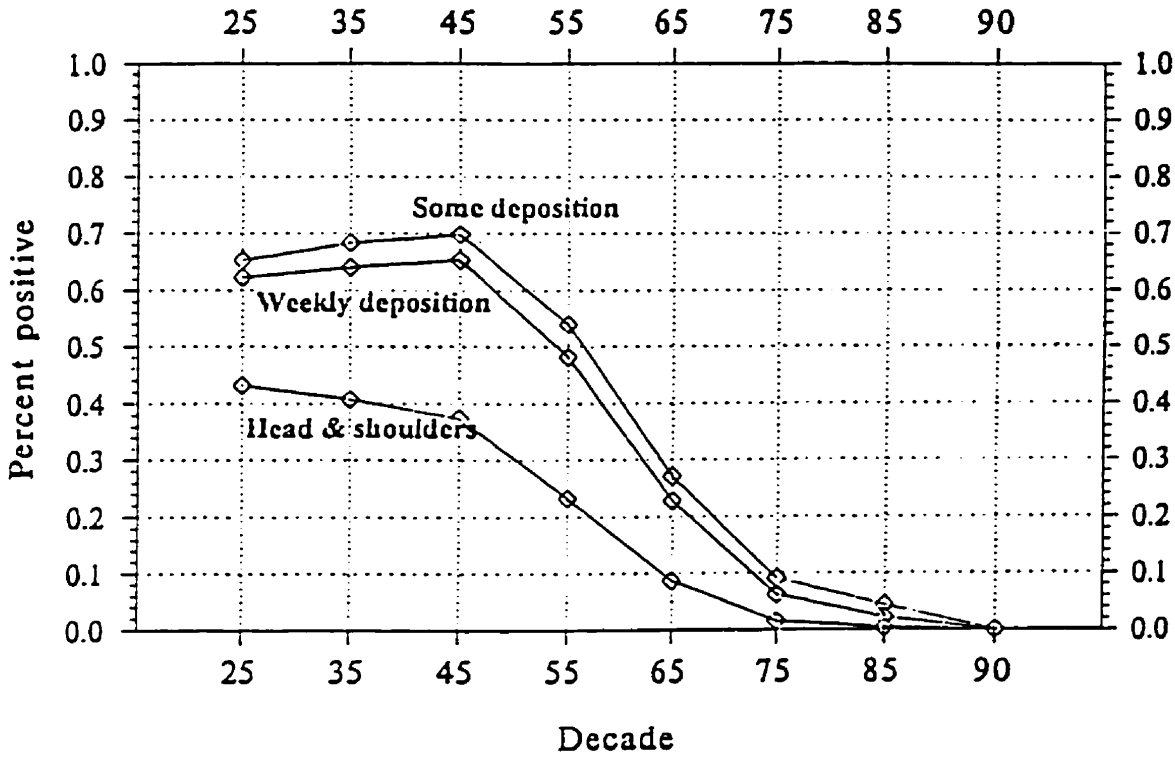


c) Results

i. Towns and Eras

Time and "era" were the strongest determinants of dust deposition sightings, and an intensity gradient was apparent among the visibility indicators in the questionnaire (detailed results in Appendix B11). Over the 1920-1949 pre-baghouse era, "some" deposition sightings represented 73% of the respondents' person-years of residence in the three mining towns, weekly sightings represented 68%, daily sightings 60%, "footprints" 43%, and "head or shoulders" 39%. PY frequencies of recalled sightings were down to 39% for "some" and 14% for "head or shoulders" over the 1950-1969 era, and down to 7% for "some" and 0% for "head or shoulders" sightings over the 1970-1979 era. The graph in Figure B-8 compares the time trends for three different indicators and represents PY proportions by decade pooled for the three mining towns.

Figure B-8 **Percent of PY[1] for Which Respondents Reported Each of Three Visible Exposure Indicators in One of the Three Mining Towns, by Decade**



(each decade is represented by its mid-point)

[1] PY: person-years.

Overall time trends and differences between towns are shown synthetically with a year-by-year time resolution in Figure B-9. The graph shows the relation between the dustiness recall score and year of exposure, for the three mining towns. The graph's y-axis indicates the average score of all the respondents in 1989 who reported living in a given town in a given year.

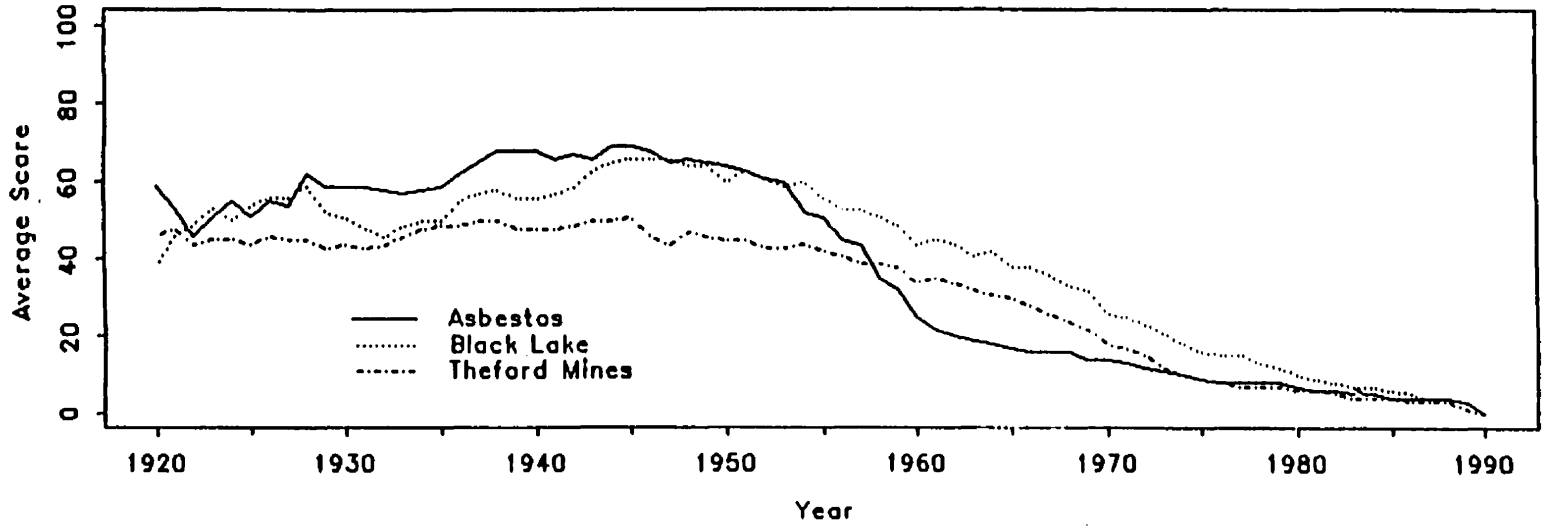
Before 1954, reported dust deposit sightings may have been more frequent in Asbestos than in Black Lake but not significantly; however they were lowest in Thetford Mines. From 1954 on, the score was constantly and substantially highest in Black Lake and lowest in Asbestos. The dustiness recall score began dropping around 1945-1950 in the three mining towns, when the first emission controls were installed. Between 1954 and 1960, recalls of visible deposits dropped most rapidly in Asbestos where the sole operator installed baghouses in 1954 and in 1958. In Thetford Mines and Black Lake, the score dropped gradually and constantly between 1955 and 1975.

ii. Neighbourhoods

The visible dustiness recall score was about the same in all neighbourhoods of Asbestos after 1945, and the apparent differences before that year were not significant. (See Appendix B12-1 on different visibility indices in Asbestos)

In Black Lake, all neighbourhoods had about equally high dustiness recall scores throughout the observation period.

Figure B-9 Dustiness Recall Score for Each of the Three Main Mining Towns, by Year



In Thetford Mines however, there were significant differences between neighbourhoods before 1974. The Old Saint-Maurice neighbourhood had the highest dustiness score from 1935 to 1970, followed closely by Saint-Georges, then by St-Alphonse, O'Meara and Mitchell, all neighbourhoods tightly surrounded by tailings piles, mining pits, dryers and mills. The dustiness score for residing in the Notre-Dame downtown area[1] was about half as elevated as in the preceding neighbourhoods. The least dusty neighbourhoods were those that developed after 1950 and were most remote from the asbestos emission sources: Sainte-Marthe, the relocated "New Saint-Maurice", and Saint-Noël.

iii. Distance From Nearest Mill

As indicated in Table B-8, the PY recall frequency of a dustiness indicator decreased markedly with increasing distance from the nearest mill, and more so for the indicators of highest "intensity".

The relation between distance[2] from nearest mill and recalls of dust deposition partly explained the lower recall of deposition sightings in Thetford Mines than in the other towns before 1950. Thus, 44% of the person-years lived in Thetford Mines before 1950 were spent more than 800 m from a mill, whereas the corresponding proportion was only 22% in Asbestos and Black Lake (Appendix B11); these self-reported distances were consistent with our own geographical estimates in Tables B-1 and B-2. As shown in Table B-9, adjusting for these distance differences by internal standardization[3] explained out the apparent difference between the PY frequency of weekly sightings in Thetford Mines and that in the other two mining towns.

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1 Downtown area, north-east of and further away from asbestos emission sources as were the above mentioned neighbourhoods.

2 "Distance" is an approximate and self-assessed variable. When respondents deemed that they lived - in their opinion - less than 1.6 km from a mill, they gave their own appreciation of the average distance from the nearest mill over that period by choosing among 5 possible distance categories. These self-reported distances from a mill seemed consistent with neighbourhoods and eyeballed distances on historical maps.

3 The adjustment weights were the sums of person-years for the three towns for each distance stratum.

Table B-8 **Percent of PY for Which Respondents Reported Seeing Dust Depositions in the Three Mining Towns between 1920 and 1980, by Reported Distance From a Mill**

Dustiness Indicator	Shortest Distance From a Mill		
	< 600 m[1]	< 1.6 km	≥ 1.6 km
	%	%	%
Some deposition	92	77	10
Weekly	89	61	4
Daily	81	50	3
Footsteps	62	27	2
Head and shoulders	48	19	2

[1] Given the small number of respondents (15 and 20) in each of the two smallest distance categories (75 m, and 150 m) and the similarity between their average PY-percentage recalls and those of the 400 m category, the three categories were merged in this table. The 600 m cutoff is simply the mid-point between the "400 m" and "800 m" categories.

Table B-9 Annual Probability of Seeing Weekly Deposits Before 1950 by Distance From Nearest Mill and by Town

Distance	Asbestos %	Black Lake %	Thetford Mines %	All Three Mining Towns	Denomin. PY (N)
> 1600 m	19%	33%	19%	20%	1500 (50)
800 -1599 m	57%	76%	54%	57%	537 (19)
400 - 799 m	76%	58%	76%	72%	1180 (39)
150 - 399 m	93%	83%	80%	87%	1757 (59)
75 - 149 m	82%	87%	79%	81%	599 (20)
< 75 m	86%	99%	97%	96%	409 (15)
Crude Aver. Prob. (PY:N)	75% (1804:60)	70% (944:33)	57% (3234:109)	65% (5982:202)	5982 (202)
Distance-Adjusted[1] Aver. Prob.	66%	66%	63%	65%	

[1] Adjustment by direct standardization : the adjustment weights were the sums of person-years for the three towns for each distance stratum.

d) Discussion

There was a logical gradient of intensity and specificity among the five dustiness indicators, the reporting frequency of an indicator being inversely proportional to the dustiness severity or intensity that it represented. The frequency of recalled deposition sightings was inversely related to distance from a mill and era; the shorter this distance and the earlier the era, the more frequent were the reports of sightings. Indeed, every dustiness indicator was reported more frequently closer to a mill and further back in time. Dust deposition recalls were thus consistent with aerosol dispersion principles and with the history of dust controls. Moreover, homogeneity of recalls was observed within the most extreme space-time categories; thus weekly depositions were reported for 96% of the PY's lived less than 75 m from the nearest mill before 1950, for less than 4% of the PY's lived in the 1970s independently of the distance from a mill, and for less than 4% of all PY's lived at more than 1.6 km from a mill independently of era.

The post-1950 patterns of rapid decline in dustiness recalls in Asbestos and slower decline in Black Lake concurred with the installation of baghouses in Asbestos on both the dryer and mill in 1954-1958, and with the combination of rapid production growth and increasing number and size of tailings piles in Black Lake. These trends were amplified by the increasing distance between emission sources and residential areas in Asbestos in contrast to the decrease in Black Lake. In Thetford Mines, the installation of dust controls was spread out over the whole 1960-1979 period while total asbestos production barely increased over the 1950-1969 period. These factors would explain the steady decrease in recalled dust deposit sightings over that period, intermediate between the sudden 1950s cleanup in Asbestos and the slower improvement in a burgeoning Black Lake.

The pre-1950 town-by-town patterns in the dustiness recall score, in particular the lower sighting frequencies recollected in Thetford Mines relative to the two other mining towns, cannot be explained by the relatively greater distance between dwellings and the nearest mill in Thetford Mines since production volume was about twice as high in Thetford Mines as in either of the two other mining towns before 1950.

Three possible explanations for this pre-1950 dustiness-production paradox might be suggested. First, the home-mill distance variable was very crude, categorical, subjective and prone to recall error, an error which would increase with time since the sightings. Therefore, the effect of home-mill distance could have been significantly underestimated due to such nondifferential misclassification. Second, recall of past sightings might be biased differentially in the three mining towns in the way of overestimating pre-1950 dustiness in Asbestos before the sudden and "dramatic" reduction in emissions in the mid-1950s. Third, since Thetford Mines was the only asbestos mining town where a significant part of mining was done underground, mining emissions and fallouts must have been lower than expected on the basis of production volume alone, resulting in proportionately less deposition of heavier dust aerosols[1].

On the other hand, "non-mill" emission sources like tailings piles were closer to residences in Thetford Mines than in Asbestos (Tables B-1 and B-2), and wind direction should also have induced more sightings in Thetford Mines, this town being more downwind from emission sources than the two other towns before 1950. Finally, there was no obvious or adequate justification of the paradoxically lower PY frequency of dust deposition sightings in Thetford Mines before 1950.

This being said, since there is no way of directly inferring airborne concentrations from the proportion of residents seeing dust deposits in a given year, it cannot be concluded that

[1] In the town of Asbestos in 1972, the governmental engineer Denizeau analyzed respirable dusts from four sampling stations and observed a greater proportion of heavy dust particles nearer to the open mining pit [Denizeau, 1973].

respirable asbestos concentrations in the ambient air would have followed the same pattern as visible dust depositions.

e) Conclusion

The recall survey on past visible dust deposits suggested the following relations: 1) dustiness was extremely high before 1950; 2) it declined radically over the 1950-1975 period; 3) weekly dust deposition sightings were reported about 8 times more frequently around 1960 than in the mid-1970s, and about twice more frequently around 1945 than in 1960; 4) after 1950, Black Lake was the most visibly polluted mining town and Asbestos was the least polluted; 5) before 1950, sightings were recalled less frequently and less vividly in Thetford Mines than in Asbestos and Black Lake; 6) distance from a mill was a major determinant of dust deposit sightings.

While asbestos concentration levels were not quantifiable from these recalls of visible sightings, the trends and patterns were used by the expert panel to estimate relative pollution levels between time periods and between towns and mostly to corroborate or qualify relative estimates based on other data.

B.1.5. Past Exposures Estimated From Lung Burden

a) Introduction

This section describes statistical analyses of available data on the lung tissue burden of asbestos fibres among occupationally and non-occupationally exposed persons. Inferences are made about the past exposure to asbestos about residents of the mining towns - both those with neighbourhood-only exposure and those with household-contact exposure. The state of knowledge on asbestos lung burden and biokinetics, the datasets, the derivation of the biokinetic model and the nonlinear regression methods are only briefly summarized.

i. From Exposure to Lung Burden

A small fraction of the respirable airborne particles in a person's breathing zone will consist of asbestos fibres and will be deposited into the gas-exchange alveolar region of the lung. These deposited asbestos fibres tend to accumulate in the deep lung. The concentration of these fibres in lung tissue collected *post mortem* constitutes the asbestos *lung burden* at time of death reported in lung tissue studies: the number of asbestos fibres > 5 μm per microgram of dry lung tissue (f/μg) counted by transmission electron microscopy (TEM).

The accumulation of asbestos fibres deposited in the lung is determined by segmentation of deposited fibres and by *clearance* mechanisms such as leaching, phagocytosis and translocation. The faster the clearance, the less representative the lung burden will be of past exposures. Numerous factors affect clearance: mineralogical type, exposure intensity, fibre

size [Lee et al., 1981; Sébastien et al., 1986; Churg and DePaoli, 1988; Churg et al., 1989; Davis, 1989; Goodglick and Kane, 1990; Sébastien et al., 1990], diseases, age [Langer et al., 1971; Case et al., 1988; Coin et al., 1994], smoking and other factors [Morgan, 1980; McFadden et al., 1986a; McFadden et al., 1986b; Churg et al., 1987; Tron et al., 1987; Davis et al., 1991; Churg et al., 1992; Churg and Stevens, 1995]. Clearance rates also change as a function of time since exposure, slowing down over time.

Despite the complexity and idiosyncratic nature of inhalation, deposition and biopersistence dynamics [Bégin and Sébastien, 1989], lung burden determined *post mortem* has been shown to correlate with past asbestos exposure levels, duration of exposure, and time since last exposure [Davis et al., 1986; Churg and DePaoli, 1988; Jones et al., 1988; Berry et al., 1989; Case and Sébastien, 1989; Sébastien et al., 1989; Dutoit, 1991]. Lung burden is thus a biomarker of past exposure to durable particulates.

ii. Lung Burden Biomarkers

Three types of asbestos biomarkers have been measured in the lungs of some residents of the asbestos mining area: chrysotile fibre, tremolite fibre and asbestos bodies. To estimate past cumulative exposures or lifetime average exposure intensities, it was necessary that the asbestos biomarker used represent long-standing exposures and thus that its clearance rate be relatively slow. The best of the available biomarkers was tremolite fibre burden of the lung.

Tremolite is a natural contaminant of chrysotile fibre ore and of the air of Quebec's mining towns. Bearing in mind that the number of tremolite fibres counted was small, the tremolite-chrysotile fibre ratio in the air was about 1:150 in Asbestos, 1:125 in Black Lake and 1:25 in Thetford Mines in 1974 [Sébastien et al., 1986]. This is consistent with lung burden data on asbestos miners and millers and residents [Case and Sébastien, 1989; McDonald and McDonald, 1995] and with recalls of some asbestos companies' geologists that the old Johnson's Mine in Thetford had significant amounts of tremolite in its serpentine ore and chrysotile veins, up to 20% during some years in the past [J. Dunnigan, oct. 1996, personal

communication]. Thus it should be respired in the same proportions. However, tremolite is an amphibole[1] which clears more slowly from the lung than chrysotile [Case et al., 1987; McConnochie et al., 1987; Case et al., 1989; Sébastien et al., 1989; Guth, 1990; Albin et al., 1994], and is found in at least as high a proportion as chrysotile in the lung burden. Paradoxically, tremolite can thus be more representative of long-term chrysotile exposure than chrysotile burden, at least in Thetford Mines and Black Lake. The pulmonary burden of asbestos bodies correlates well with exposure duration and discriminates between exposure groups, but it is less reliable than tremolite burden; less than 1% of fibres in the lung become asbestos bodies, these continue to form even after cessation of exposure, and, in the available datasets, asbestos bodies correlate less with exposure duration than tremolite does. Tremolite was thus the biomarker preferred for the lung burden analyses.

iii. Asbestos Exposure and Lung Burden Datasets

Drs. B. Case, P. Sébastien and C. McDonald have been collecting and analyzing *post mortem* lung tissues from asbestos workers and other residents of the asbestos-mining areas. For the present section, two available lung burden datasets were used: one occupational and one non-occupational, both from the Thetford Mines and Black Lake area. This geographical homogeneity facilitated extrapolations from workers to non-occupationally exposed residents since both groups were probably exposed to similar asbestos aerosols containing similar proportions of tremolite and chrysotile. Such extrapolations are the main endeavour of the present section.

iv. From Lung Burden to Past Exposure

Quantitative biokinetic models developed from animal experiments [Vincent et al., 1985; Vincent et al., 1987; Jones et al., 1989; Vacek et al., 1991] and mathematical lung function

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[1] The pulmonary clearance of different amphiboles has been studied by various investigators [Wagner et al., 1974; Pooley, 1976; Sébastien et al., 1980b; Rowlands et al., 1982; Gylseth et al., 1983; Churg et al., 1989; Davis, 1989].

models [Task Group on Lung Dynamics, 1966; Soderholm, 1981; Gerrity et al., 1983; Smith, 1985; Stöber et al., 1989] were far too complex to be applied to small and imprecise datasets on human asbestos exposure and lung burden. Instead, three approximate approaches were used to estimate the relation between past exposures and tremolite lung burden among some asbestos workers of the Thetford Mines agglomeration: an algebraic ecological projection, a set of intrinsically linear regression models, and a nonlinear biokinetic regression model. Afterwards, the three estimated relations were applied separately to a neighbourhood-only exposure group (n=22) and to a household-contact exposure group (n=10) from the Thetford Mines area to estimate their respective time-weighted average (t.w.a.) exposures[1] for the years lived in the mining area. For a household contact, this t.w.a. exposure estimate comprised both her outdoor and indoor exposures over all years lived in the asbestos area, whether sharing the household of an asbestos worker or not[2].

b) Datasets

The data and methods of the original lung burden studies have been described in detail previously [Case and Sébastien, 1987; Case and Sébastien, 1988; Case and Sébastien, 1989; Sébastien et al., 1989] and are only briefly summarized here. More detail and descriptive statistics are presented in Appendix B15.

i. Occupational Dataset

Sébastien et al. [Sébastien et al., 1989] analyzed the lung burdens of 89 miners and millers from Thetford Mines and of 72 asbestos textile workers from Charleston, South Carolina. The

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[1] Since the different exposure levels to which a person has been exposed for various durations since birth are integrated over the whole lifetime by the lung, *post mortem* analyses can only estimate an average exposure level over that person's lifetime. Assuming no pulmonary clearance, this average is fundamentally equivalent to the industrial hygiene definition of a "t.w.a.": the sum of the products of different levels of exposure by their respective durations, divided by the total exposure time (lifetime or duration of residence in the area).

[2] This should not be confused with the exposure level inside the home of household workers.

latter handled chrysotile asbestos originating from the Thetford area. Individual lifetime average occupational total dust exposures[1], age at death, duration and cessation periods were available from two previous cohort studies [Dement et al., 1983a; McDonald et al., 1993; McDonald et al., 1993; Dement et al., 1994]. Although stratified aggregate data [Sébastien et al., 1989] of both occupational datasets were used to partially validate the biokinetic model (Appendix B14), only the individual Thetford Mines occupational data were used to estimate the relation between lung burden and past average exposure intensity.

Extrapolating the biokinetic relation from the occupational group to the target non-occupational groups was moot because the groups were extremely different in crucial respects. The asbestos workers were male smokers exposed intermittently after the age of 20 to extremely high asbestos levels, whereas the non-occupational target groups were essentially female non-smokers exposed continuously since birth to asbestos levels 10-1000 times lower than in the workplace. The many interactions between these unevenly distributed factors would modify the past-exposure-lung-burden relation between the group of workers and the group of non-occupationally exposed residents; an extrapolation problem compounded by proportional exposure measurement errors [Armstrong, 1983; Doll and Peto, 1986; Armstrong, 1990] and by nonlinear biokinetics [Vincent and Donaldson, 1990; Vacek and McDonald, 1991]. Moreover, the occupational data comprised far outliers on all variables used in the model, making the model-fitting unsteady and strongly influenced by observations least comparable to the non-occupational data. For instance, the median tremolite burden of workers was 16 times higher than the median of the non-occupational group, whereas the highest occupational value was 600 times higher.

To alleviate these non-comparability and heterogeneity problems, 17 observations were excluded from the occupational dataset to improve consistence with non-occupational exposure levels and time patterns: 1 missing tremolite burden value and 4 very extreme values, 6 workers exposed less than two years and 1 who had ceased asbestos work 47 years before

[1] In the past, total respirable dusts were sampled by midget samplers and counted with an optical microscope.

death, 1 worker with an extremely high chrysotile burden, and 1 with an extreme lung burden of asbestos bodies but with low tremolite and chrysotile burdens, 3 workers with extreme mpcf.y/tremolite ratios (220, 1110 and 1589 vs. a median of 12). All excluded observations had very heavy statistical leverages. Thus, only 72 asbestos workers were retained for the analyses.

ii. Non-Occupational Dataset

The non-occupational lung burden dataset consisted of 51 cases collected and analyzed by Dr. B. Case in a previous study [Case and Sébastien, 1989] designed to compare lung burdens of neighbourhood-only, household-contact and referent exposed persons. Autopsies were identified in the same pathology department of the Thetford Mines regional hospital from January 1976 to December 1981. Occupational histories were obtained using company records and the hospital record. Fifty-one (51) cases had no history of work in asbestos mines or mills or any related industry. More occupational data, socio-demographic data, residential and household-contact exposure histories were obtained from a next-of-kin. There were 22 neighbourhood-only exposed residents of Thetford Mines or Black Lake having lived less than 10 km from an asbestos mine or mill for more than 20 of the last 30 years of their life. There were 10 household contacts of asbestos workers: residents who ever lived with a father, mother or spouse who worked in the mines or mills for more than one year. Finally, there were 18 "referents" who had lived more than 10 km from all mines and mills for more than 20 of the last 30 years of their life and had never lived with an asbestos worker. One of the 51 subjects available did not quite fit in any group and was excluded.

The investigators found that lung burden for household contacts was 5 to 10 times higher than in neighbourhood-only exposed subjects. I re-analyzed the data with the objective of estimating absolute as well as relative exposure levels.

c) *Ecological Projections*

Methods

This method is referred to as "ecological" because it is based on group averages[1] rather than individual values. Instead of the usual definition based on geography, the groups were defined here by the type of asbestos exposure experience: occupational, household-contact, neighbourhood-only, and referent exposure groups. The ecological analysis of the grouped data assumed that average lung burden was directly proportional to average cumulative exposure, that there was no pulmonary clearance of lung burden, and that time since last exposure did not affect lung burden. As in lung burden analyses conducted by other researchers, the paucity of data necessitated to assume that the relation between past exposure and lung burden was independent of sex, exposure intensity, age, physical exertion and smoking. Under these assumptions, lung burden should be a constant ratio of cumulative asbestos exposure. The ratios were computed from group averages and medians to provide more stable estimates and projections than the individual-based regressions used in the two other approaches.

The ratio between cumulative-exposure and lung-burden was estimated from the occupational data and was applied to the lung burdens of the non-occupational target groups. Estimates of past average exposure intensity were obtained by dividing each target group's estimated cumulative exposure by its estimated exposure duration calculated as the number of years of non-occupational exposure multiplied by an "occupational intermittence factor" of 4.2 (≈168h./week÷40h./workweek). The projection formulae applied for cumulative exposure estimation and past average exposure intensity estimations were:

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[1] The "averages" being either arithmetic or geometric means or medians.

$$mpcfy_{target} = burden_{target} \bullet \left(\frac{mcpfy_{workers}}{burden_{workers}} \right)$$

$$mpcf_{target} = \left(\frac{burden_{target}}{4.2 \bullet duration_{target}} \right) \bullet \left(\frac{mcpf_{workers} \bullet duration_{workers}}{burden_{workers}} \right)$$

Results

Table B-10 shows the projections of the exposure-burden-duration relations observed in the set of 72 workers and a subset of 10 workers with the lowest tremolite burdens (< 5 f/μg). These two relations were projected onto the median lung burden values of the three non-occupational exposure groups (grouped rows), using the three biomarkers in turn (individual rows). The three biomarkers gave very different exposure projections. The projections based on tremolite were about 3 times higher than those based on asbestos bodies for the household-contact and referent groups, but the projections were similar in the neighbourhood-only exposure group. Contrary to tremolite-based and asbestos bodies-based estimates, projections based on chrysotile burden were very sensitive to the group of workers (72 or 10) on which they were based.

Table B-10 **Estimates of Non-Occupational Cumulative Exposure (mpcf.y) and t.w.a. Exposure Intensity (mpcf) During the Years Lived in the Region Based on Group Averages**

Target Group	Biomarker	Cumulative Exposure Estimate : mpcf.y		Average Exposure Intensity Estimate : mpcf	
		Based on 72 Workers	Based on 10 Workers[1]	Based on 72 Workers	Based on 10 Workers
Household-Contact Group (n=10)	Tremolite	24.0	34.6	.10	.13
	Chrysotile	91.1	35.1	.36	.13
	ABs	8.3	12.4	.03	.05
Neighbourhood-Only Exposure Group (n=22)	Tremolite	4.1	5.9	.03	.03
	Chrysotile	15.9	6.1	.10	.04
	ABs	2.8	4.2	.02	.02
Referent Group (n=18)	Tremolite	.44	.63	.002	.002
	Chrysotile	1.71	.66	.006	.002
	ABs	.14	.22	.001	.001

1 Workers with tremolite< 6 f/µg were selected as most comparable with non-occupational subjects.

Note: A dust to fibre conversion can be applied to the figures in the table : 1 mpcf = 3.5 f/mL. This conversion factor was estimated for Quebec asbestos miners and millers. [Liddell et al., 1984]

[1] Workers with tremolite< 6 f/µg were selected as most comparable with non-occupational subjects.

According to the exposure projections based on *tremolite* burden, residents of the Thetford Mines agglomeration with household-contact exposure would have had a median cumulative exposure of 24-35 mpcf.y or 84-120 f-y/mL[1], with a t.w.a. exposure intensity of 0.10-0.13 mpcf or .35-.46 f/mL while they lived near the mines. The neighbourhood-only exposure group would have been exposed to 4-6 mpcf.y or 14-21 f-y/mL for an average level of about 0.03 mpcf or 0.11 f/mL while they lived near the mines. Since the additional household exposure contributed by the presence of an active asbestos worker would have been independent from the general neighbourhood exposure, household exposure was more logically surmised as being additive than multiplicative relative to neighbourhood exposure. Accordingly, these estimates suggested that household-contact exposed residents received on average about 0.3 f/mL more (= 0.4 - 0.1 f/mL) asbestos exposure than neighbourhood-only exposed residents, per year lived in the asbestos area[2].

d) Intrinsically Linear Regression Models

Methods

Interactive stepwise multiple regression was applied to the data using the best-fitting variable transformations (usually logarithmic) to improve precision, to allow for background levels and lung deposition thresholds, and to try different potential cofactors in turn. Log-linear[3] models were fitted on the set of 72 workers retained for the analyses and on a subset of 36 which experienced the lowest past average exposure intensities (mpcf < 10.9) to see if a different fit

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1 The conversion factor used here is that estimated for the cohort of Quebec asbestos miners and millers: 1 mpcf = 3.5 asbestos fibres >5 µm.

2 The value of 0.3 f/mL would be less than the average per year lived with an asbestos worker. Unfortunately, this number of years was not known from the available data. If it were the same as in our survey, it could be surmised that the average exposure level in the households of asbestos workers was about 0.5 f/mL above the average level in other households of the same neighbourhood.

3 "Log-linear" refers in this text to models with a logarithmic dependent variable. Models with logarithmic transformations are "intrinsically linear" since the form of regression models with logarithmic transformations is linear.

or model would be obtained on workers with exposure levels expectedly closer to those of the target groups. The logarithmic transformation of exposure intensity, cumulative exposure and lung burden variables and of some covariates (e.g. age, distance-from-mine) improved significantly their correlations (see Appendix B13) and the homoscedasticity of residuals, and it gave more weight to workers with lung burden values closer to those of the target groups.

Four modeling approaches were utilized. First, a predictive approach was applied whereby the logarithm of cumulative exposure was regressed on lung burden and various cofactors. Even though lung burden was really the result of past exposure, this approach could be more precise by minimizing the error of predicted cumulative exposure. A second approach applied was also predictive but modeled the logarithm of average exposure intensity as the dependent variable. In a third approach, a more cogent "etiologic" model set lung burden as a resulting function of past exposures and cofactors. To project past exposures of non-occupationally exposed subjects, the inverse of the fitted etiologic function independent variable had to be applied to the non-occupational lung burdens, a statistically dubious procedure since the regression model minimized the error of lung burden while the predicted variable was cumulative exposure and t.w.a. exposure intensity. Fourthly, a simple linear regression model obtained by other authors [Sébastien et al., 1986] from a stratified analysis (R^2=.04) of 39 lung burdens taken less than 75 months after end of employment was applied to the non-occupational data. This linear model has been weakly corroborated (n=94, R^2=.07) by still other investigators [Churg and Wright, 1994] on other occupational lung burden data from Asbestos and Thetford.

Results: Regressions on Individual Asbestos Workers

The various models are presented in Appendix B16 in a table including the explained variance proportions (R^2) of the fitted models, and the average predicted cumulative exposure and t.w.a. exposure intensity for the non-occupational target groups. Appendix B16 also provides a brief discussion of the fitted models.

The eleven "intrinsically linear" models fitted by stepwise multiple regression and based on tremolite burden gave very different exposure estimates in the small non-occupational samples. The linear model predicted negative exposure values. The etiologic models projected unreasonably low past neighbourhood exposure levels, one order of magnitude lower than levels measured even as late as in 1984. The predictive models provided estimates that spread over a range one order of magnitude wide. Projections based on the whole group of 72 workers were somewhat higher than those based on the 36 least exposed workers.

The simplest predictive model of cumulative exposure as a function of lung burden gave the lowest estimates and was the only credible log-linear model. The others predicted past exposure levels 3-15 times higher than did the simplest model and incongruously implied that lung burden was inversely proportional to exposure duration for a given past cumulative exposure. Moreover, no model had a higher adjusted R^2 than the simplest model (R^2=38%) without running into severe multicollinearity.

Based on the 36 workers with the lowest tremolite burdens, predicted values from the simplest predictive model were 0.10 mpcf for household contacts and 0.03 mpcf for neighbourhood-only exposed residents.

e) Nonlinear Biokinetic Lung Retention Model

The methods and results are summarized here. The mathematical development of the model and comparative results for both tremolite burden and chrysotile burden are exposed in more detail in Appendix B17.

Methods

The biokinetic model derives from the following elementary exposure scenario. Suppose a person without any previous asbestos exposure is exposed only for an instant to a respirable

dust exposure concentration I. Suppose also that the short-term resulting pulmonary asbestos fibre burden can be represented by a deposition-conversion K_d of exposure I. Then, if K_e is the long-term clearance rate expressed as the proportion of retained asbestos fibres cleared at the end of one year, the retention fraction of fibres in the lung at one point in time will be $(1-K_e)$ after one year, and $(1-K_e)^y$ after y years. After y years, the contribution to lung burden resulting from this single exposure will be:

$$lung\ burden = I \bullet K_d \bullet (1 - K_e)^y$$

If instead this person was exposed to a constant exposure intensity I over a continuous exposure period (duration) D, lung burden at the end of this period would be the integral of the marginal contributions of each instantaneous new dose to lung burden. If a cessation period of C years occurred between last exposure and time of death at which lung burden was measured, then only a fraction $(1 - Ke)^C$ of the lung burden at the end of the exposure period would remain in the lung at time of death, and the resulting model would be:

$$lung\,burden = I \bullet K_d \bullet \left[\frac{(1 - K_e)^D - 1}{\ln(1 - K_e)} \right] \bullet (1 - K_e)^C$$

The same formula was also derived by Berry et al. [Berry et al., 1989] under the same assumptions.

To account for the lifetime t.w.a. non-occupational exposure *envir* which contributed to the lung burden of workers[1], the above formula was divided by 4.2, the ratio of week to work hours (168 h. / 40 h.), and an appropriate expression for non-occupational exposure was included in the model:

[1] Non-occupational exposure intensity was assumed to have been the same for the 72 workers.

$$lung\,burden = \frac{1}{4.2} \bullet I_{occ} \bullet K_d \bullet \left[\frac{(1-K_e)^D - 1)}{\ln(1-K_e)} \right] \bullet (1-K_e)^C +$$
$$\frac{3.2}{4.2} \bullet I_{nonocc} \bullet K_d \bullet \left[\frac{(1-K_e)^{Age} - 1)}{\ln(1-K_e)} \right]$$

This model cannot be expressed in a linear form using logarithmic or other transformations. So nonlinear iterative regression[1] was applied to estimate K_d , K_e and *envir*. In nonlinear iterative regression [Dennis and Schnabel, 1983; Scales, 1985; Bates and Watts, 1988; SYSTAT and Wilkinson, 1990], the parameters of the model are estimated so as to minimize a loss function, often the least squares function, following an iterative estimation process such as the Quasi-Newton and Simplex minimization algorithms.

As in log-linear regressions, the loss function minimized in the nonlinear regressions was the square of the difference between the logarithms of lung burden and of the regression estimate. A weighted least squares loss function was also used to mitigate the influence of workers with extreme exposure levels; weighing by *1/mpcf* also improved homoscedasticity of the errors with respect to the exposure variable of interest [Johnston, 1984; Armitage and Berry, 1994]. Both loss functions were used and compared in the analyses.

Results of the Nonlinear Biokinetic Modeling

The results of the biokinetic models applied to tremolite burden[2] are presented in Table B-11. The weighted least squares loss function had much more explanatory power (R^2=.83 and .41) than did the logarithmic loss function (R^2=.24 and .08). The clearance rate estimate K_e was 0.9%/year with the logarithmic loss function and 3.8%/year with the weighted least squares loss function. The non-occupational lifetime average exposure intensity estimate was 1.1 mpcf

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[1] We tested the Systat non-linear regression program on different datasets published by other investigators who used weighted linear [Nicholson, 1986], biokinetic [Bartrop et al., 1977], logistic [Bremond et al., 1986; Baker, 1987] and Poisson [Lehrer, 1980; Lehrer, 1981] models with either maximum likelihood or iterative reweighted least squares loss functions; our results concurred with those published.

[2] Results with chrysotile burden are given in **Appendix B17**, Table B17-1.

with a 95%CI of -0.2 to +2.4 mpcf with the logarithmic loss function, and 0.55 mpcf with a 95%CI of 0.26-0.84 mpcf with the weighted least squares model.

After replacing the unknown biokinetic parameters K_d and K_e by their estimated values, the models were fitted to the non-occupational lung burden and exposure time data. The resulting average exposure intensity estimates for the household-contact and the neighbourhood-only exposure groups were respectively 0.49 mpcf and 0.15 mpcf with the logarithmic loss function, and 0.22 mpcf and 0.07 mpcf with the weighted least squares model.

Table B-11 Biokinetic Models Fitted on the Asbestos Workers' Tremolite Burden (n=72) and Projections for the Non-Occupationally Exposed Groups

		- Parameter Estimates -		*- Projections -*	
Loss function	**Parameter / Statistic**	**Estimate**	**95% CI**	**House-Contact mpcf**	**Neighb.-Only mpcf**
Logarithmic least squares	R^2	.24			
	K_{dt}	0.131	-0.07, +0.33		
	K_{et}	0.009	-0.04, +0.06		
	Non-occ. mpcf	1.113	-0.15, +2.38	0.485	0.151
Weighted least squares (w=1/mpcf)	R^2	.83			
	K_{dt}	0.511	-0.08, +1.11		
	K_{et}	0.038	-0.02, +0.09		
	Non-occ. mpcf	0.549	+0.26, +0.84	0.217	0.068

g) Discussion

The results of the three approaches are summarized in Table B-12. The ecological projections and the log-linear model gave very similar results, while the weighted least squares biokinetic model produced somewhat higher estimates. Among regression models, the biokinetic model with a logarithmic loss function produced the highest estimates but had the lowest R^2. However, the weighted least squares biokinetic model had the highest R^2 and produced estimates closer to the results of the ecological projections and log-linear models. The t.w.a. exposure of household contacts during the years that they lived in the asbestos area might be figured at 0.35-1.72 f/mL with a best estimate of 0.76 f/mL. The t.w.a. exposure of neighbourhood-only exposed residents during the years that they lived in the asbestos area might be figured at 0.11-0.53 f/mL with a best estimate of 0.24 f/mL. No modeling approach had a clear edge over the others, but the weighted least squares biokinetic model was preferred slightly because it had a large R^2 and was somewhat validated a) by estimating chrysotile and tremolite clearance rates that were comparable to those estimated in experimental and occupational studies (comparisons in Appendix B18), and b) by estimating an expected contrast between a substantial non-occupational asbestos exposure specific to workers in Thetford Mines and a null non-occupational exposure estimate for workers in Charleston (Appendix B14).

One difficulty with backward projections from lung burden data is that the time period to which the exposure estimates apply is vague. Lung burden naturally reflects cumulative exposure which is not time-specific. Most subjects in the non-occupational datasets died in the 1980s and the average exposure period might be considered as somewhere between 1940 and 1975.

Table B-12 Projections for the Non-Occupationally Exposed Groups From Six Analyses of Pulmonary Tremolite Burden Data

Estimation Method	Version	Workers in model fitting	R^2	Predicted average in mpcf		Predicted average in f/mL[1]		
				Household-contact exposed	Neighbourh.-only	Household-contact exposed	Neighbourh.-only	Added by Household Contact
Ecological Projections	I	72	---	0.10	0.03	0.35	0.11	0.24
Ecological Projections	II	10 (trem <6 f/μg)	---	0.13	0.03	0.46	0.11	0.35
Log-linear model	I	72	.37	0.15	0.04	0.53	0.14	0.39
Log-linear model	II	36 (mpcf < 11)	.38	0.10	0.03	0.35	0.11	0.24
Biokinetic model	Logarithm. least squares	72	.24	0.49	0.15	1.72	0.53	1.19
Biokinetic model	Weighted least squares	72	.83	0.22	0.07	0.76	0.24	0.52

[1] Based on 1 mpcf = 3.5 f/mL, a conversion factor used for the study of chrysotile miners in the region [Liddell et al., 1984].

The main limitations of the data were the very small sample sizes, the large inter-individual variability of bio-accumulation, non-differential geometric measurement errors, the absence of smoking data on non-occupational cases and the inevitable selection biases of autopsy series. These problems reduced the reliability of past non-occupational projections, and induced "regression-dilution" [Smith and Phillips, 1990; Brenner, 1992; Brenner et al., 1992] bias which would lower the R^2 and tend to obfuscate true relations between independent and dependent variables. In addition to regression biases, extrapolations from typically smoking male workers to mostly non-smoking female residents of the same area could underestimate neighbourhood and household-contact asbestos exposures since a given lung burden level generally represents less cumulative exposure in a smoker than in a non-smoker due to the slower pulmonary clearance [McFadden et al., 1986a; McFadden et al., 1986b; Churg et al., 1987; Tron et al., 1987; Churg and Stevens, 1995] in smokers. On the other hand, if clearance were faster at higher exposure levels or doses [Sébastien et al., 1986], workers would tend to have faster clearance than non-workers, which would lead to an overestimation of past exposures of non-workers. The latter bias is less documented and more speculative than the smoking effect modification on biokinetics. Overall, it seems more likely that the obtained estimates were underestimated.

Finally, the non-occupational exposure estimates from a sample of Thetford Mines residents should be generalizable to the population of Asbestos. Although unverifiable, this assumption seemed reasonable in view of the similarity of the asbestos mining and milling activities in both mining areas in the past. In any case, it was left to the panel to consider this problem on the basis of all the information gathered on the two mining regions.

h) Conclusion

The biological accumulation of tremolite asbestos in the lungs of exposed workers was used to estimate the relation between past occupational exposures and asbestos lung burden, and the estimated relation was then applied to lung burden and exposure history data of non-occupationally exposed residents of the Thetford area so as to estimate their past exposure to asbestos. Despite large uncertainties in the data and in the models, reasonable estimates of past exposure were obtained: 0.07 mpcf or 0.24 f/mL for the t.w.a. exposure of neighbourhood-only exposed residents, and 0.22 mpcf or 0.76 f/mL for the t.w.a. exposure of household contacts during the years that they lived in the mining area. These estimates pertained to an imprecise period of time, probably between 1940 and 1975, during which members of the study population resided in the exposed area. Regression-dilution bias and the extrapolation from smoking workers to non-smoking residents probably led to an underestimation of past exposures of the household-contact and neighbourhood-only exposed populations.

B.1.6. Projections From Recent Measurements and Past Production

a) Introduction

There are only two types of reliable quantitative data available that are relevant to the estimation of past asbestos exposures of the study population: measurements of asbestos pollution in the mining towns and asbestos production volume figures. The former is the most relevant. Unfortunately, environmental dust and asbestos fibre levels have only been measured since 1972 and are thus too recent relative to the exposure period of interest. On the other hand, although annual production figures are available for virtually the entire century and must have been a major determinant of asbestos pollution levels, they cannot be readily translated into outdoor asbestos concentrations in the mining towns. The challenge was to combine the recent asbestos pollution data with the long-standing production volume data so as to come up with pollution estimates for the earlier period.

The relation between ambient air concentrations and production levels was assessed over the 1972-1984[1] period and was then projected on past yearly production volumes by town to estimate past concentration levels in the mining towns.

As explained below, total dust measurements seemed more reliable than asbestos fibre measurements and were preferred in the analyses and projections. However, to be able to convert dust projections to asbestos fibre concentrations, the 1972-1984 parallel dust and fibre

[1] After 1984, asbestos pollution levels were so low in the asbestos mining towns and the sampling of the outdoor air was so brief and hardly representative, that the measurement data would have added noise to the 1972-1984 period rather than information.

measurements were used to estimate a conversion factor from dust concentrations ($\mu g/m^3$) to asbestos fibre concentration values (f/mL).

b) Datasets

i. Environmental Asbestos Concentration Measurements

Asbestos fibre concentrations had been sampled and measured once per year in five asbestos mining towns[1] since 1973 by industrial hygienists of the asbestos companies[2]. The samples were taken only once per year on a dry summer day under dominant westerly winds. They were taken following the industrial hygiene NIOSH membrane filter method [Leidel et al., 1977; Asbestos International Association, 1979; Leidel et al., 1979], albeit with longer sampling periods (6-8 hours) to increase sensitivity, and using phase contrast optical microscopy (PCOM). The direct replica method and transmission electron microscopy analyses [Zumwalde and Dement, 1977; Middleton and Jackson, 1982] were used as of 1982 in parallel with the industrial hygiene samples. The data for the years 1975, 1978 and 1980 were not available for analysis because the "results were incomplete or not available" for those years. In the 1980s, there were thirty-two (32) sampling sites in the five mining towns.

The methods were crude for environmental pollution levels which are lower and much more variable over time than occupational exposures. July was a month for which production level and wind speeds were usually very low while rain precipitations were high; as a result, asbestos concentrations were probably about half[3] of the yearly average.

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1 Thetford Mines, Black Lake, Robertsonville, Tring-Jonction, Asbestos.

2 From 1973 to 1981 the sampling and analyses were conducted by the Quebec Asbestos Mining Association (QAMA), from 1982 to 1984 by the Institute for Research and Development on Asbestos (IRDA) and since 1985 by the Asbestos Institute (an association of corporate, union and governmental parties promoting the safe use of asbestos).

3 On the basis of the weekly total dust measurement taken by the Quebec Ministry of Environment, and from the analysis of the asbestos content of these dusts in 1972 in Asbestos [Denizeau, 1973] and in 1974-1975 in Thetford Mines and Black Lake [Brulotte, 1976].

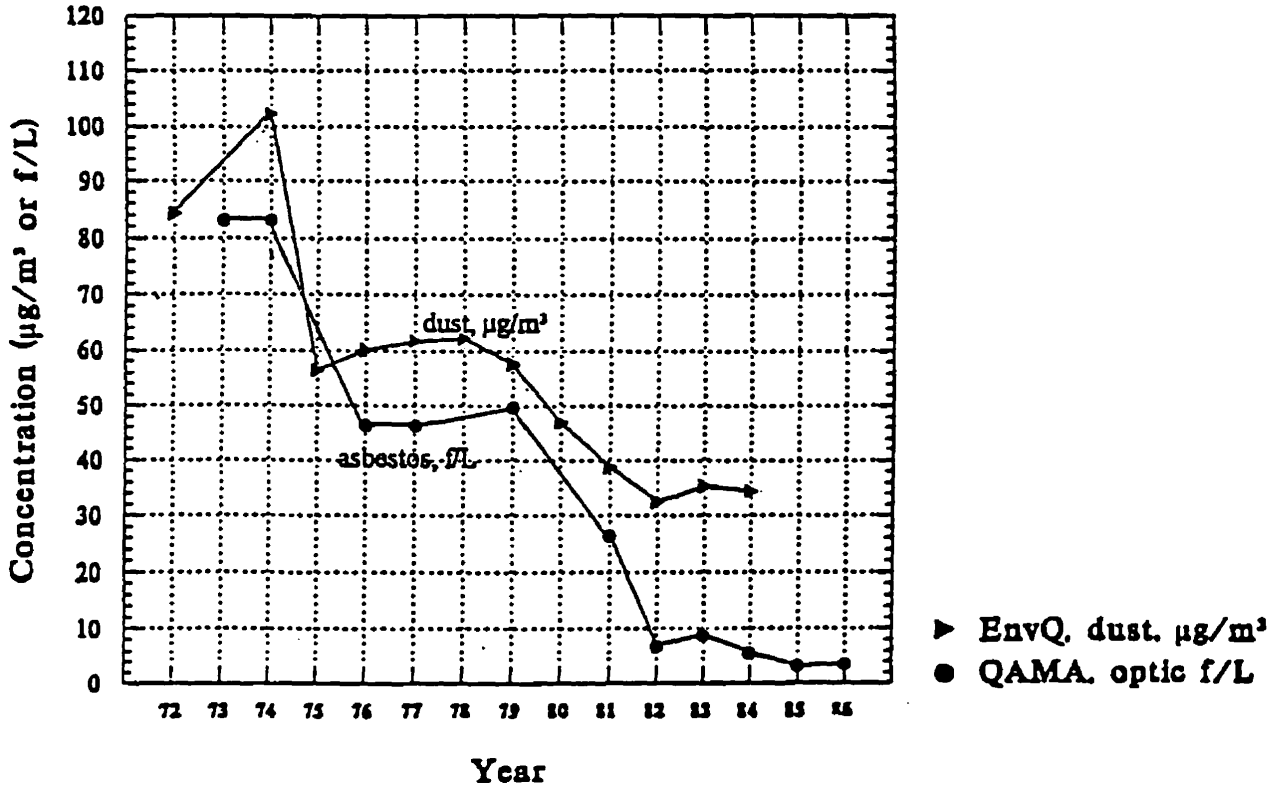
The asbestos fibre concentration data were deemed unreliable for the above reasons and were finally dismissed altogether by the expert panel. Still, regression analyses with these data produced results very similar to those obtained with total respirable dust data since the time patterns of both datasets correlated strongly (Figure B-10). Thus, the asbestos concentration data would not have changed the projections in the present Section.

ii. Environmental Dust Concentration Measurements

Total respirable dust concentrations have been collected weekly (24 hours a day, 6 days a week) by the Quebec Ministry of Environment, with Hi-Vol samplers (1.1-1.7 m^3/min) collecting particles with 10 μm > Dae > 0.3 μm [Denizeau, 1973]. They were monitored continuously for total respirable dusts since 1975 but also in 1972 in Asbestos, and since 1974 in Thetford Mines and Black Lake. Measurements were expressed in μg of respirable airborne particles per m^3 of air. There were 4 sampling stations in Asbestos, 6 in Thetford Mines and 4 in Black Lake in the mid-1970s, but the numbers of stations were halved to 2, 3 and 2 respectively in the mid-1980s. The sampling sites were originally chosen to reflect the population centroids[1] and the combined effects of emission source locations and wind directions; over the years however, the sampling stations were located where most complaints of pollution occurred and became somewhat less representative of the whole populations of the towns. The 1972-1986 data used for analyses were the annual averages by town .

[1] The population "centroid" of a town may be considered as the location in a town which minimizes the sum of squared distances between each resident's home and the centroid (a home with four occupants contributing four times to the sum of squares).

Figure B-10 Yearly Average Environmental Dust and Fibre Concentrations in the Mining Towns, 1972-1986



c) Correspondence Between Dust and Fibre Measurements

Methods

The 1972-1984 annual fibre levels by mining town were regressed on corresponding dust level averages to estimate a conversion factor from outdoor dust concentrations ($\mu g/m^3$) to outdoor asbestos fibre concentrations (PCOM f/mL) in the mining towns. The asbestos sampling may have been more representative of the towns because the hygienists took more samples (8-12 per town) over the whole urban area, whereas the dust sampling stations were located nearer to the sources or in areas where there were more fallouts. Therefore the linear regression of town-year average fibre concentrations on town-year average dust concentrations provided a conversion estimate accounting both for the numeric asbestos fibre content of the gravimetric dust concentration and for a spatial sampling correction.

Results and Discussion

Table B-13 shows that the conversion factor estimates differed by town but not significantly. The pooled conversion factor estimate of 1.0 f/L per 1 $\mu g/m^3$ with a 0.4-1.6 95%CI seemed to reflect better the real uncertainty of the data than the town-specific estimates. This rough conversion factor was the best available in the present exposure assessment. Gravimetric dust and numeric fibre levels have rarely if ever been measured concurrently in occupational or in urban settings, and neither of these settings would be readily generalizable to the outdoor environment in the asbestos mining towns. The estimated conversion factor may appear to be 33 times lower than that estimated in the EPA's risk assessment [Nicholson, 1986, p.161] as 33 f/L per 1 $\mu g/m^3$. However the two conversion factors are not at all comparable as the EPA'S factor applied to the mass of respirable asbestos fibres counted by electronic microscopy whereas the Environnement Québec measurements simply weigh all the inhalable dusts in the air.

Table B-13 Relation Between Gravimetric Dust and Numeric Fibre Levels[1] Averaged by Town, 1972-1984

	Asbestos	Black Lake	Thetford Mines	3 Towns Pooled
N[2]	8	9	9	26
R^2	.31	.60	.71	.33
Offset (f/L)	-46 f/L	-20 f/L	-56 f/L	-20 f/L
Conversion Factor (f/L per 1 $\mu g/m^3$)	1.5	0.8	2.3	**1.0**
C.F. 95%CI	0.3-3.3	0.3-1.3	1.2-3.4	0.4-1.6
Non-Asbestos Dust Background[1]	30 $\mu g/m^3$	25 $\mu g/m^3$	24 $\mu g/m^3$	20 $\mu g/m^3$

1 The dust background level at which no asbestos would remain in the outdoor air was estimated by the intercept of the regression line with the *x*-axis, corresponding to 0 f/mL.

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[1] Although measurements were in optical fibres and counted in fibres/mL, they were converted to f/L for the purpose of the regression, to prevent possible convergence or estimation problems due to the large scale difference between the dependent and independent variables and due to the fractional values of the independent variable.

[2] N: number of years for which both an annual average dust level (mpcf) and a one-day fibre sample (f/mL) were available.

The negative intercept estimated by the regression means that the 1:1 $\mu g/m^3$:f/L or 1:1 mg/m^3:f/mL conversion factor would overestimate asbestos fibre concentrations for low $\mu g/m^3$ levels; accordingly, an offset of -20 f/L should be applied for dust levels lower than 200 $\mu g/m^3$ to improve precision. Thus, a dust level of 200 $\mu g/m^3$ would imply an asbestos concentration of 180 f/L, and 100 $\mu g/m^3$ would imply 80 f/L.

d) Peak Asbestos Levels in the 1970s and 1980s

The highest pollution levels in the 1970s and 1980s may be indicative of pollution levels before the introduction of dust controls. Such values could be used as benchmarks or minima for past concentration estimates. Although asbestos fibre measurements did not represent annual levels reliably, the highest values observed with under-sensitive techniques and under-representative sampling may still be instructive about the potential for elevated pre-controls concentrations.

Similarly, pollution benchmarks were sought from maximum monthly levels measured from continuous outdoor air samples taken in 1984 by Dr. P. Sébastien [Sébastien et al., 1984; Sébastien et al., 1986] for the Canadian Ministry of Environment. The seven sampling locations were those of the Quebec Ministry of Environment. Sébastien used a Lo-Vol (110 litres/min) sampler, collecting the filters monthly over a whole year. The dust samples were analyzed using the low-temperature ashing "indirect" membrane preparation method, a transmission electronic microscope (TEM) and an energy dispersive x-ray analyzer (EDXA). Only asbestos fibres, bundles and aggregates longer than 5 µm were counted. The concentration of asbestos fibre-shaped particles with a diameter greater 0.25 µm was also estimated to approximate equivalent counts by the phase-contrast optical microscope (PCOM). Even though all modern dust controls had been installed and asbestos production was down to pre-W.W.II levels, some monthly measurements were extremely high.

The highest pollution levels measured by the asbestos companies' industrial hygienists in the 1970s and 1980s were 0.12 PCOM f/mL in Thetford Mines in 1973, and 0.10 PCOM f/mL in Asbestos in 1977. In 1984, using the best measurement techniques available, Sébastien et al. [1984; 1986] measured a few monthly average values as high as 1100 chrysotile fibrous particles (length > 5 μm) per litre in Black Lake; this corresponded to about 0.15 PCOM f/mL since 14% of the fibrous asbestos particles were large enough (> 0.25 μm) to be counted by optical microscopy. The peak was not attributed to measurement error because it was based on a month-long sample using the best sampling and analytic techniques and because other monthly values were in the same range. It is unlikely that asbestos fibre levels could have been lower before the introduction of controls, when dustiness was regularly visible, than in even the highest month in 1984 when dustiness was visible only a few days per year during breakages. Thus, annual airborne asbestos fibre levels were most likely higher than 0.15 PCOM f/mL before the 1970s.

e) Relation Between Dust and Asbestos Production Levels

A regression model was devised to estimate the relation between outdoor dust levels and asbestos production data over the period 1972-1984. The purpose was to use the fitted model to make retrospective projections of outdoor dust levels using the available pre-1972 annual production data as input to the model for each of the mining towns separately.

The following graphs illustrate the relation between pollution levels and production volume by town over time while hinting to the likely impact of improvements in dust controls. Figure B-11 shows the annual dust levels and Figure B-12 the annual production volume for each asbestos mining town for which dust measurements were available.

Figure B-11 Dust Levels in the Mining Towns, 1972-1984

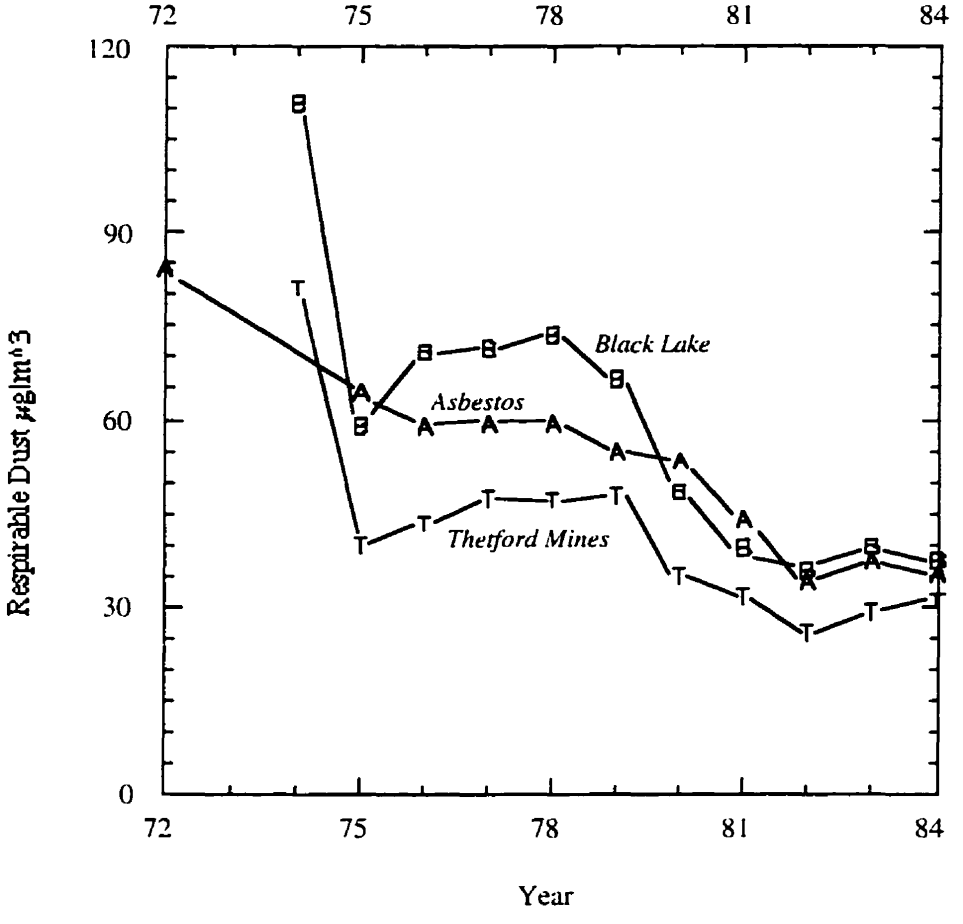
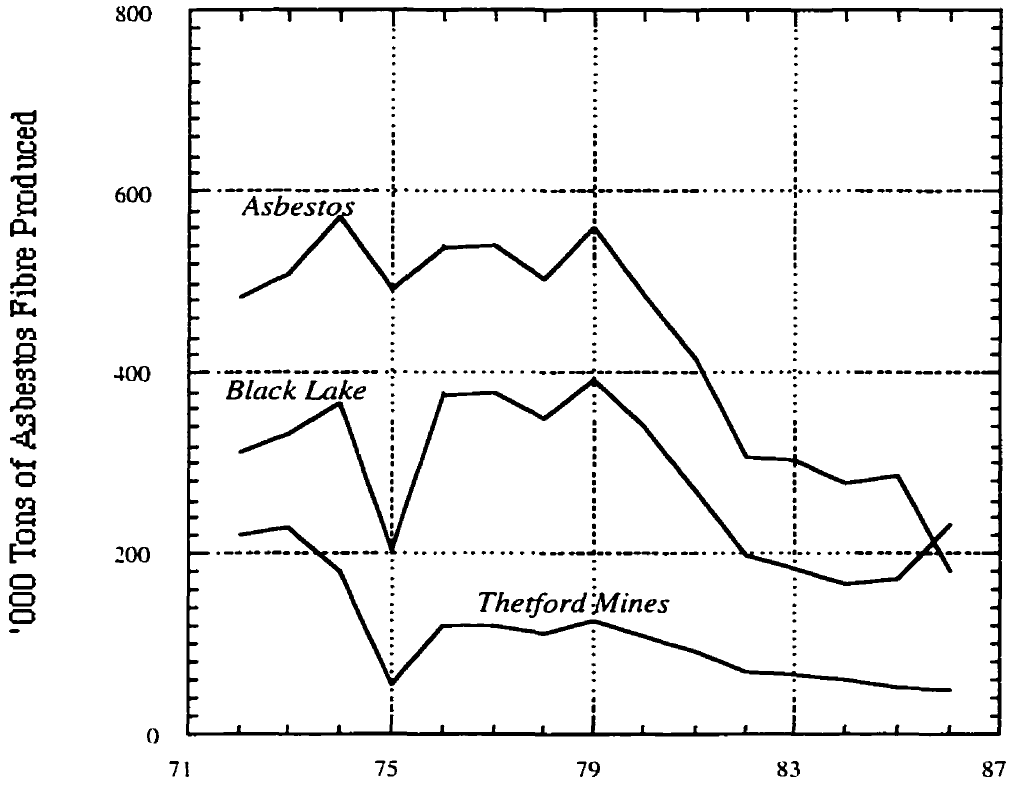


Figure B-12 Annual Asbestos Production by Mining Town, 1972-1986



The two graphs show a positive relation between dust pollution levels and asbestos production volumes in the three towns. The drop in production and dust levels from in 1975 corresponded to a seven-months labour strike in 1975 in Thetford Mines and Black Lake. However, the production 45% increase in 1976 was not followed by a similar increase in dust levels. This was the effect of the installation of major new dust control systems by many asbestos companies during the strike. To reflect the improvements in dust filtration efficiencies in a regression model, it was considered that a major step change in the dust systems had occurred in 1975 followed by a series of gradual but decreasing improvements until 1984. So changes in both production volume and emission controls were accounted for by regression to explain changes in dust concentration levels.

Methods: the Nonlinear Pollution-Production Model

According to aerosol dispersion physics, the average annual dust level in a given mining town should depend on the yearly volume of point-source emissions in that town and can be modeled as follows:

$$C_{ty} = C_{to} + E_{ty} \bullet D_{ty}$$

where C_{ty} = average concentration of respirable dust ($\mu g/m^3$) in town t in year y; ideally representing the centroid of the town's population

C_{to} = baseline background respirable dust level ($\mu g/m^3$) at the centroid of a town t, not attributable to the asbestos industry

E_{ty} = total aerosol (respirable or not) dust volume (tons) emitted by the asbestos industry over year y in town t

D_{ty} = dilution/dispersion factor ($\mu g_{\text{respirable dust}}/m^3_{\text{air}}/\text{ton}_{\text{emitted dust}}$) relating ambient respirable dusts to point-source emission volume; this complex factor depends on distance between emission sources and the town's population centroid, meteorological conditions, etc. It was assumed to be constant in the regression model.

Further, the dust emission volume itself is a function of asbestos production volume or industrial activity and of the efficiency of dust controls. So the E_{ty} term can be expressed as:

$$E_{ty} = P_{ty} \bullet r_{ty} \bullet (1 - F_{ty})$$
$$= P_{ty} \bullet r_{ty} \bullet G_{ty}$$

where E_{ty} = tons of total dust emitted in a year and town

P_{ty} = kilotons of asbestos produced in a year and town

r_{ty} = "reject fraction", ratio of useless dust aerosols (kg) generated by the production of 1 ton of commercial asbestos fibre; although this ratio varies between plants, it can be averaged over a mining town; it probably did not change significantly over time

F_{ty} = "efficiency" of filtration systems in a given year expressed as the gravimetric proportion of dusts retained by the filters

G_{ty} = gravimetric "penetrance"[1] of dust controls, the gravimetric proportion of the generated useless dusts that pass through dust controls and are emitted into the town's ambient air

Replacing E_{ty} in the first equation, the model of dust concentration can be expressed as a function of production (rather than emissions) in a given town and year as follows:

$$C_{ty} = C_{to} + D_{ty} \bullet P_{ty} \bullet r_{ty} \bullet G_{ty}$$

This simple model could not be estimated with the data at hand however, D_{ty}, r_{ty} and G_{ty} being unknown parameters that could not be estimated independently. So the product $D_{ty} \bullet r_{ty} \bullet G_{ty}$ was considered as a single parameter $\beta_{t,74}$, for year 1974, and as the product of this parameter by an abatement ratio GR_{ty} for the following years. The significant abatement of G_{ty} over the 1974-1984 period was accounted for by assuming that GR_{ty} was an exponential function of time; hence, GR_{ty} was replaced by a town-specific yearly average "penetrance ratio"[2] $gr_t^{(y-74)}$ with an exponent representing the number of years after 1974 was added to the model. The

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[1] "Penetrance" or "penetrance factor" = 1 - Efficiency, it is the proportion of dusts escaping controls. "Penetrance" is an extension of the usual term used to rate filters.

[2] Thus if gr=90%, it means that the penetrance of dust controls in any year is 90% of the penetrance in the preceding year. Conversely, it means that average yearly improvements in dust controls would abate dust emissions by 10% each year, for a given production level, i.e. would reduce emission factors by 10% each year on average. Note: penetrance *ratios* are the same as emission factor ratios seen in Section B.1.2.c, Table B-6.

supplementary dust control effort in 1975 could be accounted for by estimating an additional exponent parameter x for year 1975[1]. Thus, a town-year's specific average dust concentration level was expressed as:

$$C_{ty} = C_{to} + \beta_{t,74} \bullet P_{ty} \bullet gr_t^{[(y-74)+x\bullet(y\geq 75)]}$$

The four parameters to be estimated were: $C_{to}, \beta_{t,74}, gr_t$ and x. The model was fitted in turn with and without the x parameter.

Model-fitting was town-specific for greater validity and specificity, but the statistical estimation was hazardous due to the small sample size by town and to the lower accuracy of town-specific production volumes relative to the known volume for the three mining towns together. Finally, "mixed" partly pooled models using various combinations of first-order interactions of "town" with C_o, β_{74} and gr were also fitted on the pooled dataset.

All models were fitted statistically by nonlinear regression[2], using the least sum of squared errors loss function to fit the station-year concentration averages.

Results: Fitted Pollution-Production Model for the 1972-1984 Period

The "full model" $C_{ty} = C_{to} + \beta_{t,74} \bullet P_{ty} \bullet gr_t^{[(y-74)+x\bullet(y\geq 75)]}$ was not retained because the parameter estimate for the extra dust reduction in year 1975 (x) was unstable and not significant statistically in any analysis and because the asymptotic correlation matrices of parameter estimates suggested multicollinearity.

Table B-14 presents the parameter estimates obtained by fitting the model $C_{ty} = C_{to} + \beta_{t,74} \bullet P_{ty} \bullet gr_t^{(y-74)}$ to the 1972-1984 dust and production town-year-specific data. The estimates of the background dust level C_{to} and of the annual average penetrance ratio gr_t were not significantly different between towns; however, the rejection-dispersion parameter β was

[1] The exponent x would represent a number of years f average improvement that would be equivalent to the improvement in 1975; thus, x=5 would mean that the improvement had been 5 times more important in 1975 than in an average year between 1972 and 1984.

[2] The nonlinear regression technique has already been explained in Section B.1.4.c on lung burden modeling.

significantly different between Asbestos and the two other towns ($p<.01$), but not between Black Lake and Thetford Mines ($p=.12$). The values predicted by the town-specific models explained more of the variance (59%) of the station-year data for the three towns from 1972 to 1984 than the pooled model (49%). A mixed model with three town-specific rejection-dispersion parameters (β) fitted the pooled data nearly as well as the three town-specific models ($R^2 = 58\%$) while providing a single and more stable estimate for C_{to} and for gr. The three town-specific β parameter estimates for which the 95%CI's are given below Table B-14 were statistically different from one another at the 5% level in pairwise comparisons Tukey.

From 1974 to 1984, following the town-specific models, penetrance and emission factors would have been reduced 3.4-fold in Asbestos, 6.9-fold in Black Lake, and 3.1-fold in Thetford Mines; however, the confidence intervals overlapped considerably and were not statistically different at a 5% bilateral level. The mixed model estimated a single reduction factor of 4.0 from 1974 to 1984 (95%CI: 2.4-6.8).

Table B-14 Non-Linear Regressions of Annual Dust Levels on Production Volume and a Geometric Progression of Dust Controls, 1972-1984

Parameter	Asbestos	Black Lake	Thetford Mines	3 Towns Pooled	Mixed Model
N	49	42	63	154	154
R^2	.69	.50	.55	.49	.58
C_o	27.6	28.6	21.5	29.9	24.1
(95% CI)	(17-38)	(8-49)	(9-34)	(25-34)	(16-32)
β_{1974}	0.090	0.159	0.204	0.111	Asb.: 0.101* Bl. L.: 0.155* Th. M.: 0.215*
(95% CI)	(.07-.11)	(.11-.21)	(.15-.26)	(.09-.13)	
gr	0.886	0.824	0.893	0.867	0.871
(95% CI)	(.84-.94)	(.70-.95)	(.79-1.00)	(.83-.90)	(.83-.92)

* 95%CI's of the β estimates in the mixed model were:

.08-.12 for Asbestos, .13-.18 for Black Lake, and .16-.27 for Thetford Mines.

Fitted Pollution-Production Models :

Full Model: $C_{tv} = C_{to} + \beta_{t,74} \cdot P_{tv} \cdot gr_t^{(y-74)}$

Pooled Model: $C_{tv} = C_o + \beta_{74} \cdot P_{tv} \cdot gr^{(y-74)}$

Mixed Model: $C_{tv} = C_o + \beta_{t,74} \cdot P_{tv} \cdot gr^{(y-74)}$

Variants of the mixed pollution-production model in Table B-14 were applied to past annual production data under four scenarios. In Scenario A, the mixed model was applied postulating that the average annual emission abatement factor gr was the same in each town between the outset of dust emission controls and 1974 as from 1974 to 1984. Scenario B was similar to Scenario A, except that the gr factor used was the upper 95%CL of the parameter estimate. Scenario C applied the mixed model to each town except that the penetrance ratio expression gr_t was replaced by the annual geometrical apportionment of the 1945-1974 emission factor ratio of 35 derived in Table B-6: $gr = 35^{-1/29} = 0.885$. Scenario D was based on my own estimation of minimum progress in emission factors; given that dust emission abatement had been 4-fold from 1974 to 1984, I presumed that the installation of baghouses on both dryer and mill in Asbestos in the 1950s had abated emissions 40-fold over the 1954-1974 period; in Black Lake, which had the first baghouses installed on some operations and whose production increase was based on new plants with modern technology, I supposed a 16-fold abatement over the 1945-1974 period; in Thetford Mines, where many different control systems were tried in the 1960s, I supposed a 4-fold abatement between 1958 and 1974.

Table B-15 PY-Weighted 3-Town Average Dust Concentrations Estimated Under Three Scenarios Based on the Mixed Pollution-Production Model, 1910-1984

Year	Scenario A : $gr_t = 0.87$ Units = μg/m^3	Scenario B : $gr_t = 0.83$ [1] Units = μg/m^3	Scenario C : $GR_{t,1945} = 35$ Units = μg/m^3	Scenario D: "minimal" Units = μg/m^3
1910	120	291	279	78
1920	205	527	505	126
1930	259	655	679	174
1940	389	966	1027	245
1945	444	1167	1176	297
1950	660	1565	1944	579
1955	662	1469	1053	570
1960	390	738	443	310
1965	241	357	260	199
1970	121	141	125	112
1974	80	80	80	80
1980	43	38	42	46
1984	30	28	29	32

Notes:

1: The 1:1 μg/m^3:PCOM f/L conversion can be applied to the figures in Table B-15.

2: $GR_{Asb,1954} = 40$; $GR_{BLL,1945} = 16$; $GR_{THM,1958} = 4$.

[1] $gr_t = 0.83$ was the upper 95%CL of the mixed model.

f) Discussion

According to the four scenarios, projected past pollution levels (Table B-15) were highest around 1950, being about 8 times higher than in 1974 according to Scenario A, 20 times higher according to Scenario B, 24 times higher according to Scenario C, and 7 times according to the "minimal" Scenario D.

Scenarios C and B were relatively close and projected the highest past outdoor pollution levels. Both scenarios implied that emission factors improved 9 times more from 1945 to 1974 (abatement ratio of 35) than from 1974 to 1984 (abatement ratio of 4). The former period was 3 times longer and was characterized by controlling emissions from mills whereas the latter shorter period's major improvements were on dryers. Given that "uncontrolled" mills emitted 8-20 times more dust in the outdoor air than dryers (cf. Sections B.1.2 and B.1.3), the ratio of 9 was quite conceivable. In addition, Scenario C was based on an empirically derived emission factor ratio. For these reasons, the projections under scenarios B and C were plausible .

Scenarios A and D were relatively close and projected the lowest past outdoor pollution levels. These scenarios implied that emission factors improved 3 times more from 1945 to 1974 than from 1974 to 1984. In view of the argument of the mill vs. dryer controls eras, and since the assumptions underlying Scenario D were contrived to be conservative, the lower past pollution projections based on these scenarios may be considered as plausible.

All four scenarios were biased toward underestimating past pollution levels by not accounting for the diluting effect of urban expansion. Thus, the population centroids in Asbestos and in Thetford Mines were at least 500 meters closer to asbestos emission sources in 1945 than in 1974 (Section B.1.1.a); thus the population was more exposed by the single fact of living closer to the sources in the past. Aerosol dispersion modeling for year 1945 suggested an

approximate gradient of 0.1 f/mL for each 100 m distance from the emission sources; thus pre-1955 dust levels may have been underestimated by 0.5 f/mL.

Finally, the 0.15 f/mL minimum suggested in Section B.1.6.d should probably apply to all town-specific estimates before 1970 back to 1900 when year-round production with fiberizing plants was already going on in each town.

g) Conclusion

The global estimates for the 1945-55 period ranged from 300 $\mu g/m^3$ to 1900 $\mu g/m^3$ or from 0.3 f/mL to 1.9 f/mL. Adjusting for the urbanization effect, the range of plausible estimates for 1945-1955 should be closer to a 0.8-2.4 f/mL range.

B.1.7. Summary and Discussion of the Neighbourhood Outdoor Exposure Evidence

Clearly, the populations of the asbestos mining towns were exposed to very high environmental asbestos concentrations, at least until the 1970s. The most important sources of respirable asbestos fibre pollution were always the mills. While dust control measures undoubtedly reduced air pollution during the period from 1945 to 1974, this was to some extent counter-balanced by the increasing production volume during this period. The following factors contributed to make the mining towns very dusty and asbestos-polluted until the mid-1970s at the least: the geographic relief of the areas, localization of dwellings close to and mainly downwind from asbestos emission sources, incomplete dust control until the mid-1970s, the dependence of the local population on the asbestos industry, and the aerosolizability and respirability of the asbestos emissions.

The relative magnitude of asbestos pollution between the three mining towns changed over time. Thetford Mines was the most important asbestos production centre before the 1950s but became relatively less important thereafter. In Thetford Mines and Black Lake, the population lived closer to asbestos emission sources and was more affected by emissions from tailings piles than in Asbestos. The populations of Asbestos and Thetford Mines lived closer to emission sources before 1960 than after 1970. Dust controls were implemented earlier in Asbestos than in the other mining towns. Recalls of visible dust deposits reported in our survey of the older population gave a similar picture; from the mid 1950s to the 1980s, asbestos pollution was probably lowest in Asbestos and highest in Black Lake. Before the 1950s however, recalls of daily visible deposits were reported as frequently in the three towns,

even though pollution levels should have been higher in Thetford Mines than in the two other towns according to asbestos production volumes.

Table B-16 summarizes the main results of the different components of the exposure assessment. The upper part of the table abstracts a few highlights of the qualitative and background data, whereas the lower part summarizes the results of the five exposure substudies. The order of the item numbers (#) represents my personal opinion about the validity of each of the five exposure evaluation approaches. These assessments were quantified in various ways, and more estimates were quantified as ratios than as absolute pollution level estimates. Accordingly, all quantitative data are presented in terms of ratios of a year's estimate relative to year 1974; 1974 was the linchpin year between the better documented recent period and the past periods of interest[1]. When obtained with a given approach, the absolute outdoor level estimate appears in parentheses. The ratio estimates and ranges presented at the bottom of the table are not uniformly probable values since the validity and reliability of each approach and estimate is idiosyncratic. The ranges of outdoor levels result from the application of the ratio ranges to the .08 f/mL estimate for 1974.

Four exposure evaluation approaches estimated *pre-controls* outdoor concentrations 7 to 30 times higher in 1945 than in 1974. The lung burden analyses estimated merely a 3.1 ratio relative to 1974, but the lung burden-based estimates were average exposures over the whole period that the subjects lived in the area, i.e. from 1940 to 1980 for most subjects in the lung burden datasets and were thought to be more representative of the middle of that period, around 1960.

The pollution-production approach (#1 in the table) should probably be the backbone of the whole exposure assessment because it was based on objective year-town specific production data over the whole century, on actual environmental air pollution measurements and on transparent assumptions and models. This approach provided both a complete historical

[1] Since lung burden analyses provided no estimate for 1974, the value of 0.08 f/mL estimated from the pollution-production regressions was used as the reference for 1974.

perspective and a means of estimating absolute asbestos and dust levels. Next came the estimates based on engineering calculations (#2) and on the aerosol dispersion model (#3). The strongest point of approach #2 was the estimation of emission factors for mills and dryers in 1945 and in 1984; however the secondary data for 1974 was not substantiated explicitly enough. The aerosol dispersion modeling simply converted the emission factors estimated with approach #2 into dust concentration levels for 1945; it was validated by dust level data for year 1972; unfortunately the 1945 estimation was limited to the town of Asbestos. The three approaches were distinct but their estimates were largely based on the same piece of evidence for year 1945 and thus did not provide independent estimates for that year.

The recall survey of past visible asbestos dust deposits (item #4) was reliable statistically and the eyewitness recounts were quite consistent with known spatial and chronological factors and trends. However, this approach could not quantify past concentrations; 50% recalls of deposits cannot be said to represent 10 times higher asbestos levels than 5% recalls. Nevertheless, ratios are presented in the table merely to provide an independent qualitative corroboration of the quantitative approaches.

The lung burden approach (item #5) was based on simple biokinetic principles. Its two merits were the use of direct vestiges of past exposures (asbestos fibres retained in lung tissues), and the use of data and methods completely separate from those used in the other approaches. However, its results were statistically imprecise and pertained to an indefinite and extensive time period.

No approach other than the aerosol dispersion modeling and the recall survey took into account that population centroids were closer to the mills and dryers in 1945 than in 1974; the estimates should therefore be adjusted upwards. Since the aerosol dispersion simulation showed a gradient of about 0.1-0.2 PCOM f/mL per 100 m for the 1945-1950 period in Asbestos, the 1945 estimates should be augmented by some 0.5-1.0 PCOM f/mL. This adjustment should be made to the 1945 estimates in items #1, #2 and #3. It was applied to the asbestos f/mL ranges at the bottom of the table.

Table B-16 Summary of Neighbourhood Exposure Evidence

Note: In the table, absolute numbers represent ratios of a year's estimated level to that estimated for 1974; i.e. 1974 is the baseline.

Estimation base (section)	Approximate Year of the Projections: **1984** Clean-Air Era	**1974** Mill+Dryer Controls +	**1960** Many Mill Controls +	**1945** No Controls
Spatial patterns and trends: estimated 75% radius (B.1.1.):	most dispersed 3.4 km	dispersed 2.0 km	less dispersed 1.5 km	tight 1.25 km
Industrial process. (B.1.1.):	rare breakages	occasional breakages	frequent breakages	frequent breakages
air/output ratio vs.. 1974:	1.0	1.0	0.90	0.90
Dust control systems (B.1.1.)	regulatory + proper maintenance	governmental controls, dryer baghouses	some baghouses, mostly mills, poor maintenance	settling rooms (float shed)
Production volume in asbestos towns (B.1.1.)	0.47	1	0.76	0.32
Gravimetric emission factor ratio (B.1.2.)	0.025	1	n.a.	35
#1: Fibre concentrations based on recent pollution and past production data (B.1.6.)	0.38 (0.03 f/mL)	1 (0.08 f/mL)	6-12 (0.5-1.0 f/mL)	4-30 (0.3-1.9 f/mL)
#2: Numeric respirable dust emission ratio based on engineering data (B.1.2)	0.14	1	n.a.	8
#3: Aerosol dispersion for 1945 based on dust retained by baghouses (in Asbestos) (B.1.3.)	n.a.	1 (0.1 f/mL)	n.a.	17 (1.7 f/mL)
#4: Recall ratio of weekly visible deposits (B.1.4.) (PY%)	n.a.	1 (4%)	8.5 (34%)	17 (68%)
#5: Lung burden projection from 72 workers to 22 in "environmental group" (B.1.5.)	n.a.		3.1[1] (ratio range: 1.3-6.6) (point estimate: 0.25 f/mL)	
Range of RATIOS (vs. 1974)	< 0.4	1	3 - 12	4 - 30
Range of PCOM f/mL estimates (3-town averages)	< 0.03 f/mL	0.08-0.1 f/mL	0.2-1.2 f/mL	0.3-1.9 f/mL
Range of f/mL estimates "adjusted" for urban spreading	< 0.03 f/mL	0.08-0.1 f/mL	0.4-1.4 f/mL	0.8-2.4 f/mL

[1] Estimate applies roughly to the 1940-1980 period, since it represents average lifetime exposure. It is not specific to a year, decade or era even. It may be assigned to 1960, the mid-point of the 1945-1974 period. The ratio of 3.1 = .07 mpcf x 3.5 f/mL/mpcf ÷ .08 f/mL for 1974.

In conclusion, average environmental exposure levels in the three towns were estimated in the range of 0.08-0.10 f/mL in 1974, 0.4-1.4 f/mL in 1960 and 0.8-2.4 f/mL in 1945. Emissions and exposures must have peaked around 1945-1950, at the outset of dust emission controls and before the major urban expansions. From the mid-1950s to the mid-1970s, airborne asbestos levels declined very rapidly in Asbestos, moderately in Thetford Mines and slowly in Black Lake. After 1974, asbestos concentrations declined rapidly in all three mining towns.

B.2. SYNTHESIS OF THE EXPOSURE DATA BY A PANEL OF EXPERTS

B.2.1. Introduction

The linchpin of the exposure assessment process is the synthesis of the above exposure data into a final quantitative evaluation of past town-year-specific exposure levels by a panel of experts in the area of environmental asbestos exposure measurement. The use of expert panels to review and summarize exposure assessment, epidemiological or toxicological data and to make recommendations is a common practice in regulatory and health research agencies (e.g. Health Protection Branch of Health and Welfare Canada, U.S. Environmental Protection Agency, U.S. Consumer Product Safety Commission, U.S. National Research Council, W.H.O. International Program for Chemical Safety, W.H.O. International Agency for Research on Cancer, U.K. Health and Safety Commission, etc.). Because the above exposure substudies were based on methods that were diverse and complicated and on data that were indirect, partly qualitative and fragmentary, the task of reviewing was very difficult. We established a panel consisting of experts with long-standing experience and international scientific recognition in measuring asbestos fibre concentrations in various settings.

All five candidates approached accepted to participate in the panel. These were:

- Dr. Bruce Case, MD.., M.Sc., pathologist and epidemiologist, Royal-Victoria Hospital, McGill University: lung burden studies.
- Dr. Morton Corn, Ph.D., industrial hygienist, professor, Johns Hopkins University: asbestos risk assessment, occupational and environmental exposure studies;
- Dr. Graham Gibbs, Ph.D., epidemiologist and geologist, consultant, Alberta: asbestos risk assessment, occupational and environmental exposure and epidemiological studies;

- Dr. William Nicholson, Ph.D., geophysicist, professor, Mount Sinai School of Medicine in New York: asbestos risk assessment, occupational and environmental exposure and epidemiological studies;
- Dr. Patrick Sébastien, Ph.D., physicist, director of scientific research, Occupational Health and Safety Research Institute (IRSST), Montreal: short-term and long-term environmental exposure and lung burden studies;

The panel was convened for a two-day meeting to review the exposure evidence presented here above, to weigh the qualitative and quantitative data and to make its own estimation of past exposure levels on the basis of that evidence. Some local experts were also present at the panel meeting as "witnesses" and consultants to answer panel's questions and to supplement, clarify or correct certain points in the presentation. The local experts were: Mr. Mike William, (engineer, Director of Production JM Asbestos, President of the Quebec Asbestos Mining Association), Mr. Serge Turcotte (industrial and environmental hygienist, Environment Quebec service located in Thetford Mines, responsible for the control of environmental standards on asbestos in Quebec), and Mr. Elphège Thibodeau, (engineer, director, hygiene & safety, JM Asbestos).

The objective was to estimate the annual average environmental levels of ambient airborne asbestos exposure of the asbestos-mining populations during the period from 1900 to 1989. In effect, the assessment focused on key years representing different asbestos pollution eras or turning points in dust controls and on the three main mining towns. The panel was asked to provide best estimates of exposure levels with lower and upper plausibility limits for each estimate. Two such sets of estimates were to be estimated: one for neighbourhood outdoor exposure levels, and one for household-contact indoor levels attributable to living in the household of an employed asbestos worker.

B.2.2. Methods and Panel Deliberations

a) Preliminaries

A two-day meeting was held in February 1994 in Montreal. The methods and results presented above were summarized and sent to panel members one week before the meeting.

The panel thought that an open discussion and consensual decision process would be more valid than averaging individual independent assessments by panel members. The presentation and critical review of the presented material lasted a day and a half. Basically this consisted of my presenting to them all of the material shown in Chapter B.1 above, along with critical discussion of the pros and cons of the different datasets, approaches and analyses. The deliberations and finalization of the assessment by the experts took half a day. The local experts were present on the first day and intervened freely in the discussion.

The operational objective for the panel was set as follows: to estimate the concentrations of asbestos fibres > 5μm visible by optical microscopy per milliliter of air in the town environment for the average female resident of each town, i.e. living at the "centroid" of a given town during a given year.

When faced with many measurements as for the 1974-1984 period, the panel preferred the arithmetic mean to the geometric mean of environmental concentrations. This concept underlied the panel's estimates of past asbestos concentrations. Arithmetic exposure measurement averages seem to reflect more adequately average amounts of dust respired. Even when the distribution of exposure concentrations or dose in the study population is

lognormal within each group compared, geometric mean exposures or doses will result in a biased estimate of risk if the between-group exposure-effect relation is linear [Seixas et al., 1988; Armstrong, 1992] or sublinear. Since this thesis' null hypothesis and the prevailing conception in the scientific community assume a linear asbestos-lung cancer relationship, arithmetic mean exposure estimates seemed most appropriate.

Ideally, the panel should have evaluated past exposure intensities not only for each of the three main mining towns but also for each neighbouring municipality in the agglomerations comprising the mining towns. However the exposure information was much scarcer for neighbouring municipalities, and in any case, most of the population of interest was concentrated in the three mining towns. Furthermore, the panel had limited time. For these reasons, the panel's exposure assessment was limited to the towns of Asbestos, Thetford Mines and Black Lake.

Similarly, to make the task manageable, the panel estimated environmental exposure levels for only four (4) key years. *1984* was the most recent and best documented year with reliable and valid environmental exposure measurements which enabled a relatively precise estimate. In addition, the 1984 data could shed light on the weights to be attributed to different data elements. *1974* was a pivotal year in estimating past from present exposures, being both the earliest year for which environmental measurements were available for all three mining towns and the last year before the major organized effort to clean up the outdoor air of the asbestos mining towns under the control of governmental surveillance and regulations. *1945* was a watershed year in that it was near the end of the pre-controls era. Finally, *1960* was chosen because it was midway between the pre-controls era and the post-1974 modern controls era; baghouses had been installed on the mills and dryer in Asbestos but were on their way in Thetford and Black Lake. Estimates for 1960 provided a basis for the interpolation of values between 1945 and 1974. The panel agreed on the choice of these four key years.

The panel decided to aim first at "best estimates" for the key town-years and to leave aside the issue of subjective "confidence" limits for each estimate until after all the best point estimates had been quantified.

b) Neighbourhood Outdoor Exposure Estimation Process[1]

The panel first made estimates for the three main mining towns for year 1984 on the basis of Sébastien et al.'s year-long survey [Sébastien et al., 1986]. The maps were used to adjust for the location of sampling stations with respect to the population's centroid in each town. Then, using the 1974-1984 town-specific dust pollution-production regressions to estimate how much more asbestos dust was in the air of the main mining towns in 1974 relative to 1984, the panel multiplied its town-specific estimates for 1984 by the town-specific 1974/1984 asbestos dust ratios. The results were the panel's estimates for 1974. The estimates for year 1945 came next. Town-specific dustiness ratios between 1945 and 1974 estimates were estimated from the visible-deposit-recall survey, the production-based projections, the engineering-based calculations and, to a lesser degree, the lung burden-based estimations. Each panelist used his own weighting of these data and even analogous situations where other pollutants had been controlled in other cities.

[1] A more detailed description of the estimation process for each key year is available in **Appendix B19.**

Table B-17 Panel's Final Outdoor Exposure Estimates, PCOM f/mL

Town	Year 1984	1974	1960	1945
Asbestos	0.010	0.035	0.100	1.000
Plausible range	.003-.030	.012-.105	.033-.300	.333-3.000
Thetford	0.007	0.049	0.375	1.000
Plausible range	.002-.021	.016-.150	.125-1.125	.333-3.000
Black Lake	0.047	0.141	0.600	1.000
Plausible range	.016-.141	.047-.423	.200-1.800	.333-3.000
PY-weighted average[1]	0.021	0.075	0.365	1.000
Plausible range	.007-.063	.025-.225	.112-1.095	.333-3.000

[1] The three-town averages indicated in the table are simply weighted averages of town-specific estimates for a given year, the weights being the proportion of the study base in each town over the whole study period (1970-1989).

The panelists came up with individual but converging estimates for 1945. They chose 1 f/mL as a ballpark figure for all three mining towns. Finally, concentrations for year 1960 were estimated by interpolation while accounting for town-specific implementation of baghouses and other dust controls.

To represent the uncertainty of its subjective estimates, the panel assigned a plausibility interval around its point estimates. It thought that it was very unlikely that the true concentration values could more than three times larger or smaller than the estimated values. More positively, the panel felt that the true concentration values should fall within these 3-fold bilateral intervals. Table B-17 summarizes the panel's final estimates of past outdoor asbestos levels.

Estimates for all other years between 1900 and 1984 were interpolated or extrapolated by myself following the principle accepted by the panel that the estimates should be modulated by the production volume. I assumed that there was a continuous exponential progress of the ratios of asbestos concentrations to asbestos production levels between key years. The following formula was applied for each town:

$$C_{ty} = P_{ty} \bullet \frac{C_{tx}}{P_{tx}} \bullet \left(\frac{C_{tz}}{P_{tz}} \bullet \frac{P_{tx}}{C_{tx}} \right)^{\left(\frac{y-x}{z-x} \right)}$$

where C_{ty} is the airborne asbestos dust concentration estimate, for town t, in year y

P_{ty} is the asbestos production volume, for town t, in year y

y is the year for which the interpolation is being made,

x is the *key* year preceding year y,

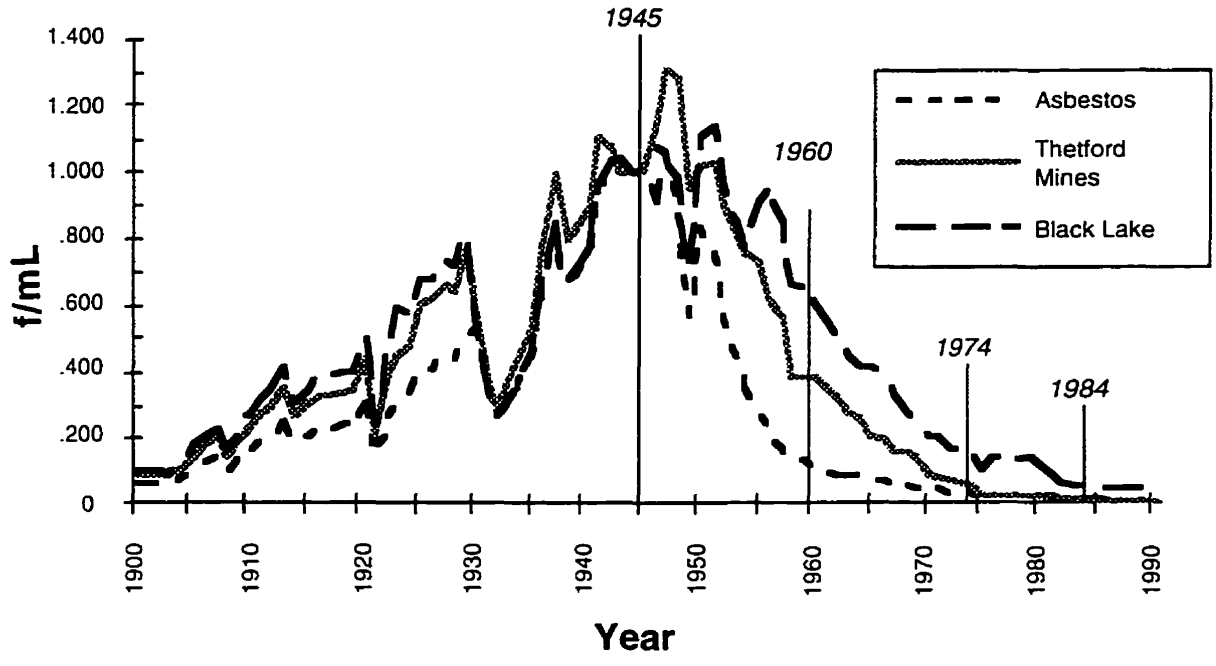
z is the *key* year following year y.

For the pre-1945 period, it was assumed that the pollution rate was constant (no significant changes in dust controls or production process during the pre-controls era), reducing the formula to:

$$C_{ty} = P_{ty} \bullet \frac{C_{t,'45}}{P_{t,'45}}$$

The graph in Figure B-13 shows the production-based interpolations for each of the three towns according to the interpolation formula stated above. The concentration levels in the other mining towns (Robertsonville, Vimy-Ridge, East Broughton) where members of study agglomerations had previously lived were estimated by assuming that the average concentration/production ratio of the three main mining towns applied to each of the other mining towns. Thus, to estimate an asbestos concentration level in a given year in another asbestos mining town, that town-year's production volume was simply multiplied by that year's average concentration/production ratio for the three main mining towns. When there was no asbestos production in an asbestos town, the background level was assumed to be 0.001 PCOM f/mL, a level 10 times higher than in today's cities' high asbestos pollution TEM levels. In the referent areas which exclude Quebec's largest cities, there have been some albeit not important asbestos industries, and it was presumed that asbestos levels measured in 1984 by Sébastien et al. in Saint-Étienne were 3 times higher than 100 years ago, and that levels had increased exponentially over that period.

Figure B-13 Interpolations Obtained From the Panel's Final Environmental Exposure Estimates



c) Household Exposures

At the time of the expert panel meeting, the lung burden-based estimates of household exposures presented to the panel were similar to albeit higher than those presented above. The estimates were based on over-parameterized regression models, resulting in multicollinearity which prevented correct adjustment for the distance differential between the household-contact and neighbourhood-only exposure groups. The ratios were larger than 6.8, which seemed excessive to at least one panelist who thought that housekeeping activities and cleaning asbestos-laden work clothes in the house could not account for more than a 2- or 3-fold ratio. Three panelists seemed to contemplate a 5-fold ratio as a plausible value. In fact, two of these (Sébastien and Case) and myself have recently discussed the household exposure issue and we now agree that it should add to rather than multiply neighbourhood exposures; the original multiplicative approach must have distorted the discussion. The panel agreed that washing and dusting clothes must have been a significant source of increased exposure in wives of asbestos workers, but they could not agree on how much exposure could be contributed by such activities. In the end, the panel did not want to make a blind estimate.

The problem remained ours (J.S. and M.C.) to decide. This issue and more indirect data sources are presented in Chapter B.3, where past household exposure is re-assessed as an additional rather than a proportional or ratio increase relative to neighbourhood levels.

B.2.3. Discussion

The backbone of the assessment was the coupling of recent environmental measurements with production data which bridged the present with the past. There was consistency between the different data sets and data analyses. The panelists used just about every piece of data presented to them at some point as they went through the assessment for the four key years; lung burden data was the least useful.

Despite the frailty and sparseness of the available data, the panel arrived at a consensus quite readily on past exposure estimates for each asbestos-mining town for the four key years. The panelists were confident in their yearly estimates within the stated plausibility intervals. These intervals were relatively close given the lack of direct measurements.

However, the final exposure assessment had some limitations. In their calculations for 1974, the panel may have forgotten to take into account the change in production volumes between 1974 and 1984, thereby underestimating the exposure levels in 1974 by a factor of 2 at the least (see Table B19-1 in Appendix B19). Correcting 1974 might have changed the panel's estimates for 1960, but likely not for 1945 however. Therefore, it would have had a small impact on the overall cumulative exposure estimate.

Assigning an identical 1 f/mL exposure level in all three towns for 1945 was as much a practical compromise between different views as a consensus and it entailed some inconsistency. Assigning the same exposure level to the three towns for 1945 was mostly justified from the visible dust deposit recall survey, where the respondents recalled less frequent visible dust depositions in Thetford Mines over the pre-1945 era. However, the contrary would be expected on the basis of production volumes, wind factors and proximity of

emission sources. As discussed in Section B.1.4.e on past visible pollution, underground mining in Thetford, and measurement and recall errors would not likely explain this paradox satisfactorily. In view of the production-based projections (Section B.1.6), Thetford's pollution levels might be underestimated by a factor of up to 2 relative to the two other mining towns over the 1900-1960 period for which 1945 was a fulcrum in the application of the interpolation formula. The error was relative in that either pollution was either underestimated in Thetford Mines or overestimated in Asbestos over that period. The expected difference was corroborated explicitly by one panelist in the "macroscopic analogy with Pittsburgh" below.

More consequential, town-year asbestos exposure levels before 1945 were probably underestimated by projecting pre-1945 exposure levels as simple production-based proportions of the 1945 estimate. As mentioned in Section B.1.6.f, by making no adjustment for the much smaller urban areas nor for the lower heights of emission sources further back in time, I would conjecture that the assessment of past exposure levels probably underestimated the true pre-1945 exposure levels by 0.5 f/mL or more.

Moreover, all the residents of the mining towns may have had higher personal exposures than would be indicated by ambient outdoor concentrations alone. Children crawled, walked on all fours and played on dust laden floors and lawns; in 1989, elderly women told me about leaving their toddlers play on grayed lawns; older kids played and slid on asbestos tailings piles [Findley et al., 1984], etc. Unfortunately, these more intimate contacts of residents with asbestos dust were not brought to the panel's attention, nor did panel members raise these issues.

There is no other similar exposure assessment to compare our results with. However, one panelist used a macroscopic analogy with major emission controls introduced in Pittsburgh between 1930 and 1980 to see how much reduction in particulate concentrations could be expected by passing from an uncontrolled to a controlled environment. The calculated dust reduction ratios were in the 17-50 range for total suspended particulate matter ($\mu g/m^3$) and 27-100 for SO_2. Applying the two maximum factors (50 and 100) to QAMA's 1984 data and

adjusting for changes in production between 1945 and 1984, he obtained estimates of roughly 200-400 f/L in Asbestos, 700-1400 f/L in Thetford Mines, 1120-2250 f/L in Black Lake, and 740-1480 f/L for the three-town average in 1945. Otherwise, a recent letter published by Rogers [Rogers, 1990] mentioned that there could have been more than 2 PCOM f/mL outside a crocidolite plant in Australia before the 1960s on the basis of re-analyzed thermal precipitator slides of dust samples; however the new analyses, methods and results have not yet been published.

The main shortcoming of the exposure assessment was that household exposures had not been assessed by the expert panel.

B.2.4. Conclusion

The panel produced outdoor asbestos fibre level estimates for each town of the asbestos-mining areas for each year from 1900 to 1984, with a bilateral 3-fold geometric plausibility interval for each estimate. The panel members agreed relatively easily on estimates that were themselves in good agreement with the estimates obtained from the various datasets. All the data presented to the panel had some use, although lung burden contributed somewhat less to the estimation process.

Although asbestos production peaked in the 1970s, neighbourhood outdoor exposure levels would have peaked in the 1940s around 1 f/mL and would have decreased monotonically from 1950 on. Outdoor levels were deemed to be highest in Black Lake over most of the century, particularly after 1950. The town of Asbestos would have had the lowest exposure levels over most of the century. However, exposure levels estimated for Thetford Mines in the pre-1945 era were probably underestimated by the panel. The estimated levels for earlier times were probably underestimated for the three mining towns because the estimation did not consider sufficiently the effect of the changing spatial size of municipalities. The panel may have miscalculated the exposures in Asbestos and Black Lake in 1974. In practice, the panel's outdoor exposure estimates seem to have been more focused and more precise on the three-town global average than on mining-town-specific pollution levels. Childhood contact with asbestos might have been more intense than indicated by ambient air levels assessed more than 2 meters from ground; this was not considered by the panel. Together, various biases suggest that the actual outdoor exposure levels have been more likely underestimated than overestimated, and more so in Thetford Mines than in the two other mining-towns.

Because of the sparseness of the data that were presented to them at the time of the meeting, the panel did not make quantitative estimates of exposure due to household contact with asbestos workers.

B.3. HOUSEHOLD-CONTACT AND OTHER INDOOR EXPOSURES

B.3.1. Introduction

Nowadays, asbestos workers leave their dirty working clothes at the mine or mill, and most take showers before returning home. Before the 1970s however, most asbestos workers had no showers at the workplace and brought their work clothes to and from home; mining pit workers were dusted with air pressure hoses before returning home but millers were not. Most asbestos workers would carry asbestos dusts on their work clothes and on their hair and skin directly into their households where the dust might settle and re-aerosolize many times due to housework such as brooming the floors, dusting of furniture and curtains, mending, dusting and washing of asbestos workers' clothes, and other dust movement due to running children, etc. According to our population survey (Chapter B.4), two thirds of the asbestos-mining towns' female population have experienced such household-contact exposure, on average for about half of the years lived in a mining town.

Independently of household-contact exposures, the real exposures of female residents of the mining towns may have been higher than outdoor exposure levels because of their housework activities. For instance, in 1920, housewives were carding and spinning asbestos wool from which they knitted socks and mittens [Fortier, 1983a]; the picture in Figure B-14 shows a woman carding asbestos wool in 1930. In 1989, elderly female residents told us about having knitted asbestos centerpieces in the past, insulating windows and pipes themselves with asbestos picked up on the ground in and around their homes, etc. They often picked off the clothes-hanging lines laundry laden with asbestos fallouts, a problem that Thetford Mines housewives complained about at least since 1912 [Cinq-Mars et al., 1994]. In fact, all housework in asbestos-polluted neighbourhoods may have exposed housewives and most female residents to higher levels than outdoor levels.

Given the high prevalence of household exposure, the assessment of household exposure was crucial for the exposure assessment. As indicated in Section B.1.5, the dataset was very small, the analyses were intricate, and the exposure panel did not agree on how to interpret household-to-neighbourhood exposure ratios which resulted from log-linear regressions. Also, there was not a clear concept of the various components of household exposure, and thus panel members erroneously opposed a) the building-filtration effect which attenuates indoor levels relative to outdoor-induced exposures to b) the additional pollution contributed by the presence of an asbestos worker in the household. The panel wanted more information on the period during which workers brought dirty clothes home, on the chores of housewives and their specific exposure circumstances, on the possible levels and frequency of peak household exposures, etc.

The present chapter addresses the panel's concerns as well as the ill-defined notion of household-to-neighbourhood exposure ratios when the difference in exposure levels must have been additive. All available direct and indirect evidence about household exposures and housework-related asbestos exposures is reviewed: 1) air samples taken in asbestos workers' households, 2) air samples taken in other indoor exposure circumstances such as offices, cafeterias, maintenance jobs in asbestos contaminated buildings and experiments with dusty clothes, and 3) re-examination of the lung burden results to conjecture how the excess lung burden of household contacts could be accounted for by different indoor and also by different outdoor exposure levels. Finally, I hypothesize how much more asbestos exposure was added by household exposure to neighbourhood exposures and how it should affect the cumulative exposure assessment.

Figure B-14 Woman Carding Asbestos Wool, Circa 1930[1]

[1] Copied from: [Fortier, 1983a] with the author's permission.

B.3.2. Measurements in Workers' Households

In the scientific literature, there are hardly any comparisons of asbestos levels in homes of asbestos workers with those of neighbouring homes or with outside ambient air. A few measurements were conducted in the fall of 1979 by Gibbs and Rowlands in Quebec's mining towns [Gibbs and Rowlands, 1980]. They found higher indoor values (0.003-0.2 PCOM f/mL) in 5 homes of Thetford Mines than in 14 outdoor air samples (0.004-0.035 PCOM f/mL) for an approximate difference of 0.16 PCOM f/mL or a ratio of geometric averages of 4:1. However, the study report did not specify if the homes were those of asbestos workers nor if they were at the same distance of asbestos emissions than were the outdoor air samplers. Most fibres visible by optical microscopy were chrysotile.

In 1973 and 1976, Nicholson [Nicholson et al., 1980] collected 13 samples (4 hours each) in the homes of workers employed at chrysotile mining and milling operations in California and in Baie-Verte, Newfoundland, and 3 samples in the homes of non-miners in Baie-Verte. The investigator reported that workers at the time had no "access to shower facilities nor did they commonly change clothes before going home". The paper does not state the distance of the sampled homes and outdoor samples from the towns' asbestos emission sources nor does it state the seasons or climatic conditions during sampling; the samples were not taken simultaneously. Concentrations of chrysotile in workers' homes ranged from 50 to 5000 ng/m^3 (arithmetic mean $\approx$ 500 ng/m^3, geometric mean $\approx$ 200 ng/m^3), while the measurements in non-miners' homes were 32, 45 and 65 ng/m^3. The measurements in asbestos miners' homes would correspond roughly to a maximum of 0.17 f/mL, an arithmetic average of 0.017 f/mL and a geometric average $\approx$ 0.007 f/mL, whereas the three outdoor air samples had a geometric mean $\approx$ 0.0015 f/mL. The approximate difference between household and neighbourhood arithmetic average exposures was approximately 0.02 PCOM f/mL.

B.3.3. Measurements in Other Indoor Environments

a) "Non-Dusty" Areas Frequented by Asbestos Workers

Some past measurements have been taken in enclosed spaces where the coming and going of people with dust laden clothes was the main source of asbestos aerosols and/or where the presence of asbestos dust was not visibly different from that in the home: asbestos companies' offices and cafeterias[1].

In 1970, Gibbs and Lachance [Gibbs and Lachance, 1972] measured 1.9 mg/m^3 or about 1.9 PCOM f/mL [2] in an asbestos company's personnel office downwind from an asbestos mill; in the office hallway 0.7 mg/m^3 or 0.7 f/mL was measured; the difference of 1.2 f/mL might reflect the indoor contribution. Some hygiene data on offices, cafeterias and other "non-production" places of asbestos mines were supplied to us for a few plants for the early 1970s. 64 fixed station measurements taken in 1970-1972 averaged 1.6 f/mL in canteens and offices, and 6 measurements in a cafeteria averaged 3.7 f/mL.

b) Asbestos-Containing Buildings

After reviewing evidence based on 1377 air samples from 198 non-litigation U.S. buildings with asbestos-containing materials, the HEI-AR estimated that these buildings contained on average twice as much TEM asbestos fibres as did the outdoor urban air [HEI-AR et al., 1991].

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[1] Some detailed measurement data are available in **Appendix B20**.

[2] Since these measurements were taken in the asbestos-mining towns, I applied the conversion factor estimated from the concurrent gravimetric total dust and fibre count measurements by Environnement Québec and the Quebec Asbestos Mining Association in Section B.1.6.c.

In buildings without asbestos-containing materials, the average ratio was less than 1.0, suggesting a building-filter effect. Re-aerosolization seemed significant since many studies [Gazzi and Crockford, 1987; Guillemin et al., 1987; Guénel and INSERM U 88 / EDF-GDF, 1989] reported higher airborne concentrations during normal active occupation compared to quiescent conditions in buildings. Although these observations do not give an estimation of household-contact exposures, they provide evidence of some factors such as a building-filter effect (suggested by M. Corn, 1994, personal communication) and re-aerosolization during activity, which will be considered in the final estimation/simulation of household exposures.

c) *Re-Aerosolization of Asbestos by Brooming, Brushing, Dusting or Other Maintenance Work*

High average and peak concentration levels have been measured inside the homes of residents of villages in a few high mesothelioma incidence areas of the world. The people of these villages used materials containing various mineral fibres (tremolite, chrysotile, or non-asbestiform zeolite) to build or whiten the walls of their houses. Fibre levels measured after brooming fibre-laden floors or brushing fibrous building materials have attained 1.38 f/mL in Turkey [Rohl et al., 1982; Baris et al., 1987], 17.9 f/mL in Greece [Constantopoulos et al., 1985; Langer et al., 1987], and up to 78 f/mL in New-Caledonia [Luce et al., 1994; Goldberg et al., 1995].

Studies[1] on asbestos exposures of custodial workers (janitors, concierges, housecleaners, etc.) during their cleaning duties have shown that extremely high airborne asbestos exposure levels could be attained during activities which must have constituted a large part of the housework of asbestos workers' wives in the past. Lumley et al. [Carter, 1970] measured an average of 12 f/mL after brushing a friable sprayed crocidolite surface. Sawyer [Case et al., 1993] reported average measurements of airborne fibre concentrations after certain tasks in a library

[1] This section is totally based on the HEI-AR report [1991, pp. 4-73 to 4-79].

contaminated with fallout debris from friable sprayed chrysotile; average levels were 4.0 PCOM f/mL after dusting, 1.6 f/mL for dry sweeping and 15 f/mL after cleaning books. In a building contaminated with chrysotile, Burdett [Burdett et al., 1989] measured up to 100 times more airborne fibres with aggressive vs. passive sampling.

Regarding maintenance activities in asbestos mills in Quebec, Gibbs and Lachance [Gibbs and Lachance, 1972] reported that, in 1968, using compressed air hoses for cleaning operations exposed maintenance workers to 140 mg/m^3, multiplying ambient levels by an order of magnitude. Maintenance workers were among the most exposed asbestos workers with average exposures of 20-31 mg/m^3 by mill.

From a table of exposure estimates compiled by the CONSAD group [CONSAD Research Corporation, 1990; HEI-AR, 1991, p.4-75], it was estimated that maintenance workers without respirators experienced a t.w.a. exposure of 0.15 f/mL over a year by performing merely 87 hours of activities per year (@ 2.74 f/mL average during 87 hours) in general commercial and residential buildings. Sawyer [Case et al., 1993] reported 1.1-7.7 PCOM f/mL levels for installation activities in the library mentioned above. Hamilton [Hamilton, 1980] reported 1-5 f/mL for various activities in a ceiling space containing sprayed asbestos.

d) Experimental Evidence on Re-Aerosolization of Deposited Dust

Experiments on the effect of housework activities on the re-aerosolization of deposited asbestos dust should tell about past exposures due to housework in asbestos workers' homes. In 1982, Hunt [Hunt et al., 1982] recounted to the Ontario Royal Commission on Asbestos [ORCA, 1984] an experiment on the effects of sweeping floors and patting clothes:

> "an experiment wherein membrane samples were fastened to the handles of sweeping brooms, and sweeping operations were carried out using mixtures of crocidolite and chrysotile dust. The dust counts obtained showed that crocidolite became airborne in respirable dimensions very readily, while chrysotile, although airborne, was present mainly in accreted clusters which were not respirable. When

cotton overalls were deliberately contaminated with the two fibres and then subjected to moderately violent patting with the hands, the crocidolite fibres became airborne in respirable fractions as opposed to the chrysotile which flocculated and presented itself as non-respirable fibre aggregates. A room 9 feet square was used to demonstrate the re-aerosolizability of crocidolite. Contaminated clothing was shaken in the room and subsequently a broom was used to move the dust from one part to another. Counts of 6 f/cc down to 2 f/cc were obtained over a period of two weeks." ORCA, 1984, p.288, note 306]

Thus household maintenance activities produced high dust levels by re-aerosolizing deposited asbestos dusts for a substantial time period. It was also reported that peak short-term exposure levels in experiments on brushing asbestos laden clothes attained 200 PCOM f/mL [JMG Davis, note 305, in ORCA, 1984, p.288]. It is not clear if Hunt and Davis referred to different experiments or not. I estimated that, for a peak of 200 f/mL to leave 2 f/mL two weeks later, a settling rate of 1.2%/hour or 15%/day would be required.

Resuspension factors[1] for asbestos have been estimated at 0.0001-0.01 m^{-1} for either handling materials and clothes covered with asbestos, high-efficiency particulate air vacuum cleaning of heavily contaminated carpets or activity in public and commercial buildings.[Carter, 1970; Kominsky et al., 1990] These data are of no direct use because their interpretation is complex and because there are no data on settled dusts on clothes or in houses of asbestos workers [HEI-AR et al., 1991].

[1] A resuspension factor is the ratio of of settled (area) concentration to the disturbed airborne (volume) concentration. It has units of inverse length (ex: cm^{-1}).

B.3.4. Scenarios Consistent With the Lung Burden Data and With Evidence About Exposures in Workers' Homes and in Other Indoor Environments

In Section B.1.5, a t.w.a. exposure level for all years lived in the mining area was estimated by applying an exposure-burden relation estimated from 72 workers to the lung burdens of 10 household contacts. The t.w.a. lifetime exposure estimate was 0.76 f/mL for the 10 residents with household-contact exposure, 0.52 f/mL higher than that of 22 residents without household exposure. This difference had been obfuscated by ratio analyses at the panel meeting. In any case, the evidence relied on very small imprecise datasets and on uncertain analyses. Moreover, neither the contribution of household exposure to the whole population's average cumulative exposure nor the additional pollution due to the presence of an asbestos worker in a household can be inferred directly from this estimate. Indeed, this composite figure comprises all the time lived in the mining area, not just the time spent in the household of an asbestos worker; it comprises also an extra outdoor exposure due to the fact that household-exposed residents lived closer to the mines, mills and tailings piles. So, for comparison with other data on indoor household exposure, the composite household-contact exposure estimate had to be dissected into its components. Using three sets of assumptions or scenarios, the resulting separate elements could be compared with the above external data sources.

First, I attempted to correct the contrast between the household-contact and neighbourhood-only exposed groups for the confounding effect of residential distance from asbestos emission sources. The two groups lived respectively 1.6 km and 4.2 km from the mines in 1980. To estimate the "confounding" effect of distance from asbestos emissions, the non-occupational lung burden data were analyzed by a rudimentary linear regression with the logarithm of burden as a function of household exposure status and years lived in the mining area. The lung

burden of household contacts was estimated to be 6.2 (p = .013) times higher than that of neighbourhood-only exposed residents. When the logarithm of distance (*km*) was added to the log-linear model, the following model was obtained:

$$Tremolite\ burden = 0.19 \bullet km^{-0.57} \bullet (4.2)^{domestic} \bullet (1.017)^{duration}$$

$$R^2 = .28, \quad n=32$$

where *km* is the estimated distance in kilometers between a person's main residence over the last 30 years and the asbestos mines

domestic is a dichotomous "0"- "1" variable indicating if the person ever lived with a father, mother or spouse having worked at least one year in an asbestos mine or mill

duration is the number of years lived less than 10 km from the mines and mills

The distance-adjusted lung burden ratio of 4.2 (95% CI: 1.0-17.2) was 32% smaller than the unadjusted estimate of 6.2. The effect of distance on lung burden ratio when comparing two groups or persons was estimated as: $\left(km_2 / km_1\right)^{-0.57}$.

Note: The estimated relation between lung burden and distance from mines was consistent with the relation between distance and gravimetric asbestos levels measured by Lanting and den Boeft [Lanting et al., 1979] downwind from European asbestos plants:

$$Asbestos\ level\ in\ mg / m^3 = 0.15 \bullet km^{-1.86}$$

The much smaller absolute value of the exponent of kilometric distance estimated from the lung burden data was expected because: 1) lung burden reflects cumulative year-round exposure whether downwind or not from the sources, 2) non-differential measurement error of lung burden would be larger, 3) distance was measured very crudely in the lung burden study, and 4) numeric fibre lung burden measurements decay much less rapidly than mass concentrations as a function of distance from the source since heavier particles are the first to fall as a dust plume moves away from a source, leaving the larger number of small particles little affected.

Three scenarios were contrived to assess various exposure levels and conditions compatible with the lung burden data, and with the above direct and indirect evidence about indoor or housework exposure levels. The tables were built under the constraints of the past t.w.a exposure estimates obtained from the lung burden analyses, the 2.6-fold distance differential between household-contact and neighbourhood-only groups in the data, the assumption based

on our survey that household contacts had been so exposed half of the years lived in the mining area, and the arbitrary assumption that females spent on average - including weekends and holidays - 5 hours per day outdoor. On the basis of our population survey in 1989, it was approximated that household-exposed females had lived with an asbestos worker 50% of their lifetime and had spent 90% of their lifetime in the mining towns, while other female residents had spent 50% of their lifetime in these towns.

Scenario #1 was a simple deduction from the available lung burden data and the preceding assumptions; the distance effect was based on the above regression and the characteristics of our sample as $(1.6km/4.2km)^{-.57} = 1.73$.

Scenario #2 assumed that the distance effect was underestimated by regression due to distance misclassification error and was corrected as $(1.6km/4.2km)^{-1.0} = 2.6$; it also incorporated the opinion of a member of the Exposure Panel (M.C.) that there should be a building-filter effect[1].

Scenario #3 made a more realistic distance adjustment presuming that the distance ratio of 2.6 in the dataset was unrepresentative and assuming instead a distance effect of $(1.0km/1.6km)^{-1.0} = 1.6$, presuming accordingly that the lung burden should have been smaller; more importantly, it assumed that indoor levels were higher than outdoor levels by 25% due to the effect of housework in any home, a supposition particularly fitting for a study population composed mostly of housewives.

The scenarios are presented in detail in Appendix B21. The results are summarized at the bottom of Tables B-18 and B-19.

[1] A 25% filtration effect (=> indoor level = 75% of outdoor level) was assumed to occur during the six coldest months of the year, for a yearly average filtration effect of 12.5%.

B.3.5. Discussion

Although each piece of evidence was weak and insufficient in itself, the various types of data reviewed were consistent and corroborated the results of the lung burden analyses. All the data pointed to a significant contribution of household exposure to overall asbestos exposure levels. Table B-18 summarizes the direct and indirect empirical evidence on household asbestos exposures. The second column indicates the data source(s) and asbestos exposure circumstances for which exposure estimates were collected. The third through sixth columns show the quantitative exposure estimates according to whether the exposure estimate pertained more to the difference (3rd col.) or ratio (4th col.) between asbestos levels inside and outside the households of asbestos workers, or to the difference (5th col.) or ratio (6th col.) between asbestos levels inside households of asbestos workers and the levels inside neighbouring households without asbestos workers. Despite different data and methods, the three direct ratio estimates (items #1, #2 and #10) converged remarkably around 4.0. Excluding experimental and short-term exposure data, the range of excess indoor relative to outdoor or neighbourhood levels was 0.15-1.2 f/mL.

Table B-19 summarizes three scenarios based on lung burden data analyses and integrating inferences and notions from other indoor exposure circumstances and data under internal logic constraints. All three scenarios produced estimates well within the above-mentioned range. Scenario #1 simply adjusted for the estimated correlation between distance from emission sources and household exposure status, while Scenarios #2 and #3 are more realistic and suggest that the indoor-outdoor difference should be between 0.34 and 0.69 f/mL, for a best estimate around 0.5 f/mL. What would be a relatively inclusive range of plausible values around this estimate? In face of the weak evidence, the above interval was enlarged so as to have a bilateral 5-fold plausibility interval relative to the 0.5 f/mL estimate: 0.1-2.5 f/mL.

Table B-18 Summary of Distance-Adjusted Household-Contact Exposure Empirical Evidence

Item #	Estimation base, Exposure or sampling period	Indoor vs. outdoor asbestos workers' home exposure levels: exposure difference	Indoor vs. outdoor asbestos workers' home exposure levels: exposure ratio	Asbestos workers' vs. neighbouring homes' indoor exposure levels: exposure difference	Asbestos workers' vs. neighbouring homes' indoor exposure levels: exposure ratio
1	Thetford + Asbestos, 1979 (Gibbs and Rowlands)	< 0.16 f/mL	≈ 4	—	—
2	Other chrysotile mines, 1973-1976 (Nicholson)	—	—	0.02 f/mL	≈ 4
3	Asbestos mining companies' administrative offices, ≈1970	< 1.2 f/mL	—	—	—
4	Schools with ACM, 1980s	—	< 2	—	—
5	Other commercial and public buildings, 1980s	—	< 1	—	—
6	Custodial activities (dusting, brooming...) in a library with chrysotile fallout debris, 1976	1.6-15 f/mL (peaks or short-term)	—	—	—
7	Commercial and public buildings maintenance workers' annual exposure (87hrs/yr @ 2.7 f/mL), 1980s	> 0.15 f/mL	—	—	—
8	Dusting / brooming experiment (Hunt)	≈ 2-6 f/mL (crocidolite)	—	—	—
9	Short-term measurements after brooming or scrubbing in houses in 3 "mesothelioma villages" in Greece, Turkey and New-Caledonia	≈ 1.4-78 f/mL (tremolite, zeolite)			
10	Loglinear distance-adjusted household-contact vs. neighbourhood-only burden ratio	—	—	—	4.2
11	Biokinetic non-occupational exposure estimated from 72 asbestos workers' lung burdens	< 1.7 f/mL	—	—	—
12	Biokinetic projection from 72 workers to 10 household-contact lung burdens	0.52 f/mL	—	—	—

Table B-19 Summary of Three Distance-Adjusted Household-Contact Exposure Scenarios

Scen. #	Main characteristics of a scenario	Indoor vs. outdoor asbestos workers' homes exposure levels		Asbestos workers' vs. neighbouring homes' indoor exposure levels	
		exposure difference	exposure ratio	exposure difference	exposure ratio
Sc.#1:	1.73 distance-effect ratio	0.90 f/mL	3.3	0.90 f/mL	3.3
Sc.#2:	2.6 distance-effect ratio and 12.5% building-filter effect	0.34 f/mL	1.5	0.42 f/mL	1.7
Sc.#3:	lower lung burden, 1.6 distance-effect ratio and +25% housework-effect, no building-filter effect	0.69 f/mL	3.2	0.62 f/mL	2.6

Lacking distance data, the three ratios probably overestimated the true distance-adjusted exposure ratios. Scenario #2 attempted a correction by assuming a stronger distance effect; the result was a lower household-exposure ratio estimate of 1.7. This scenario also assumed that all households would be subject to a building-filter effect. On the other hand, re-aerosolization of deposited asbestos fibres due to housework and all indoor activities must have been substantial and would have affected all housewives in the region. Accordingly, Scenario #3 included a housework effect and resulted in a household-exposure ratio estimate of 2.6, demonstrating that a general housework and indoor activity factor was compatible with a significant excess exposure in the homes of asbestos workers[1]. However, the only reason why we used ratios was for comparability between the three datasets, the actual value of the ratios mattered less than the fact that three independent methods and datasets coincided, leaving little doubt that there was a contribution of asbestos workers to the asbestos levels in their households.

The relevance of the various indirect data and the likelihood of the subjective estimate of 0.5 f/mL based on Scenarios #2 and #3 must be understood as the evaluation of the excess exposures of the female members of asbestos workers' households. Before asbestos work clothes were confined to the workplace and other household-protection hygiene practices were generalized in the 1970s, most housewives and females in the mining towns broomed and dusted the house, furniture and clothes; they took care of the asbestos laden lawn, they insulated the windows and the stove with patted wet chrysotile, etc. These and other similar activities caused peak exposures and re-aerosolization of deposited asbestos.

Indeed, if nearly all females in the mining towns performed such activities, then most of the study population would have had higher exposures indoor than outdoor (excess of 0.07 f/mL

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[1] Members of the exposure assessment panel had disagreed strongly on this point, one member arguing against high household exposure estimates that measurements in buildings usually indicated lower levels than outdoor. A more precise conceptual framework distinguishing outdoor from indoor sources of pollution was now accounted for by the scenarios, showing that there is not necessarily a contradiction between a building-filter effect that could be observed in households with no asbestos worker and higher indoor levels in the households of asbestos workers.

and ratio of 1.25 under Scenario #3), but females in the households of asbestos workers would have had still higher exposures (excess of 0.69 f/mL and ratio of 3.2 under Scenario #3). It is not clear to what degree female residents' asbestos exposures due to housework should be accounted for in the cumulative exposure assessment; omitting this likely excess exposure might not lead to an underestimation of lung cancer risk when we eventually apply an exposure-effect model (e.g. EPA model) derived from occupational data, because occupational asbestos exposure measurements also underrepresented the true exposure levels of workers for similar reasons as for housewives and other residents.

What peak exposures could realistically generate an average indoor excess of 0.69 f/mL? If airborne fibres deposited or were cleared out of the house at the rate of 30%/hour, it could result from a single instantaneous peak indoor exposure to 6 f/mL per day or from two daily activities with 3 f/mL peak exposure each. If the combined sedimentation and clearance rate were slower, then the required peaks would be lower too. However, if the combined sedimentation and clearance rate were 50%/hour, then the required peaks would be nearly twice as high as those mentioned. Such peak chrysotile exposures would be easily conceivable in the asbestos mining towns and even more so in the houses of asbestos workers, given that brooming an asbestos laden floor in a library (item #6) generated short-term levels of 15 f/mL.

B.3.6. Conclusion

Although sparse and circumstantial, all available direct and indirect evidence consistently suggested that asbestos exposures of housewives and other members of asbestos workers' households must have been significantly higher than neighbourhood exposure levels in the asbestos mining towns. Moreover, all housewives and females performing housework in the asbestos mining towns probably incurred higher exposures levels indoor than outdoor. The most realistic scenarios which summarized the evidence under specific assumptions suggested that housewives in asbestos workers' homes could well have been exposed to indoor levels about 0.5 PCOM f/mL higher than in other neighbouring homes, with a 5-fold bilateral plausibility interval of 0.1-2.5 PCOM f/mL. In addition, a clearer description of the various exposure components and indirect data on exposures of custodial workers suggested that indoor exposures of neighbourhood-only exposed female residents may have been higher than outdoor levels by 25%; on this account, the panel's neighbourhood exposure estimates might have been underestimated by about 15%.

B.4. CUMULATIVE EXPOSURE OF THE EXPOSED POPULATION

B.4.1. Introduction

To estimate the exposure-effect gradient due to non-occupational asbestos exposure on the basis of the lung cancer mortality requires estimating the exposed population's mean cumulative asbestos exposure, which involves the estimation of the population's past exposure intensity and exposure duration. The former is addressed by the neighbourhood and household-contact outdoor and indoor exposure estimates made above. The latter is impossible to estimate directly since many of the women whose mortality experience was part of the mortality follow-up are no longer around, because they have died or moved. In effect we need to know two parameters for the women who lived in the area: i) the average number of years of residence in the area, and ii) the average number of years of residence in the house of an asbestos worker. Since there are no historical records available that could provide such data, we had to devise some method of generating it currently. The only available window on the life experience of the women in the study base over the mortality follow-up period comes from the remaining survivors of that cohort. The elderly women who now reside in the mining towns can provide some relevant data on the life experience of the cohort of interest.

Exposure time was ascertained through a representative population survey inquiring about past residential and household exposure experience of the study population. The above outdoor and household exposure intensity estimates were then applied to the respondents' residential and household exposure histories to compute cumulative exposure estimates for the exposed population.

B.4.2. Materials and Methods

a) Female Population Survey

The present description encompasses and adds to the material presented in Section B.1.4.b, since the same questionnaire and survey was used for different purposes. In the Spring of 1989, a survey was conducted on a sample of female residents of Mégantic and Richmond, the two provincial electoral ridings in which the two asbestos-mining agglomerations are located. The sampling frame consisted of women of at least 50 years of age residing in the two exposed agglomerations as well as in surrounding municipalities and townships in the two electoral ridings so as to cover other old mining towns and most of the in- and out-migration basin of the exposed agglomerations. A total of 1,096 women were sampled from the Provincial electoral lists updated in 1985. A stratified random sampling scheme[1] was used to increase precision in the most exposed groups: the mining towns and agglomerations, and the older population. Thus 42% of the sampled women were from the Thetford Mines agglomeration, 33% from the Asbestos agglomeration and 25% from surrounding towns and townships. 20% of the sampled women were 50-59 years of age, 40% were 60-69 years of age, and 40% were 70 years of age or older so as to approximate the distribution of exposure-years of the study base by birth cohort of the study base[2].

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[1] Sample sizes by decade and age group are indicated in Table B-20 and sampling fractions in **Appendix B22.**

[2] This differential weighing was justified both by the longer exposure experience and the longer follow-up period of the earlier generations. For instance, 79-year old respondents in 1989 represented the 1910 birth-year cohort which could have accumulated up to 79 years of exposure and whose cumulative exposure could weigh throughout the 40-year follow-up period since that cohort was over 30 years of age throughout the 1950-1989 follow-up period. By comparison, 50-year old respondents representing the 1939 birth cohort would not only have a more limited exposure experience but would just weigh over the 1969-1989 period (21 years) during which the cohort was at least 30 years of age.

All women less than 70 years of age in the sample were asked to fill and return a mail questionnaire and were invited again both by mail and by phone if they had not answered the first mailing. Subjects of at least 70 years of age were invited by mail and by phone to answer the questionnaire in a personal interview with a certified nurse. 817 women (75%) answered the survey. The filled questionnaires were all verified for completeness and chronological consistency and, in a few instances, were corrected after verification with the respondent.

A 13-page questionnaire was used which sought three kinds of data: 1) history of residence in the asbestos-mining areas and history of residence in the home of an asbestos worker; 2) information on health status, use of health services and life habits to assess the comparability of the exposed with the referent populations, 3) recall of past visible dust deposits.

Regarding past outdoor neighbourhood exposures, the respondents identified all their residences since birth, their addresses and their age at arrival to and departure from each of these addresses. Only the municipalities of residence were used in the following analyses to be compatible with the town-specific historical exposure intensity data. The age data were recoded as calendar years. Regarding past household exposures, respondents' household contacts having worked in the asbestos industry were recorded along with their employment periods and the time periods that they shared household with the respondent. Regarding occupational exposure, each respondent was asked to list all occupational exposures she might have had to asbestos and the corresponding time periods.

b) Representation of the Study Base by the 1989 Cross-Sectional Survey

In this ecological mortality study, cause of death and residential exposure classification were determined only from death certificates, at the same point in time. Implicitly, the study

subjects were eligible and "at risk" only while they lived in the exposed area and subjects moving out of the exposed area were lost to follow-up . The exposed study base consisted therefore of the person-years spent by the dynamic study population (females at least 30 years of age) in either of the two exposed agglomerations during the follow-up period. Accordingly, to compute the average cumulative asbestos exposure of the exposed study base, one would have interviewed in each year of the follow-up period each female at least 30 years of age living in the exposed agglomerations and would have asked them where and when they had lived since they were born. Then, the past town-year specific asbestos concentration estimates pertaining to each eligible resident's lifetime exposure history would have summed up to the year of the interview, obtaining thereby each woman's cumulative exposure for each year that she was a member of the study base. From there, one could have assessed the average cumulative asbestos exposure of the exposed study population either over the whole follow-up period, or for each year of the observation period and for each age group, depending on whether the exposure-effect relation was based on a global SMR for the full follow-up period or on age-period-specific rates or rate ratios. Alternatively, one might have obtained this sort of information from a representative sample of eligible females in each year of the follow-up period. But such information had not been collected and could not be reconstructed retroactively.

In the present ecological study, we surveyed a sample of the "surviving" population living in 1989 in the exposed agglomerations and the nearby areas where many ex-residents of the asbestos-mining towns had moved to. Each respondent in the survey represented all the women of the same birth cohort living in the same town as the respondent in a given follow-up year.

Example:

> A woman 70 years of age living in Thetford Mines in 1989 represented all the women born in 1919 in each year that she lived in the exposed towns between 1950 and 1989. Thus, if she lived in Black Lake in 1969 when she was 50 years of age, her cumulative exposure experience in 1969 represented the cumulative exposure experience of all women born in 1919 who lived in Black Lake in 1969. Likewise, if she lived in Thetford Mines in 1959 when she was 40 years of age, her cumulative exposure experience in 1959 represented the cumulative exposure experience of all women born in 1919 who lived in Thetford Mines in 1959. Etc.

This contrived construct assumed that the women in the exposed study base who moved away or died between 1950 and 1989 did not have systematically different asbestos exposure histories than those of the same birth cohort who were eligible for our survey in 1989. If this assumption were wrong, a self-selection bias could arise if, for instance, there were significant out-migration and if asbestos workers or women born in the mining towns were either substantially over-represented or under-represented among those migrants, or if out-migrants came from certain neighbourhoods and not from others. In theory, there could also be a survival bias if women most exposed had been at much greater risk of dying, leaving a surviving population which would underrepresent the highest past exposure experiences; however, this was improbable because asbestos related mortality would in the worse case account for a very small proportion of all deaths.

While not proven, the assumed independence of out-migration and asbestos exposure patterns was reasonable for this relatively stable and homogeneous female population born for the most part before 1940. Until 1979, the population was economically attached to a prospering[1] asbestos industry on which the jobs of their fathers, brothers and/or sons depended. After 1979, the asbestos industry's slump pushed many young families to leave the mining area for economic reasons, but asbestos workers over 50 years of age were either the last ones fired due to their seniority or were forced into early retirement. Thus, even in the slumping 1980s,

[1] The asbestos industry brought such a rapid and continuous growth to the mining area that locals called it the "white gold".

relatively few of the older members of the population would leave their region, families and friends. As well, neither the population nor even local physicians[1] suspected that non-occupationally exposed residents could be at risk of asbestos diseases; most health services were provided in local or regional hospitals; and most older asbestos workers' spouses and families stayed in their town, in their economic and social support network, unwary about non-occupational asbestos exposures, whether or not a family member suffered from an asbestos related disease [Dr. C. Fortier, 1989, personal communication]. Therefore, asbestos-related diseases or mortality would not have likely induced a material differential self-selection bias in our sample.

Since age-decade-specific mortality rates and ratios were to be used to compare the outcomes in the two exposed agglomerations to those in the referent population, the exposure histories were also computed separately for each of the two agglomerations, for each age group and for each decade of the follow-up period. This stratification made possible a more precise analysis of the exposure-effect relationship. Thus, for each follow-up year that she was in the study base, each respondent was classified with respect to a) the four follow-up decades Y_j between 1950 and 1989, b) four age groups[2] A_z and c) seven geographical areas[3] of residence X_k.

Table B-20 shows the number of females in the 1989 sample N_{jz} and the number of followed-up person-years NY_{jz} of these subjects in each of 16 age-calendar year strata for the two asbestos mining agglomerations. (A similar table representing the stratum-specific sampling fractions NY_{jz}/PY_{jz} in each of the two exposed agglomerations is presented in Appendix B22) Evidently, there were some missing strata among the older population in the earlier follow-up years; extrapolations were necessary to estimate the cumulative exposure for these empty sample cells of the study base. The sample size was small in some other strata, suggesting that

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[1] Personal communication by Dr. Clément Fortier, MD, ex-medical director of the General Hospital of the Asbestos Region and author of a history of Black Lake [Fortier, 1983b; Fortier, 1983a].

[2] "30-44", "45-54" and "55-69" years of age, and "70 years and over".

[3] A different set of historical exposure level estimates was attributed to each of the following seven areas: 1-3) each of the 3 main asbestos mining towns, 4) the other municipalities in the two asbestos agglomerations, 5) other asbestos mining towns, 6) other municipalities in the two asbestos counties, and 7) other non-asbestos areas of Quebec.

cell-specific estimates would be statistically unstable and that statistical smoothing or regression was required. The strata were not independent from one another since the same respondent contributed to many strata and contributed more than one person-year to any given stratum.

Example: A woman aged 40 in 1950 and living in the same agglomeration throughout the follow-up period contributed 5 person-years in 1950-1959 in the 30-44 age group and 5 in the 45-54 group; in 1960-1969, she contributed 5 person-years in the 45-54 age group and 5 in the 55-69 group; etc.

The stratified distribution in Table B-20 was termed the "follow-up sampling distribution" of the survey, and each stratum was termed a "follow-up stratum". This three-dimensional stratified structure with respect to calendar year, age and area of residence during a follow-up year was termed the "follow-up matrix". The sampling fractions of the study base were 5.9% and 3.7% respectively for the Asbestos and Thetford Mines agglomerations' study bases.

Table B-20 Distribution of the Survey Sample by Agglomeration, Follow-Up Decade and Age Group in a Given Follow-up Year

Agglomeration of Asbestos

Decade:	1950-1959	1960-1969	1970-1979	1980-1989	Total by age group
Age group	N[1] : NY[2]	N : NY	N : NY	N : NY	N : NY
30-44	102 : 697	109 : 768	49 : 370	11 : 31	237 : 1,866
45-54	29 : 181	88 : 579	154 : 1,115	45 : 397	246 : 2,272
55-69	13 : 64	46 : 292	112 : 748	118 : 1,046	234 : 2,150
70+	0 : 0	9 : 55	41 : 327	130 : 1,222	169 : 1,604
Total by decade	131 : 942	203 : 1,694	265 : 2,560	283 : 2,696	303 : 7,892

Agglomeration of Thetford Mines

30-44	149 : 985	155 : 1,094	73 : 524	10 : 24	340 : 2,627
45-54	37 : 199	123 : 774	142 : 998	53 : 450	271 : 2,421
55-69	9 : 36	47 : 300	141 : 927	114 : 982	245 : 2,245
70+	0 : 0	7 : 48	60 : 466	158 : 1,472	212 : 1,986
Total by decade	162 : 1,220	268 : 2,216	320 : 2,915	329 : 2,928	368 : 9,279

Note: A 79 year-old person surveyed in 1989 in Black Lake but have lived in Thetford Mines until 1969 represented Black Lake residents ages "70+ years" from 1980 to 1989, those aged "60-69" from 1970 to 1979, and represented residents from Thetford Mines for the pre-1970 period: those aged "50-59" from 1960 to 1969, and those aged "40-49" from 1950 to 1959.

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[1] N: the number of persons in our 1989 sample who were in a given age group in a given follow-up period. The numbers are not exclusive but overlap because a person sampled in 1989 could have resided in the agglomeration in many decades at different ages. Thus, in a given calendar decade, the same person may be counted in one or two adjacent age strata. In a given age stratum, the same person may be counted in one or two consecutive decades.

[2] NY: the number of follow-up person-years sampled in a given age-calendar year stratum and represented by the N respondents surveyed in 1989 who lived in a given agglomeration in a given age-calendar year stratum. For example, 102 (N) respondents surveyed in 1989 lived in the agglomeration of Asbestos while they were aged between 30 and 44 years of age between 1950 and 1959; they were in that agglomeration and in age group for a total of 697 person-years (NY) during that decade. That is, each respondent in that stratum contributed 6.83 years on average during the 1950-1959 decade.

c) *Cumulative Neighbourhood Exposure*

The respondents' neighbourhood, household and occupational experiences were converted into three corresponding cumulative exposure "follow-up matrices", with a cumulative exposure estimate for each age-year-area "follow-up stratum".

For simplicity, and in the absence of strong counter-arguments, I assumed that indoor exposures of neighbourhood-only exposed female residents were equal to the outdoor levels estimated by the panel. To compute the *neighbourhood cumulative exposure estimates* for each age-year-area follow-up stratum, each sample person-year in each stratum was considered as a separate "respondent" to the past-exposure questionnaire.

> A respondent who was 35 years old in 1951 and 36 years old in 1952 and lived in Asbestos those two years was considered as two different Asbestos "pseudo-respondents" in the same age-decade stratum, "interviewed" separately for their residential histories.

For each such pseudo-respondent, her cumulative exposure (f-y/mL) was estimated by summing all the town-year specific outdoor concentration estimates (f/mL) corresponding to the years and towns where the pseudo-respondent lived in the past, discounting the most recent five years. A 5-year lag was thought to be the absolute minimum period before a lung cancer could be induced and diagnosed; such a lag is often used in occupational lung cancer studies. Each pseudo-respondent's cumulative exposure was estimated separately, and all the estimates within the same follow-up matrix stratum were averaged over the number of pseudo-respondents or sample person-years within each stratum.

If 40 respondents lived in Asbestos between 1950 and 1959 *and* spent on average 6 years in the "30-44" age group over that period, 240 pseudo-respondents' cumulative exposures were computed, and these 240 resulting values were summed and divided by 240 true respondent-years. This average cumulative exposure was assigned to the Asbestos study base in the "30-44" age group over the 1950-1959 follow-up decade.

More formally, the total N_{aik} respondents living in area k in age-stratum a over the follow-up decade i *represented* NY_{aik} sample person-years. Each of these person-years or pseudo-respondents in the *aik* follow-up stratum experienced a cumulative exposure $FY_{NY_{aik}}$ (f-y/mL). The average cumulative exposure for the respondents in each follow-up stratum was computed as:

$$\overline{FY_{aik}} = \frac{\sum_{!}^{NY_{aik}} \left(FY_{NY_{aik}} \right)}{NY_{aik}}$$

where $\overline{FY_{aik}}$ = average lifetime cumulative exposure (fibres/mL-years) in age stratum a, decade i and exposure area k

and NY_{aik} = number of follow-up years from the lifetime residential histories of the respondents surveyed in 1989; in age stratum a, decade i and exposure area k

This computation of $\overline{FY_{aik}}$ was repeated for each follow-up town-year-age stratum aik. These estimates of age-year-specific average cumulative asbestos exposures were quantified in f-y/mL). Some cells were missing[1] or were based on too few actual respondents N_{aik}. The missing values were estimated by weighted linear regression and the computed $\overline{FY_{aik}}$ were smoothed with the same regression model. The result was a matrix of smoothed and

[1] Since women less than 50 years of age were not interviewed in 1989, the younger age groups in the later calendar years of the follow-up period were hardly represented in the sample. Also, most women in the older age groups during first half of the follow-up period were not alive at the time of our cross-sectional 1989 survey and were therefore not represented in the sample.

extrapolated $\overline{\hat{FY}_{aik}}$. This smoothed matrix was the study base's cumulative exposure estimate.

A single cumulative exposure estimate for the study base in each agglomeration was obtained by computing a weighted average of the $\overline{\hat{FY}_{aik}}$ relevant to each agglomeration. The weights were simply the estimated person-years PY_{aik} in each follow-up stratum, estimated from the quinquennial age-stratified demographic Census data. The weighted average lifetime neighbourhood exposure of an agglomeration's study base over the follow-up period (1950-1989) was thus estimated for those areas k which were specific to a given agglomeration[1] as:

$$\overline{FY_{Agglo}} = \sum_a \sum_i \sum_k \left(PY_{aik} \cdot \overline{\hat{FY}_{aik}} \right) \Big/ \sum_a \sum_i \sum_k PY_{aik}$$

where $\overline{FY_{Agglo}}$ = average lifetime cumulative exposure (f-y/mL) in agglomeration *Agglo* for the whole follow-up period

$\overline{\hat{FY}_{aik}}$ = average lifetime cumulative exposure (f-y/mL) in age stratum a, decade i and exposure area k interpolated and smoothed (^) by weighted linear regression.

and PY_{aik} = number of person-years in age stratum a, decade i and exposure area k, as estimated by interpolation from quinquennial population Censuses

[1] For instance, the Thetford Mines agglomeration comprised 3 areas: the town of Thetford Mines, the town of Black Lake and the "rest of the Thetford agglomeration" (i.e. the municipalities of Rivière-Blanche and Thetford-Partie-Sud).

d) Cumulative Household Exposure

As concluded in Chapter B.3, household exposure was estimated to add on average 0.5 PCOM f/mL to neighbourhood exposure, with a plausibility interval of 0.1-2.5 f/mL. The additional contribution of household exposure to cumulative exposure can then be derived by adding 0.5 f/mL for each year that the average woman lived with an employed asbestos worker. A pseudo-respondent's average number of years of household exposure was thus estimated for each follow-up matrix stratum as:

$$\overline{hY_{aik}} = \frac{\sum_{1}^{NY_{aik}} \left(hY_{NY_{aik}} \right)}{NY_{aik}}$$

where hY represented the number of years of household exposure.

Assuming that household-contact exposures occurred 7 days a week and 19 hours per day, these average numbers of years of household exposure were multiplied by 19/24. The resulting exposure durations were multiplied by the estimated household-contact attributable exposure level of 0.5 f/mL with its 5-fold bilateral plausibility interval. The resulting stratum-specific f-y/mL values were extrapolated to the missing follow-up strata, and the available strata were replaced by the estimates of a linear regression where age was quantified as a continuous variable and decade as a categorical variable. Finally, a weighted average was estimated for each agglomeration's study base.

e) Cumulative Occupational Exposure

Occupational exposure was estimated in the same way as household exposure, except that in the absence of an occupational exposure estimate an arbitrary 5 f/mL occupational exposure level was assumed (plausibility interval of 3-15 f/mL). Moreover, since occupational exposures were intermittent, these exposure values were divided by the full-week/work-week factor of 4.2.

B.4.3. Results

a) Cumulative Neighbourhood Exposure Matrix

According to the survey, on average, the female residents of the two asbestos mining agglomerations lived 60% of their lifetime in one of the three main asbestos-mining towns. Allowing for a minimal 5-year latency period, the corresponding average cumulative exposure estimates over the 1950-1989 period were 9.3 f-y/mL in Asbestos and 19.5 f-y/mL in Thetford Mines; the corresponding estimated values over the 1970-1989 period were almost identical overall, the lower cumulative exposure in the younger age groups being offset by the higher cumulative exposure of older age groups. As shown in Table B-21, cumulative asbestos exposure increased with age, although less markedly than what would be expected intuitively. Similarly, there was an expected drop in the cumulative exposure of women less than 55 years of age in the 1970s and 1980s, although it was less apparent in Thetford Mines than in Asbestos. Cumulative exposure remained relatively constant over the 1950-1989 period despite the steep decrease in exposure intensities over that period. This paradox is explained and illustrated with an idealized example based on our exposure estimates in Appendix B23.

Table B-21 **Smoothed Cumulative Neighbourhood Exposure by Agglomeration, Follow-Up Decade and Age Group in a Given Follow-up Year**

Agglomeration of Asbestos

Age group	1950-1959	1960-1969	1970-1979	1980-1989	Total by age group
	f-y/mL	f-y/mL	f-y/mL	f-y/mL	f-y/mL
30-44	8.6	9.0	7.1	6.3	7.6
45-54	10.0	11.1	9.8	7.6	9.4
55-69	9.7	12.2	11.2	10.0	10.8
70+	9.6	10.0	12.1	11.4	11.3
Total by decade	9.0	10.2	9.5	8.8	9.3

Agglomeration of Thetford Mines

30-44	16.0	18.1	17.6	15.7	16.7
45-54	19.5	22.4	19.3	18.1	19.5
55-69	22.1	23.3	23.8	19.8	21.9
70+	27.6	26.7	23.3	22.9	24.3
Total by decade	18.8	20.5	20.0	18.8	19.5

b) Cumulative Household Exposures

70% of the exposed study base of two asbestos agglomerations lived with an asbestos worker at some point in time. Some 20% of the exposed study base had lived with a father working as an asbestos miner or miller, and 50% had lived with a husband working as an asbestos miner or miller. Although roughly 25% of the study base had lived with more than one asbestos worker in their lifetime, less than 10% lived in a household where two or more members worked as asbestos millers or millers concurrently, and that exposure to more than one asbestos worker concurrently lasted usually no more than 5 years, i.e. less than 10% of the average lifetime of the study base. Some 40% of the respondents from the two agglomerations recalled having washed or dusted asbestos laden clothes at some point in the past.

Among the respondents who ever lived in the household of an employed asbestos worker, average household exposure was about 26 years in the Thetford Mines agglomeration and 30 years in the Asbestos agglomeration. However, when averaged over the whole study base, cumulative household exposure of the study base was 18 years in the Asbestos agglomeration and 21 years in the Thetford Mines agglomeration. These exposure years were multiplied by 19/24 since it was assumed that this exposure occurred on average 19 hours per day in the female population, resulting in 14.3 and 16.6 adjusted exposure years respectively. The corresponding cumulative asbestos exposure estimates were thus 7.1 f-y/mL with a 1.4-35.7 f-y/mL plausibility interval in Asbestos and 8.3 f-y/mL with a 1.7-41.7 f-y/mL plausibility interval in Thetford Mines.

c) Cumulative Occupational Exposure

Past occupational exposures among the women of the study population were infrequent. According to our survey, 2% of the Thetford Mines agglomeration study base and 4% of the Asbestos agglomeration study base have worked in the asbestos industry in blue-collar jobs in

the past; according to McDonald et al. [McDonald et al., 1993], these were not considered "dusty jobs" in the past, and 81% of the females on the companies' payrolls had less than 30 mpcf-y or 105 f-y/mL cumulative exposure.

According to the 1941 and 1951 Censuses, 33 females in Thetford Mines worked in 1941 for the asbestos companies and 15 had direct contact with asbestos as cobbers, while 40 worked for the industry in 1951 of which 7 worked directly with asbestos (one as a miner) [Cinq-Mars et al., 1994, p.278-9]. The employment of females for handling asbestos consisted mostly in cobbing (separating long fibrous veins from pieces of ore with a hammer); females participated to this activity mostly between 1910 and 1930 and during WW-II [Cinq-Mars et al., 1994].

On average, females who had dusty occupational exposures such as cobbing performed these jobs for about 6 years between the ages of 15 and 24 years[1], and the exposures occurred mostly before the 1950s and no later than in 1963. Moreover, according to our survey, about 1% of each agglomeration's study base had performed some occasional asbestos-related work home such as mending jut bags used for shipping.

Overall, the average member of the whole exposed study base has experienced 0.34 year of occupational asbestos exposure; this figure was further divided by 4.2 to account for the fact that workers were on average exposed some 40 hours/week rather than 168 hours for continuous round-the-clock exposures. If past occupational exposures of cobbers, jute-bag menders and other females employed by the asbestos industry were 3-15 f/mL, occupational exposure would have added around 0.24-1.2 f-y/mL to the neighbourhood-only and household-contact cumulative exposures of the asbestos-mining agglomerations' study bases.

[1] This observation in our survey is corroborated by the 1921 Canadian Census as cited in the Thetford Mines Centenary Anniversary Book. The authors note that "cobbing" (*sic*, probably "cobbling"), also called "sheiding" (*sic*, probably "shedding"), was mostly a "pre-marital activity".

B.4.4. Discussion

Cumulative neighbourhood exposures could have been underestimated by some 15 f-y/mL for the 1950-59 follow-up period, 10 f-y/mL for the 1960-69 period and 5 f-y/mL for the 1970-79 period. Indeed, as discussed above in Sections B.1.6.f and B.2.3, town-year asbestos exposure levels before 1945 were probably underestimated by 0.5 f/mL or more because the exposure panel's estimates for that period were based only on asbestos production trends and did not take into account that urban areas were smaller and the emission sources were lower further back in the past. Adjusting the cumulative exposure estimate accordingly for the 1950-1989 period would add a substantial 5 f-y/mL to each agglomerations cumulative exposure estimate. However, since the final follow-up period for the Mortality Study was reduced to 1970-1989[1], the addition to the cumulative exposure over those 20 years would be of about 2 f-y/mL. In the end, since the arguments to justify these putative corrections were not discussed by the expert panel, I stretched the plausibility interval from 3-fold to 5-fold on each side of the point estimate to reflect a greater uncertainty; this pushed the arithmetic average cumulative exposure value upward by nearly 20% while not altering the geometric average based on the experts' judgment.

Cumulative occupational exposures may have been underestimated by the survey because of the cohort nature of this type of exposure. Indeed, only respondents over 60 years of age in our 1989 survey have mentioned past occupational exposures; they had been so exposed in their 20s. These respondents were younger members of the study base in the 1950s. Yet, the older females in the 1950s and 1960s could not be sampled in our 1989 survey and probably had similar exposures in their youth since women had been mobilized in the WW-II effort and in

[1] The restriction of the follow-up period to 1970-1989 is explained in the general Introduction and in the chapter on Mortality.

the 1910-1930 period. Since these occupational exposures were not accounted for, the value of 1.2 f-y/mL was deemed to be rather a best estimate than an upper estimate, resulting in a final cumulative occupational exposure estimate of 1.2 f-y/mL with a plausibility interval of 0.24-6.0 f-y/mL (5-fold interval each side of the best estimate).

Finally, the overall cumulative asbestos exposure of the study base was computed as the sum of the estimated neighbourhood-only, household-contact and occupational mean cumulative exposures as in the following table:

The combined PY-weighted average for the two agglomerations was 25 f-y/mL with a bilateral 5-fold plausibility interval of 5-125 f-y/mL. This subjective estimation and its uncertainty were geometric in nature because nearly all appraisal and computational errors were more likely to be geometric or proportional than arithmetic. Thus 25 f-y/mL was practically the geometric mean of the set of more or less likely estimates of cumulative exposure. Yet, the corresponding arithmetic mean seems to be a more appropriate exposure metric when assuming or testing a linear exposure-effect relationship [Oldham, 1965; Seixas et al., 1988]. The arithmetic mean was estimated using the following formula[1]:

$$AM = GM \bullet e^{\ln(GSD)^2/2}$$

where AM is the arithmetic mean, GM is the geometric mean,

and GSD is the geometric standard deviation: $\sqrt[1.96]{5}$ or 2.273

The result was an arithmetic average of 35 f-y/mL.

[1] The formula was taken in Seixas' paper [1988]. I checked its validity against known examples and against the arithmetic average of 2000 simulated observations from a geometric distribution.

Table B-22 Estimated Cumulative Exposures of the Exposed Population

Asbestos agglomeration	Point estimate = geometric mean (f-y/mL)	Point estimate = arithmetic mean (f-y/mL)	Plausibility interval (f-y/mL)
Neighbourhood Exposure	9.3	13.0	1.9 - 46.5
Household Exposure	7.1	9.9	1.4 - 35.5
Occupational Exposure	1.2	1.7	0.2 - 6.0
Total Exposure	17.6	24.7	3.5 - 88.0
Thetford Mines agglomeration			
Neighbourhood Exposure	19.5	27.3	3.9 - 97.5
Household Exposure	8.3	11.6	1.7 - 41.5
Occupational Exposure	1.2	1.7	0.2 - 6.0
Total Exposure	29.0	40.6	5.8 - 145.0
Both agglomerations combined			
Neighbourhood Exposure	16.0	22.4	3.2 - 80.0
Household Exposure	7.8	10.9	1.6 - 39.0
Occupational Exposure	1.2	1.7	0.2 - 6.0
Total Exposure	25.0	35.0	5.0 - 125.0

The purpose of the exposure assessment was to apply exposure-effect gradients estimated from cohorts of asbestos workers to the exposed population's cumulative exposure. However, the cumulative exposure metric (exposure intensity X years of exposure) does not actually represent the same cumulative exposure in residents of asbestos-mining towns practically exposed continuously 168 hours/week than in workers exposed roughly 40 hours/week[1]. Therefore, for comparability with the metric used for asbestos workers, the cumulative exposure estimate in this study was scaled up by a factor of 4.2 (=168/40) to estimate the "occupationally equivalent" cumulative exposure of this continuously exposed population. The resulting estimate had a geometric average of 105 f-y/mL with a plausibility interval of 21-525 f-y/mL and an arithmetic average of 147 f-y/mL.

The estimated average cumulative exposure in this population was comparable to those of seven exposed occupational cohorts on which asbestos risk assessments have been based[2]. Yet, the average exposure intensity (0.5-1.0 f/mL) in this study's exposed population was almost two orders of magnitude lower than average intensities in the occupational cohorts. Two main factors account for this apparent paradox: 1) continuous non-occupational exposures represented about 4 times more exposure time than intermittent occupational exposures according to the linear cumulative exposure metric; 2) neighbourhood exposures lasted on average 4-10 times more years than occupational asbestos exposures in asbestos cohorts.

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[1] The work week was longer than 40 hours/week before the 1950s, but sick leaves, holidays, vacations and temporary layoffs would lower the average number of hours worked for any given time period. The EPA and HEI-AR also assumed that members of historical occupational cohorts worked on average 40 hours/week.

[2] Average cumulative exposures in the seven less exposed cohorts were roughly: 100 f-y/mL [Berry and Newhouse, 1983], 45 f-y/mL [Dement et al., 1983b], 110 f-y/mL [Finkelstein, 1983], 35 f-y/mL [McDonald et al., 1983a], 50 f-y/mL [McDonald et al., 1983b], 65 f-y/mL [Seidman et al., 1986], 35 f-y/mL [McDonald et al., 1984].

B.4.5. Conclusion

Applying the study population's residential and household exposure histories to the panel's concentration estimates resulted in a geometric average cumulative asbestos exposure of the exposed study base of 25 f-y/mL with a plausibility interval of 5-125 f-y/mL, implying an arithmetic average of 35 f-y/mL. According to the conventional linear cumulative-exposure metric, these cumulative values of continuous environmental exposures must be scaled up by a factor of 4.2 for comparability with cumulative exposures of asbestos workers.

B.5. ASBESTOS EXPOSURE OF THE REFERENT POPULATION

The exposure assessment concerned only past exposures of the female residents of Quebec's asbestos-mining agglomerations. It was presumed that their exposures were orders of magnitude above those of the referent population and that the latter were negligible, because the linear exposure-effect model supposes that the contribution of asbestos to the risk of lung cancer in the referent population be negligible so that referent lung cancer rates can be used as the baseline rates of the exposed population. This assumption appeared to be realistic.

In the referent agglomerations, asbestos levels must have increased continuously from 1876 until the end of the 1970s due to the monotonically increasing use of asbestos products over that period. Therefore, recent measurements of ambient asbestos in the comparison regions should provide an estimated upper limit of historical levels. Exhaustive year-long measurements taken in 1984 in downtown Montreal and in the rural village of St.-Étienne by Sébastien et al. [Sébastien et al., 1986] were pooled and averaged to estimate background exposure (0.0004 PCOM f/mL) in the referent agglomerations. As a result, the background average cumulative exposure of the referent population had to be lower than 0.03 f-y/mL.

As well, the effect of migration from the exposed to the referent areas could not have altered significantly the background exposure of the referent population. The exposed population represented 1/35 of the referent population. It had little economic incentive to migrate to the referent agglomerations, preferring moving to Montreal, Quebec or Sherbrooke, agglomerations which were excluded from the study. Even if an implausible 2% of the exposed population had migrated each year to the referent agglomerations and had a remaining life expectancy of 40 years after migration, the contamination of the referent population would have been less than 1.6%, assuming normal population growth rates, thereby adding at the most 0.35 f-y/mL (1.6% of 22.4 f-y/mL estimated for the exposed population) to the average cumulative exposure of the referent population.

As to occupational and household exposures, the referent agglomerations comprised very few asbestos-related industries. Still, supposing that as many as 1,000 referent women had been

exposed to asbestos occupationally or in the household, they could hardly have added more than 0.04 f-y/mL to the average cumulative exposure of the referent population.

Altogether, the three mentioned sources of asbestos exposure in the referent population would add to less than 0.4 f-y/mL average cumulative asbestos exposure. This was a personal assessment of contrived extreme scenarios. This exaggerated estimate of the average cumulative exposure of the referent population represented 1% of the estimate for the exposed population. Accordingly, although the excess relative risk of lung cancer due to asbestos should be underestimated in the present study by assuming that the referent population's average cumulative asbestos exposure is null or negligible, it should not be underestimated by more than 1% .

PART C. MORTALITY STUDY

C.1. Introduction

The present chapter presents relative risks of lung cancer mortality among females in the Quebec mining areas. In the process, I describe the socio-demographic profiles of the exposed and referent populations and the procedures that were undertaken to control or to counteract suspected biases. As explained in the Thesis Outline, while the original intention was to present results for the entire period from 1950 to 1989, it turned out that the data for the early years were of dubious validity. Consequently, the period of mortality observation was 1970 to 1989.

C.2. Materials and Methods

C.2.1. Study Design

To briefly summarize the design outlined in Section A.4, this study was an ecological study comparing the exposed agglomerations with unexposed agglomerations. The two exposed agglomerations were those comprising the main asbestos mines, namely Asbestos and Thetford Mines. The unexposed agglomerations comprised 60 agglomerations in the Province of Quebec, that excluded the large metropolitan areas as well as the most rural areas. The outcome of interest was lung cancer risk. Mortality was used as a surrogate for incidence because: a) mortality data were available for a longer period of time with a more constant quality of data; b) lung cancer mortality reflects incidence due to the short survival of lung cancer cases, and c) lung cancer mortality was the outcome used in the asbestos occupational studies on which environmental asbestos risk assessments have been based.

The computation of lung cancer relative risks required the definition of numerators of lung cancer deaths in the exposed and unexposed areas, and the definition of denominators of person-years in the exposed and unexposed areas. The numerators were based on data from the mortality database of the Province of Quebec, and the denominators were based on data from successive quinquennial Canadian censuses.

C.2.2. Numerators for the Computation of Mortality Rates

The cases were identified from individual computerized death certificates[1] of the Quebec Mortality Registry for the years 1970 to 1989. The underlying cause of death coded in the Mortality Registry was used to classify the health outcome. Since the follow-up period covered ICD-6 through ICD-9, a modified version (M. Goldberg, 1991) of the ICD-LCDC[2] correspondence table developed by Health Canada was used to convert different versions of the ICD codes to today's LCDC codes (ICD and LCD codes used are listed by cause of death in Appendix C1).

5% of all death certificates of women at least 30 years of age were not included in the study because either the municipality of residence, the cause of death or the age could not be coded properly. The 5% were spread relatively uniformly over time. There was no reason to suspect that the proportion excluded would differ between the exposed and unexposed populations; it was therefore unlikely to produce any bias in our comparisons.

The municipality of residence recorded on the death certificate was used for exposure classification. Municipal codes were processed by the Bureau de la statistique du Québec to account for municipal fusions and code changes over the whole study period. To standardize to a single municipal code throughout the observation period, all year-specific codes from 1970 to 1989 were converted to the corresponding municipal code or "geocode" in year 1987. For those records for which no valid geocode had been assigned on the original data entry, an automatic record-linkage algorithm was used to match municipality names or parts of these names to the 1987 correspondence table of municipal names and codes. A few death certificates had to be coded manually.

1983b], 65 f-y/mL [Seidman et al., 1986], 35 f-y/mL [McDonald et al., 1984].

[1] The records were made available to us after removing all identifiers such as persons' names, social insurance number and specific addresses.

[2] LCDC: Laboratory Centre for Disease Control, a branch of the ministry of Health Canada.

C.2.3. Denominators for the Computation of Mortality Rates

Four Canadian quinquennial censuses were used to compute populations and person-years[1] for each municipality of the 60 referent and 2 exposed agglomerations. The geocodes were readily available from the computerized census datasets. I compiled a correspondence table from the mortality datasets and applied it to the census data to convert the census-specific geocodes to 1987 geocodes. Checks and corrections (where necessary) were applied to the correspondence between Federal census data and Provincial mortality data; indeed, changes of geocodes sometimes lagged by up to one year between federal and provincial datasets.

After ensuring the consistency of the municipal geocodes, the denominator census data and the numerator mortality data were matched on year 1987's geocodes and both numerator and denominator data were aggregated by OPDQ agglomeration[2].

C.2.4. Standardized Mortality Measures of Effect

Since conventional risk assessments postulate a linear relation between lung cancer relative risk and cumulative asbestos exposure, the excess lung cancer mortality of the exposed population was analyzed using ratio measures of effect: age-calendar-year-standardized proportionate mortality ratios (SPMRs), indirectly standardized mortality ratios (SMRs) and directly standardized mortality rate ratios (SRRs). 95% confidence intervals were computed using conventional estimation formulae. Referring to Vol. 2 of Breslow and Day's "Statistical Methods in Cancer Research" [Breslow and Day, 1987], the 95%CI were calculated using formula 2.17 (p.77) for SPMRs, Byar's approximation for SMRs (formula 2.13, p.69), and

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[1] Denominator person-years were computed by interpolating yearly population estimates between quinquennial censuses, using a geometric or exponential population progression model.

[2] As mentioned in Section A.4 on Study Design, the Office de planification et de développement du Québec (OPDQ) defined an "agglomeration" as a grouping of municipalities with continuous or adjacent built spaces comprising at least 4,500 residents.

formula 2.6 (p.64) for SRRs. These ratio measures and confidence intervals were computed for 42 exhaustive and mutually exclusive causes of death (listed in Appendix C1) including lung cancer (ICD-9 162). Standardization was carried over four calendar-year strata and six age strata ("30-34", "35-44", "45-54", "55-64", "65-69", "70 years and over"), for a total of 24 calendar-year-age strata.

Both for validity and for statistical inference, the SPMR requires that the total number of deaths from all causes be much larger than the number of deaths from the cause of interest; this condition was fulfilled since there were 30 deaths from other causes for each death from lung cancer in the study base. In addition, SMR, SPMR and SRR statistics such as standard errors, p-values, and confidence intervals "assume that the standard population is very large relative to the exposed population, so that sampling errors in the standard rates may be ignored" [Breslow et al., 1987], otherwise statistical error will be underestimated by the conventional statistical formulae applied here. This condition was fulfilled since the number of person-years in the whole study base was 40 times larger than that in the exposed study base.

A chi-square test of homogeneity [Breslow and Day, 1987] based on the Poisson distribution was applied to the calendar-year-age strata for each cause of death to see if standardized summary measures of effect were appropriate and if mortality did not concentrate in a specific decade or in a specific age group.

C.3. Socio-Demographic Comparisons of the Exposed and Unexposed Populations

The 212 municipalities in the 62 non-metropolitan agglomerations of the study were small to mid-sized. This and the restriction of the study to women at least 30 years of age ensured that the study population was relatively homogeneous, being composed for the most part of Catholic French Canadian females with relatively similar lifestyle, work, education and socio-economic profiles. Still, potential confounding could not be excluded outright and socio-demographic comparability was assessed more specifically. In the present section, the exposed and referent areas are compared socio-demographically for two separate time periods. For the mid-1970s, many socio-demographic variables were available for each of the 67 agglomerations but concerned the whole population of the agglomerations (i.e. males and females, all ages); these data were examined both in univariate (section C.3.1) and multivariate analyses (section C.3.2). For the late 1980s, both whole population data (section C.3.3) and data specific to the older female population (section C.3.4: our local survey and the Quebec Health Survey) were available for the two exposed agglomerations but were not reliable by individual referent agglomeration. So for the late 1980s, the comparison was between the two exposed agglomerations and a referent area pooling the seven large "Régions socio-sanitaires" comprising the 60 referent agglomerations.

C.3.1. Univariate Comparison of the Exposed and Referent Agglomerations, 1974-1978 Data

Ecological data by agglomeration on various potential confounders were obtained for 1976-1981 from a document of the Ministry of Quebec Health and Welfare [Pampalon, 1985]. The latter aggregated data on 43 demographic, social, hygiene, and economic variables[1] by socio-economic region, census division, and agglomeration. These data were available for all 62 agglomerations in the study. We supplemented these data with statistics contained in a publication of the Canada Mortgage and Housing Corporation [Bond et al., 1979] on social and urban characteristics of Canadian cities of intermediate size and with data from the 1976 Census on Canada's municipalities of 5,000 population and over [Bulletin #92-810]. The main socio-demographic variables from the different sources are summarized in Table C-1, contrasting population-weighted averages for the 2 exposed agglomerations against those for the 60 referent agglomerations. However, a total of 58 socio-demographic variables have been considered and analyzed. Whenever a characteristic's average for both exposed agglomerations combined was significantly[2] different ($p \leq .05$) from the average of the 60 referent agglomerations, the corresponding value among the exposed is underlined in the table.

In the early 1970s the two exposed agglomerations seemed to have a somewhat higher socio-economic status than the referents. The two exposed agglomerations combined had a slightly lower multivariate "Dépatie poverty score[3]", a smaller proportion of tenants, a lower male unemployment rate, homes were less crowded and were better equipped relative to the 60 referent agglomerations; all these differences were small but statistically significant. Average

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[1] The data were compiled from various official sources (Canadian Census, Ministry of Social Services, Ministry of Transport, Ministry of Environment, Ministry of Commerce, Imperial Tobacco, etc.).

[2] Statistical significance was estimated by using the number of agglomerations for which a given variable was available as the sample size for that variable.

[3] The Dépatie poverty score is an indicator developed by a French sociologist (Dépatie), and was applied to Quebec in 1976 by P. Cliche [Cliche, 1976; Cliche, 1977]. The more positive the score, the poorer the population.

outcome per capita and per household were both slightly higher in the exposed agglomerations but not significantly so.

Regarding health habits, per capita sales of tobacco and alcohol were slightly lower in the large "health administration areas"[1] (the "Quebec" and "Eastern Townships" regions) comprising the two exposed agglomerations than in the 8 health administration areas comprising the 60 referent agglomerations. However these regional differences might not be transposable to the more specific agglomeration level.

Relative to the referent agglomerations, population density was significantly lower for the exposed agglomerations combined, but there was no such difference when restricting the comparison to each agglomeration's main municipality or urban centre. In fact, the exposed and referent agglomerations were comparable in terms of most socio-demographic characteristics: population size, ethnicity, proportion of francophones, family size, age of housing facilities, level of education, labour force, and number of hospital beds per capita. However, there were slightly fewer general practitioners per capita in the exposed agglomerations. The occupational structures of the employed population of the exposed and referent agglomerations were intrinsically very different in four industrial sectors; relative to the referents, the exposed agglomerations had proportionately more workers in the mining industry but proportionately less workers in the forestry, the pulp and paper and the textile industries.

There were some socio-economic differences between the two exposed agglomerations. Relative to Asbestos, the Thetford Mines agglomeration was somewhat more urbanized, more prosperous, more educated and comprised a larger proportion of French speaking inhabitants. Thetford Mines also had more hospital beds per capita. Moreover, Thetford Mines had a regional college and had proportionately more transformation and service industries than did Asbestos.

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[1] Regional administrative units termed by Quebec's Ministry of Health and Social Services: "régions socio-sanitaire".

Table C-1 Socio-Demographic Characteristics of Exposed and Referent Agglomerations Over the 1971-1976 Period

Characteristic	Agglomerations			
	Thetford Mines	**Asbestos**	**2 Exposed Agglomerations**	**60 Referent Agglomerations**
Female population per agglomeration (mean)	14,325	7,380	10,853	9,911
Population per km^2 (mean)	27	83	46[1]	217
Density of largest municipality	802	933	847	755
French Canadian Population (%)	95	88	93	93
Dépatie poverty score [2] (mean)	- 8	- 4	- 7	- 2
Average income per capita ($)	6,272	6,434	6,327	6,269
≥10 years schooling (%)	41	36	39	44
Active female workforce (%)	37	33	36	39
Female unemployment (%)	10	16	12	12
Male unemployment (%)	8	5	7	9
Tenants (%)	36	45	39	50
Single-parent families (%)	9	7	9	10
Homes > 1.1 person/room (%)	11	15	12	14
Life expectancy of females	78	76	77	77
Hospital beds/1,000 population	12	3	9	10
General practitioners/10,000 pop.	5	6	5	7
Bacteria in tap water[3]	104	35	81	90
Index[4] of fibre consumption	150	149	150	148
Index of sugar consumption	235	239	236	228
Index of fat consumption	364	327	351	365
Cigarettes sales per capita (pop.≥15 years of age)	88	83	86	99
Litres of pure alcohol sold per capita (pop.≥15 years of age)	10	9	9	11

[1] Statistically significant differences ($p \leq .05$) between exposed and referent agglomerations are underlined.

[2] The Dépatie poverty score is an indicator ≤+1 and ≥-1, developed by a French sociologist (Dépatie). Here, the score is multiplied by 10 for readability. The more positive the score, the poorer the population. It was applied to Quebec in 1976 by P. Cliche [Cliche, 1976; Cliche, 1977].

[3] These and the following data were not available by agglomeration, but only by the larger health administration area ("région socio-sanitaire"). The values are those of the agglomerations' health administration areas.

[4] The three indices indicated here were developed by the Ministry of Health to represent the conformity of the population's eating habits with the dietary recommendations of the Ministry of Health. The higher the index, the healthier the eating habits.

C.3.2. Multivariate Analyses of Agglomeration-Based 1974-1978 Data

Fifty-eight socio-demographic variables, including those listed in Table C-1, were analyzed with principal components analysis[1] to synthesize the numerous variables to a few factors and with cluster analysis to seek a reduced set of more homogeneous and more comparable referent agglomerations with respect to the exposed agglomerations. These analyses did not reveal any subset of referent agglomerations which would be socio-economically more comparable with the two exposed agglomerations. The exclusion of any subgroup of agglomerations could slightly improve comparability on certain covariates but only at the expense of comparability on other equally significant covariates. Hence it was not possible to improve comparability with the exposed agglomerations by restricting the referents to a smaller set of agglomerations.

In a second approach, precision-weighted linear regression[2] of the lung cancer SMR on the above variables was used to identify potential confounders and to adjust for their influence on the effect of the agglomeration's asbestos exposure status ("exposed" vs. "referent"). The dependent variable was the lung cancer SMR by agglomeration as computed by the Quebec Health Ministry for the 1974-1983 period[3]. The analysis was restricted to the 60 referent agglomerations to seek socio-demographic determinants of lung cancer SMR independently of asbestos exposure. The 58 socio-economic variables were tried in turn by groups of six in stepwise regression, and a final regression used all the variables selected in "six-at-a-time" steps. If the fitting had been successful, the fitted model would have been applied to the exposed agglomerations to estimate an "expected" lung cancer SMR in the exposed population due to non-asbestos risk factors. A corrected lung cancer SMR would then have been obtained

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1 SPSS statistical software for SUN/UNIX systems was used for these analyses.

2 Using inverse variance of the SMR as the weight of the loss function in the SYSTAT non-linear regression program [Dennis and Schnabel, 1983; Scales, 1985; Bates and Watts, 1988; SYSTAT and Wilkinson, 1990], the parameters of various model were estimated so as to minimize a Poisson loss function, following the Simplex iterative estimation minimization algorithm.

3 The above data on socio-economic variables pertained specifically to the 1974-1978 period, but the agglomeration-based lung cancer mortality ratios computed by the Health Ministry for the 1974-1978 period [Pampalon, 1985] were merged with those computed by the same source for the 1979-1983 period [Pampalon, 1986] so as to increase the statistical stability of the ratios.

by dividing the observed lung cancer SMR over the 1970-1989 period by the "expected" SMR due to non-asbestos risk factors alone. However, the weighted linear regressions did not suggest any statistical socio-economic determinants of the agglomeration-based relative risk of lung cancer, as no variable nor any combination of variables explained a significant proportion of the variance of the lung cancer SMR.

A similar series of regressions was performed with the 62 study agglomerations, forcing a dichotomous asbestos exposure status variable in the models. No covariate nor set of covariates produced a material change in the parameter estimate of the asbestos exposure status effect. However the analyses were limited by collinearity.

In fact, the relatively small number of exposed agglomerations and the large statistical imprecision of agglomeration-specific mortality and covariate data and the correlation or the overlap between most socio-demographic variables hampered the control of confounding. Thus covariates were all distributed slightly unevenly between the exposed and unexposed areas, introducing a lot of collinearity with even just a few covariates. Moreover, the available information on covariates was not age-period-specific. We had to assume temporal stability of these characteristics. Finally information on smoking, the main potential confounder, was only available at the level of the Province's 12 Health Care Areas, not at the agglomeration level. These limitations of the ecological data might explain the failure of the multivariate analyses to identify a useful set of confounders.

C.3.3. Multivariate Analyses of the 1987 Quebec Health Survey (QHS) by the Quebec Ministry of Health

In 1987, Quebec's Health Ministry conducted the first Quebec Health Survey ("Enquête Santé-Québec"), gathering data from over 32,000 Quebecois. Data on household characteristics and on children less than 15 years of age were gathered through over 12,000 interviews of

household heads; personal data on persons 15 years of age and over were obtained by self-administered questionnaires, 20,000 of which were completed.

Pampalon et al. [Pampalon et al., 1990] carried out a multivariate analysis of individual QHS data of all male and female respondents at least 15 years of age living either in a Canadian census agglomeration (adjacent municipalities totaling ≥ 10,000 pop.), in an OPDQ agglomeration (adjacent municipalities totaling ≥ 4,500 pop.) or in a "town" (any municipality ≥ 2,500 pop.). Four (4) health determinants, six (6) health state indicators (self-reported) and three (3) so-called "health consequences"[1] were correlated with various socio-economic factors using multivariate analyses, resulting in the selection of seven (7) socio-economic predictors of "health". The resulting model was applied to each census enumeration unit of a town or agglomeration of the Province of Quebec. Each unit was thus characterized as either "disadvantaged", "intermediate" or "advantaged" in 1987 in terms of socio-economic determinants of health.

The Asbestos agglomeration (19 units) and the Thetford Mines agglomeration (37 units) comprised a total of 56 census units. In terms of socio-economic determinants of health, 23% of these exposed census units were characterized as "disadvantaged" census units, 62% as "intermediate" and 15% as "advantaged" census units. In comparison, in the seven other non-metropolitan Health Care Areas of Quebec, 16% of the census units were characterized as "disadvantaged", 53% as "intermediate" and 31% as "advantaged" census units. Thus, at the end of the 1980s, the population of the exposed agglomerations seemed somewhat disadvantaged in terms of socio-economic determinants of health behaviours and health outcomes than the population of unexposed non-metropolitan areas of Quebec.

This observation for the whole population at least 15 years of age may not apply directly however to our female study population at least 30 years of age, since sex and age modify the effects of socio-economic status, smoking and drinking on the risk of lung cancer. For

[1] "Health consequences" were defined by Pampalon et al. as self-reported 1) restriction of physical mobility, 2) having used the services of a health professional in the last two weeks, or 3) having taken at least 3 medications in the last two days.

instance, in the asbestos-mining areas, males might drink and smoke more and females less than the population of the rest of Quebec. Such sex-age-cofactor interactions make it difficult to generalize these results to the female study population.

C.3.4. Specific Health Behaviours and Problems of the Older Female Population of the Exposed Agglomerations Compared With Similar Data From the Quebec Health Survey

As described in Sections B.1.4 and B.4.2 of the Exposure Assessment, we carried out a survey in 1989 among females aged 50 years and over living in the exposed agglomerations to determine the lifetime residential histories of the study population (B.4.2) and to determine when and how frequently did the study population witness visible asbestos dust exposures in the past (B.1.4). Yet another objective was to obtain data on potential confounder variables in the exposed population for comparison with data from the 1987 Quebec Health Survey (QHS) for the whole Province of Quebec. Thus questions were added to our survey inquiring about smoking, drinking, education, body mass and height, use of health services, history of medical problems. The wording of our questions was exactly the same as in the QHS.

The number of females at least 50 years of age interviewed in the QHS survey was very small by agglomeration, and the correspondence between our municipal geocodes and the postal codes used by the QHS was often incongruous, making it difficult to identify agglomerations of residence. Consequently, instead of using the 60 referent agglomerations for the comparison, I preferred the "Région Socio-Sanitaire" (Health Care Area)[1] of the QHS respondent because it was always clearly identified in the QHS dataset. There were 11 Health Care Areas[2] in the 1987 QHS.

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[1] These administrative Health Care Areas are akin to Quebec's large administrative regions which correspond to socio-economic entities with relative internal homogeneity and external differentiation.

[2] The 11 Health Care Areas were: 1) Bas-Saint-Laurent-Gaspésie (south shore of the St-Lawrence estuary), 2) Saguenay-Lac-Saint-Jean, 3) Québec (urban community of the Provincial Capital), 4) Mauricie-Trois-Rivières, 5) Estrie-Cantons de l'Est (Eastern Townships), 6) Montreal metropolitan area, 7) Laurentides-

The proportions of the female population at least 50 years of age reporting given health behaviours or events were directly age-standardized[1] in our *ad hoc* survey and in the corresponding QHS data. The standardized proportions were compared between the exposed agglomerations and the group of seven (7) Health Care Areas remaining after excluding the Montreal, Quebec, Eastern Townships and Outaouais areas. The main age-standardized responses are shown in Table C-2. Results of the QHS for the Eastern Townships Area are also indicated in the table because this area includes the agglomeration of Asbestos and is the most comparable to the two asbestos-mining agglomerations.

The female population at least 50 years of age of the Eastern Townships Area was similar to that of the other non-metropolitan regions of the Province in regard to the health variables investigated in the *ad hoc* survey. The only difference was its higher education level, probably due to the importance of the large town of Sherbrooke in the Eastern Townships.

Relative to the QHS respondents from the seven non-metropolitan regions, the respondents from the Thetford Mines agglomeration in our survey reported lower education, lower alcohol consumption, and less frequent histories of cancer and hypertension, but they reported at least as much smoking as the QHS respondents. The respondents from the agglomeration of Asbestos reported less drinking and smoking, and better health on just about every self-reported indicator. Having had a Pap smear test in the last 2 years was used as an indicator of attitudes toward health care services; there was no difference between areas.

Lanaudière, 8) Montérégie (south-west of the St-Lawrence Valley), 9) Outaouais (Ottawa River), 10) Abitibi-Témiscamingue, 11) Côte-Nord (north shore of the St-Lawrence estuary).

[1] The percentages were directly standardized to the 5-year age-distribution of the female population ≥ 50 years of age in the Province of Quebec in 1986.

Table C-2 Age-Adjusted Socio-Demographic Characteristics of Exposed and Referent Female Populations ≥ 50 Years of Age

	Present Survey (1989)			Santé-Québec (1987)	
Agglomeration / Region -->	Thetford Mines	Asbestos	2 Exposed Agglom.	Eastern Townships Region	7 Other Non- Metro. Regions
Maximum sample per question -->	n = 308	n = 235	n = 543	n = 356	n = 2,652
Born in Quebec (%)	99	96	98	96	97
Widow (%)	27	21	25	19	22
Separated or divorced (%)	2	2	2	3	3
School ≥ 10 years (%)	18	35	23	37	29
Current smokers (%)	31	14	25	29	31
Ever smoked (%)	63	35	54	55	55
Alcohol > 4 drinks / week (%)	1	1	1	6	5
Alcohol > 1 drinks / week (%)	7	7	7	16	15
Pap smear in last 2 years (%)	39	43	40	40	38
Serious illness in last year (%)	12	6	10	9	11
Lifetime prevalence of:					
Non malignant respiratory disease (%)	12	5	10	14	14
Breast cancer (%)	1	1	1	4	2
Cancer (%)	3	3	3	14	14
Hypertension (%)	32	26	30	39	39
Heart disease (%)	19	9	15	19	18
Diabetes (%)	5	5	5	7	10

Note: Proportions (percentage fractions) were directly standardized by 5-year age intervals, according to the study base's PY age distribution over 50 years of age.

These differences suggest that the exposed study population - particularly that of Asbestos - might have been less exposed to risk factors for lung cancer and for other diseases than the referent population. These potential biases are considered and are tentatively adjusted for in the following mortality analysis.

The above differences should not be taken at face value however. Our *ad hoc* survey was methodologically different from the QHS. In our survey, all respondents at least 70 years of age were interviewed in person by a single experienced female interviewer. By contrast, the QHS relied on a self-administered questionnaire for the questions concerned in the present comparison. Females in institutions were included in our survey whereas they were excluded in the QHS. Our sampling frame was the 1989 electoral list, whereas the QHS used geographic cluster sampling of city blocks and systematic random sampling of households within clusters. The questions reported in the above table constituted merely one of the three main sections of our *ad hoc* survey and were asked at the end of the questionnaire, whereas these questions were part of the very essence of the QHS and were spread throughout the questionnaire. Finally, the seven Health Care Areas for which QHS data were computed did not correspond exactly to the referent agglomerations and, in particular, comprised more rural areas. Such differences may have biased the comparison of our data on the exposed population with the QHS data on a population equivalent to the referent population.

C.3.5. Potential Confounding of Lung Cancer SMRs

Smoking was the main potential confounder of lung cancer mortality ratios. According to our *ad hoc* survey and the QHS, the age-standardized proportion of "ever smokers" among females at least 50 years of age was similar in the exposed (54%) and comparison (55%) areas. However, the age-standardized proportion of current smokers was 25.4% (95%CI: 21.7%-29.0%) in the exposed population compared to 30.6% (95%CI: 28.6%-32.7%) in the seven non-metropolitan Health Care Areas.

The potential bias that these differences could produce on the relative risk of lung cancer between the asbestos and referent agglomerations was estimated on the basis of the risks estimated by Doll and Peto in a cohort of British Doctors [Doll and Peto, 1976; Doll et al., 1980]. This approach has been used in previous risk assessments on asbestos and cancer[1]. Accordingly, a smoker would on average have a RR of for lung cancer relative to an otherwise comparable non-smoker, a RR equivalent to a consumption of 15 cigarettes per day. Ex-smokers - two thirds of which used to smoke "regularly" - were assigned a lung cancer RR of 5, the relative risk estimated in the British Doctors study for ex-smokers. Using the differential distribution of "never", "past" and "current" smokers among the exposed and referent populations surveyed, it was estimated that the lung cancer SMR for both exposed agglomerations combined might be underestimated by as much as 7% due to the presumably lower proportion of past smokers among the exposed population with respect to the comparison population.

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[1] I used the approach of the Royal Commission on Asbestos in Ontario (1984). The EPA (Nicholson, 1986) relied on lung cancer mortality rates in nonsmokers estimated by Garfinkel in 1981, on population rates and on age-sex-specific proportions of smokers estimated in 1966 by Hammond, for an estimated RR=9 in women. The HEI-AR (1991) used a smoker/non-smoker lung cancer RR of about 15. The ratio may change between assessments, but all assume a constant RR.

To summarize the data on potential confounders, in the mid-1970s the whole population of the exposed agglomerations seemed advantaged socio-economically relative to the population of the referent agglomerations, and alcohol and tobacco sales were lower in the larger regions comprising the exposed agglomerations. In the late 1980s, the population at least 15 years of age of the exposed agglomerations was on average somewhat disadvantaged socio-economically[1] and in terms of social determinants of health relative to the referent population. Moreover, according to our *ad hoc* survey, the older female population of the exposed agglomerations reported less detrimental health habits and less major health problems than in non-metropolitan regions of Quebec. The socio-economic decline of the exposed agglomerations in the last half of the follow-up period was consistent with the dramatic decline of the asbestos industry in the early 1980s. This relatively recent socio-economic decline probably did not have an immediate effect on cancer mortality of the older female population. However, mortality of the exposed population over the study period could have been lower than that of the referent population due to the better socio-economic status of the exposed population that prevailed during the most prosperous decades of the asbestos industry and of its labour force, that is from the mid-1950s until the late 1970s.

SMRs might be biased downward by the above-mentioned differences in socio-economic profile and health behaviour between the two compared populations. Indeed such a downward bias is suggested by the lower self-reported lifetime occurrences of various health problems in the exposed population. Therefore, the SMRs for lung cancer and other causes of death would tend to be biased downward and, in particular, the lung cancer SMR could be underestimated by 7% due to differences in smoking patterns, a bias which could be even larger due to differences in other determinants of health.

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[1] The apparent socio-economic decline of the asbestos-mining agglomerations over the 1970-1989 period corresponds with the slump of the asbestos industry at the end of the 1970s.

C.4. Results of the Mortality Study

This section provides the main results of the Mortality Study over the 1970-1989 period. The SMRs and SPMRs summarized age-calendar-year-specific mortality ratios for most causes of death. The chi-square test of homogeneity of the SMR was not statistically significant for lung cancer ($\chi^2 = 5.59$, $p = 0.348$), nor for most causes of death The exceptions were laryngeal cancer ($\chi^2 = 21.64$ $p = 0.001$) and asbestosis ($\chi^2 = 10^8$, $p = 0.000$) which had caused only 1 and 2 deaths respectively. Although no test of heterogeneity was performed for the SPMRs[1], results would be practically identical since any test of heterogeneity is based on the proportional distributions of observations and expected counts among the different age-calendar-year strata.

C.4.1. Standardized Mortality Ratios (SMRs)

Tables C-3a, C-3b, C-3c, and C-3d show SMRs for a) Major Causes of Death, b) Selected Circulatory and Respiratory Causes of Death, c) Major Cancer Site Categories and d) Respiratory and Digestive Cancer Sites. SRRs were also computed but are not shown because they were virtually identical to the corresponding SMRs for each cause of death examined. The age-calendar year standardized SMRs compare cause-specific mortality rates of the exposed study population to those of the referent study population.

[1] I have not found references about a test of heterogeneity applied to SPMRs.

In the exposed population, mortality from *all causes* was significantly[1] lower than in the referent population, as can be seen in Table C-3a. SMRs were lower by 11% in Thetford Mines, by 6% in Asbestos and by 9% for both agglomerations combined. The deficit in total mortality (-200 deaths) was mainly attributable to significant deficits in mortality by circulatory diseases (-120 deaths), neoplastic diseases (-48 deaths), non-neoplastic respiratory diseases (-20 deaths), and non-neoplastic digestive diseases (-17 deaths). These deficits were proportionately equivalent in each asbestos agglomeration.

Asbestosis was the only non-neoplastic disease for which there was a statistically significant excess mortality (Table C-3b). Despite the fact that only 1 death by asbestosis occurred in each asbestos-mining agglomeration, the SMR for asbestosis was extremely elevated due to the very low incidence of this disease in the referent female population. Acute myocardial infarction was the cause of death which accounted for the largest number of excess deaths (+23 deaths), but this excess was not significant statistically (p=0.25).

There was no statistically significant deficit nor excess of any major cancer site category (Table C-3c) in the two agglomerations combined. However, there was a significant excess of "eye, brain and central nervous system" cancers (+5.5 deaths) in the agglomeration of Asbestos, and a significant deficit of genital cancers (-13 deaths) in Thetford Mines. As for cancer categories that have been more or less associated with asbestos exposure in the scientific literature, respiratory cancers showed a seeming excess of 6% (+4.6 deaths, 95%CI: 0.84-1.32), whereas digestive cancers (-4%), oral cancers (-32%) and urinary cancers (-16%) were all lower than expected.

The SMR for broncho-pulmonary cancer was 0.99 for both exposed agglomerations combined, 0.98 in Thetford Mines and 1.03 in Asbestos. There was a remarkable excess of pleural

[1] Although the basic approach of the mortality study was to estimate excess relative risks rather than to test a universal and blind null hypothesis of no excess risk, some measures of effect were termed *"significant"* or *"statistically significant"* to mean that the parameter's 95%CI did not include - or nearly excluded - the null value of 1.0 for risk ratios or 0% for excess relative risks. These expressions were simply used as approximate indications of sampling errors relative to the null values. The controversial issue of defining and correcting for "multiple comparisons" was irrelevant as the focus was on a single outcome - lung cancer mortality - and as statistics on other mortality ratios were interpreted merely as relative indications of sampling error.

cancers in both exposed agglomerations combined (7 observed vs. 0.9 expected), an excess entirely concentrated in the agglomeration of Thetford Mines. The 6 excess pleural cancer deaths accounted for the 6% excess respiratory cancers for both agglomerations combined. No pleural cancer was observed in Asbestos, but this could be due to random variation (0.3 death was expected, and the SMR upper 95%CL was 11.8). Among digestive cancers, the 18% excess risk (+8 deaths) for stomach cancers and 22% deficit (-11 deaths) for "other digestive cancers" were not significant.

Table C-3a SMRs for Major Causes of Death Among Females, by Exposed Agglomeration, 1970-1989

	Asbestos			Thetford Mines			Both Mining Agglomerations		
Cause of Death	n	SMR	95%CI[1]	n	SMR	95%CI	n	SMR	95%CI
All Causes of Death[2]	787	0.94	0.88-1.01	1,455	0.89	0.84-0.94	2,242	0.91	0.87-0.95
Infective + Parasitic Diseases[3]	7	1.35	0.54-2.77	9	0.89	0.41-1.69	16	1.04	0.60-1.70
Blood, Endocrine, Metabolic	43	1.05	0.76-1.42	96	1.20	0.97-1.47	139	1.15	0.97-1.36
Mental + Nervous Disorders	18	0.94	0.56-1.49	33	0.85	0.59-1.20	51	0.88	0.66-1.16
Circulatory Diseases	376	0.91	0.82-1.00	711	0.88	0.81-0.94	1087	0.89	0.83-0.94
Respiratory Diseases	33	0.76	0.52-1.07	71	0.83	0.65-1.04	104	0.81	0.66-0.98
Digestive Diseases	26	0.85	0.56-1.25	42	0.70	0.51-0.95	68	0.75	0.58-0.95
Accidents, Unnatural Causes	46	1.34	0.98-1.79	64	0.93	0.72-1.19	110	1.07	0.88-1.29
Ill-Defined Symptoms	10	1.34	0.64-2.46	11	0.76	0.38-1.36	21	0.96	0.59-1.46
Other Non-Neoplastic Dis.	16	0.80	0.46-1.30	35	0.88	0.61-1.22	51	0.85	0.64-1.12
Neoplastic Diseases	212	0.97	0.84-1.11	383	0.90	0.81-0.99	595	0.92	0.85-1.00

1 Confidence limits were computed using Byar's approximation.

2 The SMRs for all non-neoplastic diseases were similar to to those for total mortality. There was a total of 1,647 deaths in both exposed agglomerations attributed to non-neoplastic causes, for a SMR of 0.90 with a 95%CI of 0.86-9.95. The agglomeration-specific SMRs were 0.94 in Asbestos (n=575) and 0.89 in Thetford mines (n=1,072).

3 Infective and parasitic diseases *other than* pulmonary or respiratory.

Table C-3b SMRs for Selected Circulatory and Respiratory Causes of Death Among Females, by Exposed Agglomeration, 1970-1989

	Asbestos			Thetford Mines			Both Mining Agglomerations		
Cause of Death	n	SMR	95%CI	n	SMR	95%CI	n	SMR	95%CI
Circulatory Diseases									
Acute Myocardial Infarction	141	1.07	0.90-1.26	271	1.06	0.94-1.19	412	1.06	0.96-1.17
Other Ischemic Heart Diseases	63	0.77	0.59-0.99	122	0.76	0.63-0.91	185	0.77	0.66-0.89
Hypertensive Diseases	11	0.72	0.36-1.29	30	1.01	0.68-1.44	41	0.91	0.65-1.24
Other Circulatory Diseases	161	0.87	0.74-1.01	288	0.79	0.70-0.88	449	0.81	0.74-0.89
Respiratory Diseases									
Asbestosis	1	35.85	0.47-199.48	1	17.47	0.23-97.21	2	23.49	2.64-84.83
Other Pneumoconioses	0	0.00	--- - ∞	0	0.00	--- - ∞	0	0.00	--- - ∞
Chronic Bronchitis, etc.[1]	14	0.81	0.44-1.35	25	0.73	0.47-1.08	39	0.76	0.54-1.03
Tuberculoses+ Other Respir.	18	0.70	0.41-1.10	45	0.88	0.64-1.17	63	0.82	0.63-1.04

[1] Chronic Bronchitis, Emphysema, Asthma

Table C-3c SMRs For Major Cancer Site Categories Among Females, by Exposed Agglomeration, 1970-1989

	Asbestos			Thetford Mines			Both Mining Agglomerations		
Cancer Site	n	SMR	95%CI	n	SMR	95%CI	n	SMR	95%CI
Neoplastic Diseases	212	0.97	0.84-1.11	383	0.90	0.81-0.99	595	0.92	0.85-1.00
Respiratory Cancers	27	1.03	0.68-1.50	55	1.08	0.81-1.40	82	1.06	0.84-1.32
Digestive Cancers	66	0.91	0.70-1.16	139	0.99	0.83-1.17	205	0.96	0.83-1.10
Oral Cancers	1	0.50	0.01-2.76	3	0.78	0.16-2.27	4	0.68	0.18-1.74
Breast Cancer	44	0.95	0.69-1.28	76	0.85	0.67-1.06	120	0.88	0.73-1.06
Genital Cancer	27	1.04	0.69-1.52	37	0.74	0.52-1.02	64	0.85	0.65-1.08
Urinary Cancer	5	0.66	0.21-1.53	14	0.94	0.51-1.58	19	0.84	0.51-1.32
Bone Tissue, Skin	2	0.62	0.07-2.23	2	0.32	0.04-1.15	4	0.42	0.11-1.07
Eye, Brain, C.N.S.	11	2.01	1.00-3.60	7	0.65	0.26-1.35	18	1.11	0.66-1.76
Endocrine Glands	1	0.85	0.01-4.73	5	2.19	0.71-5.12	6	1.74	0.63-3.78
Lymphatic and Hematopoietic	12	0.66	0.34-1.15	30	0.84	0.57-1.20	42	0.78	0.56-1.05
Ill-Defined, Other, Unspecified	16	1.50	0.86-2.43	15	0.73	0.41-1.20	31	0.99	0.67-1.40

Table C-3d SMRs for Respiratory and Digestive Cancer Sites Among Females, by Exposed Agglomeration, 1970-1989

	Asbestos			Thetford Mines			Both Mining Agglomerations		
Cancer Site	n	SMR	95%CI	n	SMR	95%CI	n	SMR	95%CI
Respiratory Cancers									
Larynx	1	0.94	0.01-5.21	1	0.49	0.01-2.73	2	0.64	0.07-2.33
Lung, Bronchus	25	1.03	0.67-1.52	46	0.98	0.71-1.30	71	0.99	0.78-1.26
Pleura	0	0.00	0.00-11.79	7	11.56	4.63-23.81	7	7.63	3.06-15.73
Other Respiratory	1	1.54	0.02-8.59	1	0.81	0.01-4.49	2	1.06	0.12-3.83
Digestive Cancers									
Œsophagus	2	1.05	0.12-3.78	3	0.82	0.17-2.41	5	0.90	0.29-2.10
Stomach	14	1.16	0.63-1.94	28	1.20	0.79-1.73	42	1.18	0.85-1.60
Small Intestine	1	2.56	0.03-14.24	1	1.31	0.02-7.29	2	1.73	0.20-6.26
Colon and Rectum	35	0.96	0.67-1.34	70	0.99	0.77-1.25	105	0.98	0.80-1.19
Peritoneum	1	1.24	0.02-6.89	2	1.28	0.14-4.62	3	1.27	0.25-3.70
Other Digestive[1]	13	0.62	0.33-1.06	35	0.86	0.60-1.19	48	0.78	0.57-1.03

[1] Pancreas, Liver, Biliary, etc.

C.4.2. Proportional Mortality Ratios (SPMRs)

Tables C-3e and C-3f show respectively the SPMRs for Major Cancer Site Categories and those for Respiratory and Digestive Cancer Sites. All cause-specific SPMRs were higher than their corresponding SMRs by about 10% for both asbestos-mining agglomerations combined, by about 6% for Asbestos and 12% for Thetford Mines.

As indicated by the SPMRs in Table C-3e, there was a non-significant 2% excess proportional mortality (+10 deaths) due to all neoplastic diseases; neither was there any statistically significant deficit nor excess of a major cancer site category[1]. Regarding categories more or less associated with asbestos exposure in the scientific literature, respiratory and digestive cancers showed non-significant excesses of 17% (+11.9 deaths) and 6% (+11.6 deaths) respectively in proportional mortality, oral and urinary cancers showed non-significant deficits of 25% (-1.3 deaths) and 6% (-1.2 deaths) respectively in proportional mortality.

Among SPMRs for specific cancer sites, there was a non-significant 10% excess of broncho-pulmonary cancers in each asbestos-mining agglomeration, and in both agglomerations combined, the 95%CI of the excess SPMR extending from -14% to +39%, and its compatibility with the null hypothesis being p=0.23 (one-sided). The extraordinary excess of pleural cancers in Thetford Mines was similar to that measured with the SMR. The 31% excess of stomach cancers in both agglomerations combined was almost statistically significant (95%CI: -3% to +76%).

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[1] There were, however, some agglomeration-specific excess deaths due to diseases which have not previously been associated with asbestos exposure. In the agglomeration of Asbestos, an excess of 5.7 deaths due to "eye, brain and c.n.s." and an excess of 2.6 deaths due to "ill-defined", "other" or "unspecified" cancers were statistically significant. In Thetford Mines, an excess of 3.0 deaths due to cancer of the endocrine glands was also statistically significant.

Table C-3e Standardized PMRs for Major Cancer Site Categories Among Females, by Exposed Agglomeration, 1970-1989

	Asbestos			Thetford Mines			Both Mining Agglomerations		
Cancer Site	n	SPMR	95%CI[1]	n	SPMR	95%CI	n	SPMR	95%CI
Neoplastic Diseases	212	1.03	0.92-1.14	383	1.01	0.93-1.10	595	1.02	0.96-1.10
Respiratory Cancers	27	1.10	0.76-1.58	55	1.21	0.94-1.57	82	1.17	0.95-1.45
Digestive Cancers	66	0.98	0.78-1.23	139	1.11	0.95-1.30	205	1.06	0.93-1.21
Oral Cancers	1	0.54	0.08-3.80	3	0.87	0.28-2.68	4	0.75	0.28-2.00
Breast Cancer	44	0.99	0.75-1.31	76	0.95	0.77-1.18	120	0.97	0.82-1.15
Genital Cancer	27	1.10	0.77-1.58	37	0.83	0.61-1.14	64	0.93	0.73-1.18
Urinary Cancer	5	0.71	0.30-1.69	14	1.06	0.63-1.78	19	0.94	0.60-1.46
Bone Tissue, Skin	2	0.63	0.16-2.43	2	0.35	0.09-1.38	4	0.45	0.17-1.20
Eye, Brain, C.N.S.	11	2.08	1.17-3.72	7	0.73	0.35-1.53	18	1.21	0.77-1.92
Endocrine Glands	1	0.88	0.12-6.23	5	2.44	1.02-5.84	6	1.89	0.85-4.18
Lymphatic and Hematopoietic	12	0.69	0.40-1.21	30	0.94	0.66-1.33	42	0.85	0.63-1.15
Ill-Defined, Other, Unspecified	16	1.60	0.99-2.59	15	0.81	0.49-1.34	31	1.09	0.77-1.54

[1] Confidence limits for SPMRs were computed using the approximate formula 2.17 (p.77) for SPMRs in Vol. 2 of Breslow and Day's "Statistical Methods in Cancer Research" [Breslow and Day, 1987].

Table C-3f Standardized PMRs for Respiratory and Digestive Cancer Sites Among Females, by Exposed Agglomeration, 1970-1989

	Asbestos			Thetford Mines			Both Mining Agglomerations		
Cancer Site	n	SPMR	95%CI	n	SPMR	95%CI	n	SPMR	95%CI
Respiratory Cancers									
Larynx	1	1.03	0.15-7.11	1	0.55	0.08-3.85	2	0.72	0.18-2.83
Lung, Bronchus	25	1.10	0.75-1.60	46	1.10	0.83-1.46	71	1.10	0.88-1.38
Pleura	0	0.00	0.00-11.78	7	12.71	6.08-26.57	7	8.21	3.92-17.18
Other Respiratory	1	1.65	0.23-11.69	1	0.90	0.13-6.32	2	1.17	0.29-4.65
Digestive Cancers									
Œsophagus	2	1.14	0.28-4.52	3	0.93	0.30-2.86	5	1.00	0.42-2.40
Stomach	14	1.24	0.74-2.07	28	1.34	0.93-1.93	42	1.31	0.97-1.76
Small Intestine	1	2.69	0.39-18.73	1	1.45	0.20-10.26	2	1.88	0.47-7.51
Colon and Rectum	35	1.03	0.75-1.42	70	1.11	0.89-1.40	105	1.08	0.90-1.31
Peritoneum	1	1.32	0.19-9.22	2	1.43	0.36-5.69	3	1.39	0.45-4.30
Other Digestive[1]	13	0.67	0.39-1.14	35	0.97	0.70-1.34	48	0.86	0.65-1.14

[1] Pancreas, Liver, Biliary, etc.

C.5. Discussion

The cause-specific mortality of the female population at least 30 years of age in the asbestos-mining agglomerations was compared to that of the study population in 60 agglomerations of Quebec over the 1970-1989 period. According to the SMR analysis, there was no excess lung cancer mortality in the exposed agglomerations (-0.5%), whereas a +10% excess was estimated on the basis of the SPMR. Although the lung cancer SMR and SPMR were both compatible with a null hypothesis of no excess risk of lung cancer, the main purpose of the analysis was to obtain a point estimate with confidence limits of the excess risk of lung cancer mortality to be compared with projections based on the EPA's linear exposure-effect model. In fact, the EPA model was the real "null hypothesis" in the present thesis; hence the best estimate of excess lung cancer mortality in the exposed population will be compared to the projections based on the EPA model and other risk assessments in the next chapter. Accordingly, the present discussion focused on finalizing the best lung cancer risk estimate rather than on testing a theoretical null hypothesis of no effect[1].

Some limitations of the data may have been amplified by the ecological study design. Still, key features of the populations, geographic areas and asbestos-mining concentration made this ecological study efficient and robust. As explained in Section A.4, asbestos exposures of the exposed population were homogeneously orders of magnitude above those of the referent population. Accordingly, migration, confounding and misclassification errors were not nearly as contrasted as asbestos exposure and would thus have had minimal effect on the large

[1] Of course, a best estimate of effect statistically that would be compatible by random error alone with the null effect value would be less meaningful than an estimate whose confidence interval did not comprise the null value. It would nevertheless remain the best estimate, contrary to the null value. Moreover, in the present state of knowledge, a null hypothesis of no effect of asbestos exposure on lung cancer risk has rather less credibility than a hypothesis of a positive relationship.

relative risk predicted by the EPA model (see Section D.2). Furthermore, the exclusion of Quebec's largest urban areas reduced confounding, the effect of misclassification of place of residence at time of death, and the effect of inter-regional migrations. Finally, the study population was relatively stable, with little migration in the older age groups who are at highest risk of lung cancer. The discussion shows how these characteristics have checked most potential biases.

C.5.1. Confounding and Residual Selection Bias

In Section C.3, multivariable statistical analyses failed to control for potential confounding in this small ecological dataset of 2 exposed and 60 referent observations; agglomeration-specific lung cancer mortality rates were too unstable statistically relative to the potential effects of ecological covariates. Still, despite the overall socio-economic comparability of the exposed and referent populations, it was concluded in Section C.3.5 that the lung cancer SMR might be underestimated by at most 7% due to moderate differences in smoking patterns, and that this bias might be accentuated by small but consistent differences in other behavioural and socio-economic determinants of health. This purported downward bias in mortality due to lung cancer and other causes of death was evidenced 1) by the lower self-reported lifetime prevalence of various diseases in the exposed population (non-malignant respiratory diseases, cancer, breast cancer, hypertension, heart diseases), 2) by the low SMRs of the exposed population for the corresponding causes of death (non-neoplastic respiratory diseases, cancer, breast cancer, hypertension and heart diseases), and 3) by the low SMRs for all causes of death combined (SMR=0.91), for neoplastic (SMR=0.92) and non-neoplastic (SMR=0.89) causes of death. Barring the unlikely possibility that asbestos exposure might protect against major causes of death, the low SMRs supported the analysis of potential confounders in suggesting a lower baseline risk of mortality for the major causes of death, including most cancers and probably lung cancer.

C.5.2. Misclassification of Cause of Death

The death certificate data in Quebec's Mortality Registry were used to classify the exposure and outcome status of the cases. The agglomeration of residence at time of death and the cause of death have been assessed independently one from the other, and misclassification of each variable must have been random. Nevertheless, diagnostic misclassification of lung cancer may still have been differential with respect to exposure status of the agglomeration. A reliability study [Pampalon, 1981] of deaths coded as cancers in Quebec's Mortality Registry over the period 1966-1977 estimated that, for the whole of Quebec, 8.5% of deaths coded as respiratory cancers were really due to other causes, while 1.0% of deaths coded as other cancers were really respiratory cancers. Since there were 6.3 times more deaths due to other cancers than due to respiratory cancers in our study population, respiratory cancer mortality could have been underestimated by 2.2% in the Province. This difference would be canceled out if about 1% of non-malignant respiratory diseases were really false negative respiratory cancers, a possibility observed by other investigators [Newhouse and Wagner, 1969; McDonald et al., 1971; Nicholson et al., 1979][1]. If such misclassifications were distributed proportionately between the exposed and referent populations, there would be no resulting bias. Unfortunately, the document does not compare the diagnostic reliability of lung cancer across regions. As to the diagnostic reliability of cancer [Pampalon, 1981, p.30], the proportion of cancers[2] correctly diagnosed as cancers was similar in the Eastern Townships (100%) and in comparable Health Care Areas[3] (99.3%). However, the proportion of correctly diagnosed cancer sites in the Eastern Townships (85%) was somewhat higher than in comparable Health Care Areas (77%), implying that lung cancers could be more

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[1] In a review of the Newhouse and Wagner reporting data on 301 ex-asbestos factory workers [Doll and Peto, 1987], a table (Table 4-2) shows that 39 deaths were attributed to lung cancer on the death certificates, but this number was revised to 42 cases, after reviewing hospital and pathological data. 5 "lung cancers" were in reality pleural mesotheliomas; on the other hand, 2 actual lung cancers had been misdiagnosed as "other cancers" (neither respiratory nor digestive), 4 as "asbestosis", and 2 as "other non-malignant diseases".

[2] The proportion of deaths coded as cancers that were really neoplastic diseases, whether the site was correctly identified or not, after verification with various sources: original death certificate, hospital records, Tumour Registry, etc.

[3] Excluding the Montreal, Quebec and Outaouais areas.

overdiagnosed in the referent than in the exposed population. If true, this would tend to bias the lung cancer SMR toward the null, unless the alleged overdiagnosis differential was compensated by a proportionate underdiagnosis differential. In the absence of overdiagnosis and underdiagnosis data specific to lung cancer, the direction and size of a potential bias cannot be assessed.

C.5.3. Misclassification of Exposure and Losses to Follow-Up

All persons who have ever resided in a Quebec asbestos-mining agglomeration have been exposed to environmental asbestos. Since exposure status was determined from death certificate data, misclassification of exposure occurred when a former female resident of an asbestos-mining agglomeration moved to a referent agglomeration before dying. Assuming a positive relation between asbestos exposure and the risk of dying from lung cancer, such exposure misclassification would have biased the lung cancer SMR and SPMR toward the null. However, it is unlikely that such a bias produced more than a negligible effect. Given that we excluded the largest urban areas and the smallest rural areas from the study and referent population, out-migrants from the exposed area would not have been much attracted by the referent agglomerations which did not offer better economic, infrastructural or social prospects than the asbestos-mining agglomerations. Out-migrants would rather have been drawn towards the resources and jobs of large urban centres or by the quietness and the social network in non-agglomerated rural areas where they were born or where families and friends lived[1].

Other exposure misclassification such as miscoding an exposed municipality of residence on a death certificate as a referent municipality or *vice versa* was unlikely, it should have been simply proportional to the population of each agglomeration independently of its exposure status. Thus it should not have affected the SMRs and SPMRs.

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[1] In our *ad hoc* survey, among respondents who lived away from but close to the asbestos-mining agglomerations in 1989, many resided in the region where they were born after having lived in an asbestos-mining town for many years.

The main problem consecutive to out-migration was *loss to follow-up* since out-migrants would most likely move to a municipality or area excluded from the study. Losses to follow-up have naturally reduced the statistical power of our study[1], but they could also have biased the lung cancer SMR and SPMR if the age-year-adjusted lung cancer RR between out-migrants and non-migrants of the exposed population differed from that of the referent population. Economically and socially motivated migration, which constitutes the main component of inter-regional migration, would not have induced such a bias. However, lung cancer SMR and SPMR would be biased *if a)* new lung cancer cases were more likely to move to large urban centres than other residents of the study agglomerations (for instance, to obtain specialized health care) *or b)* they were more likely to be hospitalized for many months in an outside hospital[2], *or c)* the propensity of lung cancer cases to migrate out or to die in an outside hospital differed between the exposed and referent populations. Yet this hypothetical bias must have smaller than 1%[3] since the exposed and referent regions did not have materially different access to health care services and since this type of migration would have constituted only a small fraction of the <2% yearly out-migration rate of agglomerations.

C.5.4. Lung Cancer SPMR: Best Estimate of the Effect of Non-Occupational Asbestos Exposure in the Asbestos-Mining Agglomerations

While the SMR is often considered a better estimate of relative risk than the SPMR because the SMR for each cause is independent of the SMRs for other causes, the SPMR may sometimes be closer to the true relative risk than the SMR, providing that deaths due to the cause of interest constitute only a small proportion of all deaths. Thus, if the exposed and unexposed

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[1] This effect is already accounted for in that the estimated confidence intervals of the SMRs and SPMRs are larger than if there had been no losses to follow-up.

[2] If they died more frequently after a stay of at least six months in an outside hospital, the hospital's address would be indicated on the death certificate as the place of residence at time of death.

[3] If 2% of the population migrated out of the study agglomerations each year, and if out-migrants had a lung cancer RR of 1.5 vs. non-migrants of the same age in the referent population and a corresponding RR of 2.0 in the exposed population, the lung cancer SMR between exposed and referent populations would be underestimated by less than 1%.

populations do not have the same background risk of lung cancer mortality, the SPMR may correct biases which act roughly equally on lung cancer mortality and on all causes of death combined [Breslow et al., 1987; Checkoway et al., 1989]. In our study, the lung cancer SPMR was 10% higher than the SMR, this difference was larger than the 7% estimated for smoking alone. In fact, all cause-specific SPMRs were 7-12% higher than their corresponding SMRs, as would be expected with an SMR of 0.91 for all causes of death. As a whole, the SPMR was considered as correcting the main suspected confounding (7%) and misclassification biases mentioned above.

It may be argued that the different background characteristics of the exposed and referent populations might act roughly equally on most malignant causes of death but not so on non-malignant causes of death. If true, it would be preferable to express "the mortality for a particular cancer site as a proportion of all cancer mortality" [Checkoway et al., 1989b] a ratio that has been termed the Standardized Proportionate Cancer Mortality Ratio or SPCMR [McMichael, 1976]. In the present study, the lung cancer SPCMR and SPMR differed merely by 1% and thus one did not have a real advantage over the other. However, contrary to the SPCMR, the SPMR could be applied to all causes of death, many of which (e.g. heart and circulatory diseases, respiratory diseases, etc.) would have been affected by lower lifetime smoking prevalence and relatively advantageous socio-economic status. Finally, potential biases due to misclassification of outcome and exposure would affect most causes of death, not only cancer.

The SPMR seemed the most prudent estimate of the relative risk of dying from lung cancer due to asbestos exposure in the exposed population.

C.5.5. Mixed Occupational and Non-Occupational Exposures

Although it would have been preferable to assess the risk of lung cancer only among females who had never been exposed to asbestos occupationally, it was not possible to do so[1]. Consequently, if female residents with past occupational asbestos exposure had been at much greater risk of developing lung cancer, attributing those "occupational cases" to non-occupational exposure would have biased the SMR and SPMR upward. However, this conjecture was unlikely in view of two indirect indications. First, among the 440 female workers included in the large cohort study [McDonald et al., 1993] of asbestos workers in the Asbestos and Thetford Mines areas, 84 had died up to 1976[2], including 1 lung cancer (vs. 1.19 expected) and 1 mesothelioma. Thus, although the evidence is limited, there is no indication yet that the risk of lung cancer was much higher in female asbestos workers than in the general female population of the area (this study). Second, since occupational asbestos exposure contributed less than 7% to the estimated average cumulative asbestos exposure of the female residents of the asbestos-mining agglomerations, occupational exposure should have contributed less than 7% to the excess SPMR [7% x (1.10 - 1) = 0.007] for lung cancer according to the linear exposure-effect model, a negligible bias.

C.5.6. Consistency and Meaning of the Results in View of Other Non-Occupational Asbestos Studies

Other epidemiological studies have looked at cancer mortality [Wigle, 1977; Pampalon et al., 1982; Loslier, 1983] or incidence [Graham, 1981] in Quebec's asbestos-mining area for different time periods between 1966 and 1977, using different spatial units and different referent populations. In addition, SMRs have been computed for each of the 67

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[1] The death certificate data had been stripped from all individual identifiers which could have been matched with lists of past asbestos workers.

[2] Unfortunately, follow-up of the female members of this cohort has not been updated since 1975.

agglomerations over the 1974-1978 and 1979-1983 periods by Quebec's Health Ministry [Pampalon, 1985; Pampalon, 1986]. The lung cancer SMRs of the asbestos-mining area differed substantially between these studies, but the variations were easily accountable by random statistical variation. The study with greatest statistical power covered the 1966-1977 period [Pampalon et al., 1982]. Despite differing age distributions, referent populations and follow-up periods, the Pampalon study produced SRRs nearly identical to those obtained in the present study for total mortality (SRR=0.90) and cancer mortality (SRR=0.91). The 6% discrepancy for respiratory cancers (SRR=1.00) and that of 11% for digestive cancers (SRR=1.06) could be explained by random statistical variation alone. As to statistical precision, the 95%CI (0.85-1.32) of the respiratory cancer SRR in the present study was half as large as in the Pampalon study (0.63-1.49). Lung cancer mortality was not assessed specifically in the Pampalon study, neither was it in a follow-up report on that study [Siemiatycki, 1982]. To conclude, the present study was consistent with previous studies in the same area, but doubled the statistical precision and looked at lung cancer mortality specifically.

Four epidemiological studies have borne on non-occupational asbestos exposures and lung cancer in different populations and exposure settings from those in the present study (Table C-4). Three have found higher relative risks of lung cancer than in the present study. The lung cancer SPMR estimated in the present study was lower than the estimate (SMR=1.24, 95%CI=1.00-1.53) obtained from pooling the 4 other studies. The difference between those studies and ours might be due to: a) random statistical variation, b) different doses, exposure levels or durations, c) different carcinogenic potentials according to type of asbestos fibre and industrial process, or d) risk overestimation biases. Airborne concentrations of asbestos fibre could hardly have been as high in other non-occupational exposure environments as in the present study, except maybe for the South African crocidolite mining area[1]. On the other hand,

[1] It can be surmised that this study population probably experienced the highest asbestos exposure, followed by the South African study population whose larger geographical area probably diluted the average level of the most exposed population, while the three other study populations must have been significantly less exposed to asbestos fibres.

the results of the environmental studies seem to parallel the carcinogenic differences between asbestos fibre types and industrial processes suggested by the occupational studies. Notwithstanding these purported differences, incorporating the present study reduced the pooled estimate to a still statistically significant SMR=1.18 (95%CI=1.00-1.37) for lung cancer mortality. However, had we used this study's SMR rather than the SPMR, the pooled SMR would be lowered to a non-significant 1.12 (95%CI=0.95-1.31). If the study on South African crocidolite-mining districts were excluded to focus on non-crocidolite fibres, the pooled estimate and confidence interval of the three other studies would be identical to those of the present study's SPMR, and the pooled "non-crocidolite" estimate including this study's SPMR would be lowered to a non-significant SMR=1.10, 95%CI=0.92-1.30. Had this study's SMR been used instead of the SPMR, the pooled SMR would have been lowered to a non-significant 1.04 (95%CI=0.87-1.23).

The present mortality study has nearly doubled the direct quantitative information available on the risk of lung cancer of populations non-occupationally exposed to asbestos in terms of the cumulative number of expected or observed cases in various studies. Despite the tremendously high cumulative exposure of this study's exposed population, our results lower the pooled estimate of relative risk based on all available studies on lung cancer and environmental asbestos. More importantly however, it is the first non-occupational study to provide a quantitative estimate of the study population's exposure, providing an opportunity to characterize the risk of cancer associated with non-occupational exposure to asbestos.

Table C-4 Comparison of the Relative Risks of Lung Cancer Estimated in Five Non-Overlapping[1] Epidemiological Studies on Non-Occupational Asbestos Exposures

Study / Pooled dataset	Observed Cases	Expected Cases	Observed / Expected	O/E 95%CI
Individual studies				
Crocidolite-Mining Districts (S. Africa) [Botha et al., 1986]	21	9.7	2.16	1.34-3.31
Families of amosite factory workers (20-year latency) (New Jersey) [Anderson, 1982]	20	10.8	1.85	1.13-2.86
Wives of asbestos-cement workers (chrysotile+crocidolite) (Italy) [Magnani et al., 1993]	6	4.0	1.50	0.55-3.26
Male residents in neighbourhood of amosite factory (New Jersey) [Hammond et al., 1979a]	41	46.2	0.89	0.67-1.15
This Study (SPMR)	71	64.5	1.10	0.86-1.39
Pooled datasets				
First 4 Studies (excluding this study)	88	70.7	1.24	1.00-1.53
All 5 Studies (including this study)	159	135.2	1.18	1.00-1.37
3 non-crocidolite-mining studies (excluding this study)	67	61.0	1.10	0.85-1.39
All 4 non-crocidolite-mining studies (including this study)	138	125.5	1.10	0.92-1.30

[1] Only epidemiological studies whose populations did not overlap are listed in the table.

C.6. Conclusion

On the basis of age-year-adjusted SPMRs, the estimated excess relative risk of lung cancer in the female population of the agglomerations of Asbestos and Thetford Mines was +10% over the 1970-1989 period, with a 95% confidence interval ranging from -14% to +39%. The lung cancer SPMR was identical in each of the two asbestos-mining agglomerations. Although the exposure history of individual cases was not ascertained, occupational exposures of the female study population would not likely explain the estimated excess risk of lung cancer mortality. The excess risk would be mostly attributable to non-occupational exposures to chrysotile asbestos. The present study corroborated but lowered the pooled estimate of the association between lung cancer risk and non-occupational asbestos exposures based on the whole epidemiological evidence available to this day. The estimate obtained from pooling this study with available studies on populations non-occupationally exposed to asbestos suggests a significant excess risk of lung cancer of +18%, with a 95% confidence interval ranging from 0% to +30%. These pooled datasets included populations with different exposure circumstances.

The SPMR was preferred to the SMR because its 10% higher estimation of the RR seemed to correct for suspected biases that might have depressed the SMR estimate. This subjective decision may have been wrong however. Had the SMR been used instead of the SPMR, the excess RR in this study would have been estimated at -0.5% with a 95%CI of -22% to +26%, and the pooled estimate from studies of non-occupationally exposed populations would have been +12%, with a 95%CI of -5% to +31%.

As to other asbestos-related causes of death, there was a large and statistically significant excess of pleural cancer (7 cases) and asbestosis (2 cases). The incidence of pleural mesothelioma is presently being investigated in a separate study over the 1970-1989 period with extensive case ascertainment and pathological review. For both asbestos-mining agglomerations combined, there was no excess mortality from other cancers previously associated with asbestos exposure such as digestive and laryngeal, oral and kidney cancers. This suggests that risk assessments on non-occupational exposure to asbestos fibres are correct in focusing their estimates on pulmonary and mesothelial cancers.

PART D. COMPARISON OF RISK ASSESSMENTS WITH THE OBSERVED EXCESS MORTALITY FROM LUNG CANCER

D.1. Introduction

The main objective was to compare the excess risk of lung cancer predicted by the EPA's risk assessment model with the excess risk of lung cancer observed in the female population of Quebec's asbestos-mining agglomerations. The crucial exposure-effect gradient parameter used by the EPA to predict risks in the general population is estimated from historical occupational cohort studies. I examined the impact of using different occupational studies or weighing them differently for predicting risk by comparing predictions based on the EPA' risk assessment on asbestos with those based on other environmental risk assessments on asbestos. The predicted relative risk estimates were compared with this study's lung cancer SPMR and SMR. The implications of these comparisons for exposure-effect modeling, risk assessment methodology and public health policy are discussed.

D.2. Methods

D.2.1. Projected Relative Risk of Lung Cancer Mortality

The arithmetic average lifetime asbestos exposure of the female residents of the asbestos-mining agglomerations was used to predict the relative risk of lung cancer in this population[1]. The conventional excess relative risk model used in asbestos risk assessments expresses the rate of lung cancer in a target population as a linear function of that population's cumulative asbestos exposure and background lung cancer rate. As shown in the HEI-AR report [1991, p.6-11] and here below, the equation can be expressed in relative terms independent of absolute lung cancer rates, reducing the unknown parameters to the exposure-effect gradient and the cumulative exposure in the exposed population:

$$\lambda_k = \lambda_0 \bullet \left(1 + K_L . \overline{cE}\right) \text{ or, equivalently,}$$

$$RR_s = {}^{\lambda_k}\!/_{\lambda_0} = 1 + K_L . \overline{cE}$$

where K_L = toxicity gradient, the increase in lung cancer excess relative risk per unit of "cumulative exposure"

λ_k = standardized lung cancer incidence or mortality rate in the exposed population

λ_0 = standardized background lung cancer incidence or mortality rate in an unexposed but otherwise comparable population with similar smoking habits

$\overline{cE}$ = occupationally equivalent (40hrs./wk) mean cumulative exposure $\left(\frac{\text{f-y}}{\text{mL}}\right)$

and RR_s = standardized rate ratios or relative risks (SMR, SRR, SPMR, OR, RR).

[1] Using the arithmetic rather than the geometric average was justified in Section B.4.4.

The present study's arithmetic cumulative exposure estimate and the EPA's mean linear exposure-effect gradient[1] K_L were fed into the above equation to estimate the lung cancer relative risk expected by the EPA model in the present non-occupationally exposed population. "Low" and "high" relative risk estimates were obtained by using in turn the lower and upper plausibility limits of this study's cumulative exposure estimate. To see how different would be risk predictions obtained by assessments using the EPA's methodology with different subsets or appreciations of the occupational data, other environmental risk assessments' point estimates of K_L were also fed into the above equation. For instance, the gradient K_L estimated by the EPA was 1%. This means that each unit of occupationally equivalent[2] cumulative occupational exposure (f-y/mL) would add 1% of the expected incidence of lung cancer to the observed excess incidence. The occupationally equivalent[3] cumulative asbestos exposure (arithmetic mean) of the female residents of the asbestos-mining towns was estimated at 147 f-y/mL with a plausibility interval of 21-525 f-y/mL. Accordingly, the expected RR in the exposed would be 1 + (147 x 1%) = 2.47, with a lower plausibility limit of 1 + (21 x 1%) = 1.21, and an upper plausibility limit of 1 + (525 x 1%) = 6.25. This is how the "best", "low" and "high" estimates were obtained.

The estimation was somewhat more complex for the RR predictions based on the U.S. Consumer Product Safety Commission's (CPSC). This agency produced a 10-fold range of K_L values centered on the overall median rather than a point estimate. For the calculation of the expected RR, this range was considered as the 95%CI of a log-normal distribution. Similarly, the plausibility interval of the present cumulative asbestos exposure estimate was considered as the 95%CI (21-525 f-y/mL) of a geometric distribution with a geometric mean of 105 f-y/mL. For the CPSC's assessment, the expected relative risk of lung cancer was the product of the median K_L by the arithmetic mean of the cumulative exposure variable. The "low" and "high"

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[1] The exposure-effect gradient has been also referred to as "potency" and "toxicity gradient" with respect to experimental data but also with respect to epidemiological data [Hughes and Weill, 1994].

[2] The exposure-effect gradients were estimated from workers exposed some 40 hours per week.

[3] This was estimated in Section B.4.4 as 4.2 times the cumulative exposure (35 f-y/mL) of this continuously exposed population.

estimates were the exponentiated 95% confidence limits of the sum of the logarithmic transformation of the cumulative asbestos exposure and K_L variables.

In addition to the EPA's point estimate of K_L, the EPA's report estimated a 95%CI of 0.004-0.027 around the K_L gradient of 0.01 and mentioned that there should be a geometric 100-fold 95% prediction interval around specific risk projections. Yet, these intervals were barely justified and were only mentioned in a note at the bottom of the table of risk estimates. These intervals were not mentioned in ensuing asbestos policies nor in the HEI-AR and INSERM reviews which used the EPA's K_L point estimate for their own risk projections. Accordingly, RR projections based on this confidence interval were computed separately from those based on the point estimate. The simulation method used for the CPSC's projections was applied here to account for the purported probability distribution of the EPA estimate of K_L.

Two environmental risk assessments on asbestos used the same methodology as the EPA but did not produce a summary point estimate or confidence interval of K_L. The Ontario Royal Commission on Asbestos (ORCA) and the Health and Safety Executive in 1983 (HSE) favored an industry-specific prediction approach. The exposure-effect gradient estimated from McDonald's cohort of Quebec chrysotile miners and millers[1] was used as the ORCA's and HSE's estimate. Similarly, separate risk predictions were made using exposure-effect gradients estimated from individual studies of chrysotile miners and millers to see if restricting the comparison to a chrysotile-mining and milling environment resulted in more accurate RR predictions, as would be expected if the asbestos species or the type of industrial process modified the relation between the risk of lung cancer and cumulative asbestos exposure.

The lung cancer relative risks expected from various K_L estimates and the present exposure assessment were considered as "null hypotheses" to be tested against the excess lung cancer

[1] The ORCA did not conclude on an indiscriminate point estimate of K_L. Rather, it proposed that risk estimates be specific to the type of industrial process and asbestos involved [Royal Commission on Matters of Health and Safety .... 1984, Vol. 1 p.8, §4,5,6, Vol. 2 p.503]. The range of K_L estimates was: 0.0002-0.042. Similarly, the 1983 HSE report by Acheson and Gardner [HSE et al., 1983] concluded on a range of estimates of 0.0004-0.053, the proper estimate being that adapted to specific exposure circumstances. For the present chrysotile mining/milling environment, I supposed that the ORCA and HSE would have relied on the McDonald et al. data on chrysotile miners and millers.

mortality assessed in the exposed population. The two-sided 95% confidence interval of the SPMR was compared in turn with the plausible range of each predicted relative risk. An overlap between a confidence interval and the plausibility range might be interpreted as "compatibility" between our estimate and the hypothesized K_L. However, this overly conservative approach would imply that all possible values within the plausibility intervals of cumulative exposure and of the resulting RR projections were equally likely as in a uniform distribution.

D.2.2. Formal Statistical Comparisons

To account for the uneven probability distribution of the cumulative exposure estimate and of the resulting expected RRs, and to make quantitative contrasts between these expected RRs and the observed SPMR and SMR, the whole statistical distributions of the SPMR, SMR and expected RR were compared directly as follows. Despite their different statistical distributions, both the SPMR and SMR could be approximated with the same log-normal distribution form if sampling error of the standard population were negligible and if the fraction of lung cancers among all deaths were very small [Breslow and Day, 1987]. Since these conditions were fulfilled, the distributions were approximated with a log-normal distribution having the following standard error:

$$s.e.(\ln SPMR) \equiv s.e.(\ln SMR) \equiv \frac{1}{\sqrt{O}}$$

where O represents the total observed deaths from lung cancer

(Breslow and Day , 1987 , vol . II . p . 67 , 77)

Thus, the two approximate log-normal distributions have the same standard error but differ by the location of their log-normal means: ln(SPMR) and ln(SMR). As to the probability

distribution of relative risk projections based on a given hypothesized K_L, the statistical error depended fundamentally on the "plausibility distribution" of the estimated cumulative asbestos exposure. Since this estimation was expressed as a geometric distribution with a 5-fold plausibility range on each side of a geometric mean, and since this plausibility range was intended to err on the side of caution (i.e. to be wider rather than narrower), the cumulative exposure estimate was considered as a geometric distribution with a mean of 105 f-y/mL and a 95%CI of 21-525 f-y/mL, i.e. with a geometric standard error of $5^{1/1.96}$.

The expected and measured relative risks of lung cancer mortality were random variables that did not belong to the same family of statistical distributions[1], and no simple mathematical formula could be used to compute directly the difference or ratio of these random variables. Instead, stochastic simulations of the two purported probability distributions were carried out, and then the two simulated series of estimates were compared statistically. As the SPMR and the SMR were estimated independently from the average cumulative exposure in this study, the observed and predicted relative risks were simulated independently from one another.

A probability distribution was simulated for each estimate by generating 2000 random SPMR and SMR estimates and 2000 random cumulative exposure estimates, following their respective log-normal distributions. Each hypothesized exposure-effect gradient K_L was applied to the 2000 simulated cumulative exposure values to generate a predicted or expected relative risk estimate[2]. The proportion of the 2000 simulations where the "projected" RR was smaller than the simulated SPMR (or SMR) was computed. This proportion was considered as an approximate probability that the predicted RR be smaller than either the SPMR or the SMR due to random exposure measurement and population sampling errors. This "probability" was equivalent to the probability that the exposure-effect gradient estimated in this study (excess SPMR or SMR divided by average cumulative exposure) would be smaller than the

[1] The SPMR and SMR were approximately log-normally distributed whereas the expected RR was not, although the expected excess RR was log-normally distributed like the average cumulative exposure estimate.

[2] As indicated above, the range of gradient K_L estimates produced by the C.P.S.C. was considered as the 95%CI of a log-normal distribution. This distribution of K_L was simulated in 2000 random trials. Each value obtained was then used to multiply a randomly simulated cumulative exposure value to estimate the corresponding expected relative risk. The same algorithm was applied to the EPA's 95%CI.

hypothesized K_L due to random error. It might in fact be considered as a one-sided p-value in those instances where the hypothesized K_L was larger than the exposure-effect gradient estimated in this study. However, a hypothesis testing approach would have been overly simplistic given that 1) the exposure circumstances of this study's exposed population differed materially from that of the population targeted by risk assessments, and 2) the p-values were based on subjective estimations of cumulative exposure and on imperfect discrete simulations. Rather, the computed "probability" was simply an imperfect quantification of the compatibility of this study's asbestos exposure-effect assessment with an exposure-effect gradient K_L suggested by a given risk assessment or occupational study. The higher the probability was, the more similarity there was between this study and the hypothesis; inversely, the smaller the "probability", the more convincing the discrepancy between this study and the hypothesized K_L.

Additionally, a sort of 95%CI of the exposure-effect gradient in the present study was estimated by dividing each of the 2000 simulated SPMRs and SMRs by one of the 2000 simulated cumulative exposure estimates on a one-on-one basis. This simulated 95%CI was compared with the gradients estimated from risk assessments or cohorts of chrysotile miners and millers to evaluate whether the present study was compatible or not with those hypothesized gradients.

D.3. Results

Table D-1 shows lung cancer risk ratios that would be expected if the K_L gradients estimated in selected risk assessments and occupational studies were applied to the estimated cumulative asbestos exposure of the female residents of the asbestos-mining towns. The RRs corresponding to the arithmetic average and plausibility limits of the cumulative exposure estimate are shown for each K_L. The risk assessments and occupational studies are listed separately, by decreasing order of K_L. This study's SPMR, SMR and their 95%CIs are presented at the bottom of the table for comparison.

The EPA's archetypal risk assessment and all risk assessments other than the ORCA's and HSE's would predict much higher lung cancer RRs than measured with the SPMR or the SMR. Thus, based on the SMRs, the EPA model would have predicted 105 excess lung cancer deaths vs. none observed in this study's exposed population; based on SPMRs, an excess of 95 excess lung cancer deaths would have been predicted compared to 6.5 observed. Accounting for the 95%CI of the EPA's estimate did not close-in meaningfully on the observed risk of lung cancer mortality. The NRC would have predicted twice as many excess lung cancer deaths as predicted by the EPA. Except for the ORCA, the "low" estimates of all risk assessments were all above the observed SPMR and SMR, suggesting statistically significant overestimations of risk. Their "high" estimates were more than one order of magnitude above the upper 95%CL of the observed SPMR-based excess (24.5 deaths), predicting between 339 and 1,405 excess lung cancer deaths. Still, the statistical significance of the discrepancies between predicted and observed lung cancer mortality is not clear from Table D-1 since the plausibility intervals of the RR projections overlap somewhat with the 95%CIs of the SPMR and of the SMR.

The ORCA's and HSE's predictions were different from those based on the other risk assessments. Their predictions would likely have relied on the exposure-effect relation estimated in Quebec chrysotile miners and millers which predicted a RR almost identical to the observed SPMR, but higher than the SMR. RRs extrapolated from the two other studies of chrysotile miners and millers were also very close to this study's SPMR, and their plausibility intervals overlapped considerably with the 95%CIs of both the SPMR and SMR. Indeed, the exposure-effect gradients estimated from all three cohorts of chrysotile miners and millers (K_L = 0.0006-0.0017) were similar to this study's SPMR-based estimate (K_L = 0.00068). In particular, the exposure-effect gradient of 0.0006 estimated in chrysotile-miners and millers from the exposed study area[1] was almost identical to that estimated in this study.

Table D-2 presents the simulated "probabilities" that the observed SPMR or SMR be higher than the predicted RRs due to the uncertainty of the present study's exposure and mortality estimates. Except for the ORCA's and HSE's assessments, assuming that the linear exposure-effect gradients represented the true underlying exposure-effect relation in this study's population, there would have been less than a 3.3% probability of observing a lung cancer SPMR as low as or lower than that observed due to chance alone. The equivalent probability would have been less than 1% of observing a SMR as low as or lower than that observed, due to chance alone. By contrast, the present study's results did not differ significantly from the extrapolations based on cohorts of chrysotile miners and millers nor from the ORCA's and HSE's predictions since these were based on the McDonald et al. exposure-effect data.

Finally, the exposure-effect gradient K_L in the present study was estimated at +0.00068 with a 95%CI of -0.0021 to +0.0083 using the SPMR, whereas it would be -0.00003 with a 95%CI of -0.0039 to +0.0041 using the SMR. These simulated "confidence intervals" did not overlap with the values of K_L estimated and used by the U.S. EPA, the U.S. NRC, the U.S. CPSC and

[1] The evidence from Nicholson's study of workers in the same area was not as strong as the McDonald study. The higher gradient estimated by Nicholson could be due to several factors: the restriction of the cohort to the town of Thetford Mines where risks appeared to be higher, a different referent population with lower smoking rates (Canada lung cancer rates), the absence of smoking data, a much smaller sample size (544 vs. 11,000 in the McDonald study), a less exhaustive exposure assessment, and the reduction of internal comparisons to 2 groups of workers (compared with 16 in the McDonald study).

the U.K. HSC(1985), whereas the 95%CIs of both the SMR-based and SPMR-based K_L estimates overlapped with all three K_L gradients estimated from the cohorts of chrysotile miners and millers, which would have been used by the Canadian ORCA and the U.K. HSE(1983). The EPA's and CPSC's 95%CIs of K_L barely overlapped with this study's upper 95%CL.

Table D-1 Lung Cancer Risk Ratios Projected by Various Risk Assessments and Occupational Studies Using the Linear Excess Relative Cancer Risk Model Compared With Observed

Data Source, Reference	Number of Studies by Type of Asbestos Fibre	Estimated Gradient K_L	Risk Ratio Estimate[1] Best	Low	High
Risk Assessments Based on Multiple Fibre Types					
U.S.Environmental Protection Agency (1986), Health Effects Institute - Asbestos Research (1991), Institut National de la Santé et de la Recherche Médicale (1996)	6 "chrysotile", 1 amosite, 4 mixed EPA's 95%CI:	0.01 0.004-0.027	2.47 2.47	1.21 1.16	6.25 7.69
U.S. National Research Council (1984)	5 "chrysotile"*, 1 amosite, 3 mixed	0.02	3.94	1.42	11.50
U.K. Health and Safety Commission (Doll, Peto, 1985)	2 "chrysotile" textile (with some crocidolite)	0.01	2.47	1.21	6.25
U.S. Consumer Product Safety Commission (1983)	6 "chrysotile", 1 amosite, 4 mixed	0.003-0.03 [2]	2.39	1.14	8.21
Ontario Royal Commission on Asbestos (1984) and U.K. Health Safety Executive (Acheson, Gardner, 1983)	4 "chrysotile", 3 mixed; only the estimate from chrysotile-mining was used	0.0006 [3]	1.09	1.02	1.32
Risk Projections Based on Chrysotile-Mining Cohorts					
Nicholson et al. (1979)	1 chrysotile mining-milling (Thetford Mines)	0.0017	1.25	1.04	1.89
McDonald et al. (1980)	1 chrysotile mining-milling (Quebec)	0.0006	1.09	1.02	1.32
Rubino et al. (1979)	1 chrysotile mining-milling (Balangero, Italy)	0.00075	1.11	1.02	1.39
Observed Relative Risks in This Study					
SPMR	1 female residents of chrysotile-mining towns	0.00068	1.10	0.88	1.38
SMR	(idem)	- 0.00003	0.995	0.77	1.26

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1 For risk assessments and occupational studies, the "best" estimate was the cumulative exposure of the study population (147 f-y/mL) multiplied by a given K_L. The "low" and "high" estimates were the lower 21 f-y/mL and upper 525 f-y/mL limits of the exposure plausibility interval stated in the Exposure Assessment multiplied by a given K_L. As to the results of the present study, the "best", "low" and "high" estimates were the point estimate and the 95%CI of the SPMR and SMR.

* Nearly all "chrysotile" cohorts have, in fact, been exposed to significant amounts of crocidolite and/or amosite.

2 The fork of gradient K_L estimates produced by the CPSC was considered as a 95%CI of a log-normal distribution. For this assessment, the "best" estimate of the relative risk was the product of the geometric mean of K_L by the arithmetic mean of the cumulative exposure variable. The "low" and "high" estimates were the exponentiated 95% confidence limits of the sum of the log-normal distributions of *cumulative exposure* and K_L.

3 The ORCA and HSE did not conclude on a single point estimate of K_L but rather concluded that risk estimates must be specific to the type of industrial process involved for lung cancer[McDonald et al., Vol.1, p.8-12], whereas the variety of asbestos determined different risks of mesothelioma. The range of K_L estimates was: 0.02-4.2 %. For a mining and milling environment, the ORCA and probably the HSE would have relied on the McDonald et al. study on chrysotile miners/millers.

Table D-2 Simulated Statistical Comparison of the Observed Lung Cancer SPMR and SMR With the Relative Risks Estimated from Various Asbestos Risk Assessments and Other Risk Projections

Data Source, Reference	Comment	Estimated Gradient K_L	"Probability"[1] that SPMR ≥ RR estimate for a given K_L	"Probability"[1] that SMR ≥ RR estimate for a given K_L
Risk Assessments Based on Multiple Fibre Types				
U.S.Environmental Protection Agency (1986), Health Effects Institute - Asbestos Research (1991), Institut National de la Santé et de la Recherche Médicale (1996)	EPA:	0.01	.017	.008
	EPA's 95%CI:	0.004-0.027	.030	.010
U.S. National Research Council (1984)		0.02	.004	.001
U.K.Health and Safety Commission (Doll, Peto, 1985)		0.01	.017	.008
U.S.Consumer Product Safety Commission (1983)		0.003-0.03	.033	.010
Ontario Royal Commission on Asbestos (1984) and U.K. Health Safety Executive(Acheson, Gardner, 1983)	Estimate based on chrysotile-mining	0.0006	.537	.256
Risk Projections Based on Chrysotile-Mining Cohorts				
Nicholson et al. (1979)	Thetford Mines, Quebec	0.0017	.295	.106
McDonald et al. (1980)	Eastern Townships, Quebec	0.0006	.537	.256
Rubino et al. (1979)	Balangero, Italy	0.00075	.492	.226
Observed Relative Risks in This Study				
SPMR	females of chrysotile-mining towns	0.00068	.500	-----
SMR	(idem)	- 0.00003	-----	.500

[1] Probability of the observed SPMR or SMR being smaller than the projected RR by chance alone for a given K_L. This is akin to an upper-tail P-value when testing a given K_L as a null hypothesis. This "probability" was estimated by comparing 2,000 simulations of the predicted RR with 2,000 independent simulations of the SPMR and SMR.

D.4. Discussion

This study compared the risk of lung cancer observed in a population environmentally exposed to intermediate levels of asbestos fibres with risks predicted from environmental risk assessments using roughly the methodology prescribed by the U.S. EPA. At issue are uncertain assumptions central to risk assessments on asbestos and outlined in Table D-3.

It appears that the excess risk of lung cancer observed in the female population of Quebec's asbestos-mining agglomerations over the 1970-1989 period was considerably and significantly lower than expected from "synthetic"[1] risk assessments that have produced omnibus bottom-line estimates of risks. The EPA's K_L estimate of 0.01 was about 15 times larger than this study's K_L estimate of 0.00068 based on the SPMR. The discrepancy was significant. Indeed, if the EPA's estimate of the exposure-effect gradient and its variability were true, and accounting for this study's mortality sampling error and exposure measurement uncertainty, there would be no more than a 3% probability that this study's SPMR could be as low or lower than observed due to random error alone. Indeed, the considerable overestimation of asbestos-related lung cancers could not be explained by random, nondifferential or even potential differential errors in the present study.

Alternative explanations are sought: 1) non-occupational exposures in chrysotile-mining areas may not be relevant for assessing risks induced by environmental exposures to asbestos in general populations targeted by risk assessments, 2) "synthetic" or uniform risk assessments (ex.: EPA) may be much more imprecise than suggested by their bottom-line estimates and confidence intervals, 3) the likely errors may well be skewed toward lower risks. Finally, this

[1] I have termed "synthetic" the risk assessments that have concluded on a single exposure-effect estimate, be it with a surrounding confidence interval.

study's concordance with cohorts of chrysotile miners/millers and with more "analytic" risk assessments (ex.: ORCA) suggests that risk assessments should be adapted or geared to specific or better characterized environmental exposure circumstances.

D.4.1. External Validity of the Present Study

In terms of average asbestos concentration levels alone, this study (≈0.5 f/mL) is much closer to the general environmental exposures (≈0.0005 f/mL) to which risk assessments are often applied than are occupational data (≈30 f/mL) at the basis of these risk assessments. It is also more relevant in terms of age at first exposure (childhood), duration (lifetime) and time pattern (continuous). Accordingly, the present non-occupational study may be said to be about two orders of magnitude closer to the general population's exposure circumstances than are risk assessments based on occupational data. In addition, the present study was less susceptible than occupational studies to comparison biases such as the healthy worker effect, heavier smoking than in the referent population, etc.

The only potential generalizability or external validity limitation of the present study concerns the kind of asbestos fibres to which residents of these chrysotile-mining towns have been exposed. The carcinogenic potency of the same concentration of airborne asbestos fibres might be lower in chrysotile-mining towns than in general populations' targeted by environmental risk assessments on asbestos. Thus Nicholson [1996] excluded chrysotile miners and millers from the final estimation of the exposure-effect gradient K_L, because respirable fibres in that environment would allegedly be atypical of those to which general populations of would be exposed.

In terms of the chemical characteristics of respirable asbestos fibre aerosols, a lung cancer risk differential by asbestos species has been suggested [McDonald and McDonald, 1986; Hughes, 1991], and would be expected due to differences in biopersistence (argued in Literature

Review, Section A.3.3.b). If such a mineralogical differential were true, it might be argued that the population of the chrysotile-mining areas has been exposed to mineralogically *less* toxic fibres (chrysotile < amosite ≤ tremolite < crocidolite) than general populations targeted by environmental risk assessments. In fact, chrysotile is by far the main asbestos fibre to which general populations in North America are exposed (>98% chrysotile[1]) and to which the chrysotile-mining area's population has been exposed [Sébastien et al., 1986]. Yet, both the study and target populations were exposed to some amphibole exposure. In buildings with highest releases from asbestos-containing materials, there may be up to 5% of amphiboles among all respirable airborne asbestos structures [Corn, 1994]. Similar data are not available for past neighbourhood and household exposure levels in the asbestos-mining regions, but the proportion of amosite, tremolite[2] and crocidolite fibres in the asbestos burden of lung tissues appears to be larger in non-occupationally exposed residents of Quebec's chrysotile-mining areas than in the general population [Case and Sébastien, 1988; 1989]. As in North American general populations, the chrysotile-mining area's population was basically exposed to chrysotile asbestos fibres and was also exposed to small levels of amphiboles albeit in a larger proportion than in the general population. With respect to fibre species, the present study was consistent with the lung burden-based recommendation by Langer and Nolan: "*The assessment of risk to asbestos disease in the general population of the US, exposed to chrysotile, should be based on appropriate chrysotile-exposed cohorts.*" [Langer and Nolan, 1989]

The physical dimensions of respirable fibres are a more established lung cancer risk factor than mineralogical species. Longer fibres and fibres with greater length to diameter aspect ratios are more carcinogenic in the lung [Davis and Jones, 1988; Wagner, 1990]. As in most asbestos industries, asbestos fibres in the mining environment are much longer and have a greater aspect

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[1] According to 8 studies compiled by Nicholson [1989], mass concentrations of asbestos fibres measured by TEM in the USA (7 studies, 233 samples) comprised 2.4% amphibole, 97.6% chrysotile. Lee et al. [Lee et al., 1992; Corn, 1994] have counted 2.5% of amphiboles among asbestos "structures" in U.S. schools, 97.5% chrysotile. According to Corn [1994] and Nicholson [1989], asbestos in outdoor air is practically all chrysotile. Note: The mass proportion of amphiboles was elevated (40%) in Paris (135 samples) [Sébastien et al., 1976; Sébastien et al., 1980a], however half of the samples were from the same highly contaminated building.

[2] Tremolite is a natural contaminant of the Eastern Townships' serpentine ore and has been measured in air samples in a proportion of about 1-2 structures for 100 chrysotile structures [Sébastien et al., 1986].

ratio [Gibbs and Hwang, 1980; Churg, 1986; Churg and Wiggs, 1986; Case and Sébastien, 1989; Lee et al., 1992] than in the general population's environment where more than 99%[1] of asbestos fibres are "short fibres". Due to their physical dimensions, asbestos fibre aerosols in Quebec's asbestos-mining region should be more carcinogenic than in the general environment.

So, in terms of both chemical and physical characteristics, airborne respirable asbestos fibres in this study's exposed population should be no less toxic than in the general population. In fact, the carcinogenicity of asbestos fibres respired by the non-occupationally exposed population of Quebec's chrysotile-mining area would likely be intermediate between fibres respired by the general population and those respired by asbestos workers in the 11 cohorts used in the EPA's risk assessments [Hughes and Weill, 1986; Hughes, 1994]. As to the chemical nature of the fibres, 9 of the studies used in risk assessments involved heavy exposures either to amosite exclusively, or to chrysotile-amosite-crocidolite mixtures (even in "chrysotile textile" industries). As to fibre dimensions, 4 of these cohorts were exposed to the significantly longer chrysotile textile fibres, and 9 were exposed to fibres much longer than those to which the general population is exposed[2]. Thus, contrary to the preconception underlying the cursory exclusion of cohorts of chrysotile miners and millers from the final environmental health risk assessment of the EPA [Nicholson, 1986; HEI-AR, 1991; Dement and Brown, 1994], the present study on lung cancer risk in a population living near chrysotile mines should be more germane to the average risk in general populations in North America[3] exposed to essentially short chrysotile fibres than are risk assessments based on occupational cohorts over-

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[1] Corn [1994] and Lee [1992] reported 99.7% of TEM asbestos structures to be shorter than 5 µm in length in American schools.

[2] Although airborne respirable asbestos fibres in textile industries are longer than in mining and milling industries and have exhibited a strong carcinogenicity, the general population in North America is more likely to be exposed to fibres released from asbestos products frequently used in buildings and from car and truck breaks than to fibres released from asbestos textile products. Asbestos milling produces the whole array of fibre lengths used by other industries since it supplies the fibres to all users.

[3] It has been noted that "European buildings may have more of an amphibole contaminant problem" [Ameille et al., 1994; Churg et al., 1994] and that proportionately more amphiboles have been used and more recently so in the United Kingdom than in North America [Weill and Hughes, 1995]. As noted in the Literature Review, a French study [Sébastien, 1976; 1980] of airborne asbestos in buildings in which half of the samples came from the same building found 40% of the mass of airborne asbestos fibres to consist of amphiboles, to be compared with 2.5% in the U.S.A. [Nicholson, 1989; Lee, 1992].

representing longer chrysotile fibres and amphibole fibres [Langer and Nolan, 1989]. Thus, risk assessments aimed at North American populations should have been able to predict correctly the asbestos-induced lung cancer risk in the present asbestos-mining population.

D.4.2. Imprecision of Point Estimates of Risk

By producing single point estimates of risk which also concur between assessments, risk assessments on asbestos give the impression of being reliable. However, the concordance between assessments is not meaningful since the risk assessments rely on similar methods, assumptions and occupational data, building upon their predecessors. Thus their risk estimates are not independent. More important, risk assessments are replete with *caveats* about crucial assumptions, models, data, analyses and estimates[1], but risk assessors conclude nevertheless on single point estimates of risk without evaluating the global extent and consequences of the uncertainties[2]. Risk assessors justify this "bottom-line" approach as follows:

> " Risk assessment calculations of this sort are "best estimates" in the sense that we have no direct evidence that they are too high or too low. However, no meaningful upper confidence limits can be assigned to them, due to the many uncertainties in the reliability and representativeness of the exposure data, as well as the scientific uncertainties relating to the model itself." [HEI-AR, 1991, p.8-9]

According to this reasoning, there are so many uncertainties that confidence limits would not be "meaningful". The following list of major sources of uncertainty in risk assessments (Table D-3) is impressive and difficult to quantify indeed. But then a measure of central tendency such as a point estimate of risk cannot be much more "meaningful" - i.e. realistic or reliable - than its confidence or plausibility interval. Ironically, not bounding the risk estimate

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[1] Asbestos risk assessments are quite consistent in their listing and discussion of their uncertainties and weaknesses [v.g. Nicholson, 1986, pp.171-7; HEI-AR, 1991, pp.6-12, 6-13 and 6-31 to 6-35].

[2] For instance, a footnote to the EPA's final risk estimates [Nicholson, 1986, Tables 6-1, 6-2 and 6-3] mentions that a 100-fold 95%CI should be applied to undocumented exposure circumstances. The implications were neither discussed nor quantified. Policy making and mortality projections have been based on the bottom-line estimates or on alleged upper bounds of these estimates.

because of uncertainty leaves those who use the results of these risk assessments with a single point estimate which conveys the impression of precision and reliability.

Only in the EPA's risk assessment [Nicholson, 1986] was there a formal attempt to estimate a confidence interval around the point estimate of K_L. However, the interval was too narrow to account for the low excess risk observed in the present study. In fact, the major uncertainties listed in Table D-3 were not accounted for (items #2, #4, #5, #7, #8) in the EPA's 95%CI. These uncertainties might multiply by one or two orders of magnitude the EPA's estimation error of the exposure-effect relationship. For instance, the risk ratio between a linear model and a sublinear relationship can be five orders of magnitude at low doses (Section A.3.3.b). Lung cancer risk estimates based on industry-specific or species-specific risks can differ up to 200-fold [Nicholson, 1986, p.81]. Other uncertainties listed in the table might not be as consequential but they add up. The global uncertainty cannot be estimated adequately, but it might well account for the discrepancy between the present study and risk assessments.

Risk assessors should not be governed by the need to make a "bottom-line" risk estimate, overstating the reliability of the assessment [Hattis and Kennedy, 1986; Cohrssen and Covello, 1989]. Instead, they could factor in the uncertainties and present a distribution of final estimates [Roberts, 1990]. Thus, a range of numbers could be provided for each assumption and be fed into a simulation program to generate a risk distribution using Monte Carlo techniques [Finkel, 1990]. At the very least, there should be a "most likely" scenario, along with a "worst-case" and a "best-case" scenarios to bound the limits of uncertainty.

Table D-3 Uncertain Assumptions Underlying the Exposure-Effect Gradients Estimated by Risk Assessments on Asbestos and Lung Cancer

1. Linearity of the exposure-effect relationship across the 10^6 range between past occupational and present environmental exposures
2. a) Equal potency of different asbestos species and industrial processes, attributing the 200-fold industry-specific disparity of risk gradients to random errors, or
   b) Representativeness of general environmental exposures by the occupational cohorts used by risk assessments
3. Reliability of past exposure data in the occupational cohorts used by the risk assessments
4. Definition of etiologic "asbestos fibres" as those longer than 5 µm
5. Validity of the cumulative exposure metameter for dose and the implicit independence of risk with respect to time since exposure
6. Validity of mortality comparisons and data analyses in studies of asbestos workers and validity of their re-analyses by risk assessors
7. Equivalence of fibres counted with optical microscopy and with electronic microscopy when making risk projections from historical occupational cohorts to present general populations[1]
8. Reliability of the exposure assessment and exposure characterization of the general population
9. Independence of relative risk with respect to age at first exposure
10. Equal susceptibility of females and males

[1] Risk assessments are based on occupational studies with exposure measurements in fibres longer than 5 µm seen by optical microscopy. In the HEI-AR and INSERM assessments, risk projections to general populations were based on fibres longer than 5 µm seen by electronic microscopy which counts 1.1-10 times more fibres than does optical microscopy.

D.4.3. Accumulation of Safety Biases Throughout Risk Assessments

Contrary to the HEI-AR's declared equal likelihood of the true estimate being higher or lower than the "best estimate", there are many conservative "biases" in risk assessments on asbestos and lung cancer suggesting that asbestos risk assessments on lung cancer are bound to overestimate risk.

Table D-4 lists the conservative biases discussed in the Literature Review. The table also includes rough overestimation factors to illustrate the relative importance of each potential bias. Notwithstanding the linearity assumption, the cumulation of the other safety factors or biases suggest an overestimation of risk by at least one order of magnitude. By itself, using a linear model might overestimate the risk by orders of magnitude. Using the cumulative exposure metric was deemed to bear opposing biases which could cancel out one another.

Most conservative biases or factors are deliberate and legitimate. At each step of the risk assessment, risk assessors are faced with imprecise data of varying quality; they must select the most pertinent data, make crucial assumptions, transpose or correct obsolete or indirect historical data into etiologically meaningful and updated data, correct for missing or biased data, choose the most adequate analytic methods, etc. In making these decisions, risk assessors have a responsibility toward the public's health and may even be subject to potential litigation. Risk assessors prefer to err on the side of overestimation of risk when in doubt, an inclination propounded by the EPA's risk assessment guidelines [Federal Register # 51 FR 33992-34054, Sept. 24, 1986]. However, when safety factors are introduced at each step of the process, they propagate throughout the assessment akin to the propagation of uncertainties throughout risk assessments [Cohrssen and Covello, 1989, p.94]. As a result, the final exposure-effect estimate is probably not a "best estimate" but rather a "conservative estimate"[1].

[1] Even if it is based on the mean or median of cohort-specific estimates, the selection of the cohorts, their weights in the pooled estimate, the linear model and the data analyses are all conservatively biased.

Table D-4 Safety Factors in Environmental Asbestos Risk Assessments, Particularly in the EPA, HEI-AR and INSERM Assessments[1]

1. Using a linear rather than a sigmoid exposure-effect model may overestimate [Vacek and McDonald, 1991] environmental risk by up to 5 orders of magnitude [Brown and Mantel, 1978].

2. Excluding cohorts of chrysotile miners and millers has inflated the mean estimate of K_L by a factor of 1.5 [Nicholson, 1986].

3. Overrepresentation, with respect to general environmental exposures, of longer fibres, and of amosite and crocidolite asbestos in the occupational cohorts used by risk assessments could overestimate[2] environmental lung cancer risks by a factor of up to 4.

4. The use of geometric rather than arithmetic means in estimating occupational exposure levels would have inflated K_L by a factor of 1.27 if the geometric standard error were 2.

5. Underestimation of pre-1965 exposures of asbestos workers might inflate the exposure-effect gradient by a factor of up to 1.5.

6. A tendency to select higher K_L estimates among the various regressions[3] applied by Nicholson [1986] to each occupational dataset might have inflated the exposure-effect relation by a factor of up to 2 according to my own rough estimate.

7. The cumulative exposure metric would, according to biokinetics, overestimate risk by giving too much weight to recent lower exposure levels, but would also, on the other hand, underestimate risk as a result of random exposure misclassification; these opposing effects could offset each other.

8. Projection[4] of exposure-specific risks estimated from PCOM fibre counts (Ø>0.25µm, L>5µm) in occupational studies onto environmental EM measurements (Ø >0.01µm, L>5µm) tend to overestimate[5] environmental risks by a factor of 1.1-10.

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[1] These assessments are singled out because they are the most recent and complete. Moreover, they are central in the scientific and political debate on environmental asbestos.

[2] My own estimate was based on weighing chrysotile textile by 4%, friction products by 40%, mining and milling by 40%, amosite manufacturing by 1%, insulation and other mixed fibre products industries by 10%.

[3] Nicholson used inverse variance-weighted and unweighted linear regression, with and without forced-zero-intercept, with and without relative-slope adjustment (dividing the slope by the zero-intercept SMR), averaging each cohort as a single-point, adjusting SMRs to local rates when available and when State or national rates seemed inappropriate, etc. Ten times out of 14 Nicholson chose a higher estimate than would have produced the simple weighted linear regression (twice these simple slope estimates were negative to begin with).

[4] The HEI-AR and INSERM risk assessments simply applied the K_L estimated from occupational data onto environmental exposure measurements made by electron microscopy.

[5] These factors were estimated for occupational exposures and fibres longer than 5 µm [Hwang and Wang, 1983; Berman and Chatfield, 1989; Rogers, 1990; Marconi et al., 1983; Dement and Wallingford, 1990]. The EM:PCM factor of 60:1 used by the NRC [1984] and ATSDR [1993] pertained to EM fibres of all lengths.

The results of the present study and the list of estimated "safety factors" in Table D-4 may be used to suggest by how much risk assessments tend to overestimate the risk of lung cancer in general environmental exposure circumstances. The factors listed in Table D-4 tend to overestimate the risk of lung cancer in North American general populations and, except for item #8, might account for the 15-fold overestimation of risk observed in the present study on the basis of the SPMR[1]. If the 15-fold overshoot were indeed attributable to items #1 through #7 and if the present study were representative of the general population's exposures, it would imply that risk assessments overestimate the risk of asbestos-induced lung cancer in general populations by a factor between 16 to 150 (applying item #8 to the 15-fold overshoot).

D.4.4. This Study's Concordance With Cohorts of Chrysotile Miners and Millers and With "Analytic" Risk Assessments

Although the EPA's risk assessment greatly overestimated the risk observed in this study, the lung cancer risk estimates extrapolated from male chrysotile miners and millers agreed very closely with this study's results. However, risks estimated (K_L=0.01) from Australian miners and millers exposed to crocidolite [Armstrong et al., 1988; de Klerk et al., 1989] would greatly overestimate the risk observed in the present study.

Based on the observed SPMR, the evidence is compatible *prima facie* with three assumptions of risk assessments:

1) the linear exposure-effect model, since the same linear exposure-effect gradient was estimated at different asbestos exposure levels (neighbourhood and household vs. occupational) for the same type of asbestos fibres (chrysotile-mining and milling);

2) the cumulative exposure metameter, since the same exposure-effect gradient was estimated in a continuously and an intermittently (workers) exposed populations;

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[1] According to the SPMR, the excess RR was estimated at +10%, whereas the EPA model would predict an excess of +147%, about 15 times more than observed.

3) the equal susceptibility of men and women to asbestos-induced lung cancer, since the observed risk in females was compatible with that extrapolated downward from a cohort of male miners and millers.

However, the evidence challenges two other assumptions which underlie risk assessments on asbestos and lung cancer which have produced point risk estimates:

1) equal potency of fibres associated with different *industrial processes*;
   - the risk of lung cancer in this non-occupational chrysotile-mining and milling environment was consistent with risks measured in cohorts of chrysotile miners and millers but not with those measured in chrysotile-textile workers;

2) equal potency of different *mineralogical asbestos species*;
   - the risk of lung cancer in this non-occupational chrysotile-mining and milling environment was consistent with risks estimated from chrysotile miners and millers but not with those estimated from crocidolite miners and millers.

These explanations are mere suggestions since exposure-effect estimates in the cohort of workers and in the present study have a large statistical variation, and since this study's estimate is compatible with an absence of effect. Moreover, if the SMR were preferred to the SPMR, the asbestos-attributable risk of lung cancer would be overestimated - although not significantly - by the projections from chrysotile miners and millers of the same area. This would be more compatible with a sublinear model than with a linear model. Therefore, no form of exposure-effect relation can be excluded from the comparison of this study's non-occupationally exposed population with the workers of the same region.

D.4.5. Conclusion

Based on this study's exposure assessment, the EPA's environmental risk assessment on asbestos and lung cancer [Nicholson, 1986] and similar risk assessments which have produced

point estimates of risk [HEI-AR, INSERM, US-NRC, US-CPSC, UK-HSC] would significantly and greatly overestimate the observed risk for lung cancer in the female population of Quebec's chrysotile-mining agglomerations. By contrast, lung cancer risks extrapolated from studies of chrysotile miners and millers were in line with the risk observed in this study's non-occupationally exposed population.

Due to statistical uncertainties and other insufficiencies of the evidence, no single explanation of the overestimation of risk by synthetic risk assessments using the EPA methodology can be inferred from the data. Still, together, the various risk comparisons suggest that risk assessments on asbestos are wrong in summarizing heterogeneous occupational exposure-effect data with a single point estimate; instead, it seems that asbestos risk estimations and predictions should be industry-specific and mineralogy-specific.

Assuming that the types of asbestos fibres to which the present population was exposed in the past were no less toxic than those to which general environmentally exposed populations are exposed today, the present study suggests a) that environmental risk assessments on asbestos significantly overestimate the risk of lung cancer in non-occupationally exposed populations and b) that no single exposure-effect estimate can be used to characterize asbestos exposure circumstances which differ greatly in terms of industrial and mineralogical types of asbestos fibres.

PART E. CONCLUSION

Regarding risk assessments on environmental asbestos exposure, risk assessors and various expert groups have been unanimous in deploring the absence of non-occupational risk datasets with quantification of exposure. The present research represented an attempt to answer this need and an attempt to validate existing risk assessments on asbestos and lung cancer.

The average cumulative continuous exposure of the female population of Quebec's chrysotile-mining agglomerations was estimated at 35 f-y/mL, equivalent to an occupational exposure of 147 f-y/mL. This was about 3 orders of magnitude more than the lifetime asbestos exposure of general populations in North America today and one order of magnitude more than that of today's asbestos workers.

Over the 1970-1989 period, the mortality of this non-occupationally exposed population was compared to that of socio-economically similar agglomerations of the Province of Quebec. Despite their elevated asbestos exposures, the female residents of chrysotile-mining agglomerations did not experience a significant excess risk of lung cancer. The lung cancer SPMR was 1.10 with a 95%CI of 0.88-1.38, whereas the SMR was 0.99 with a 95%CI of 0.77-1.26.

Based on the average cumulative exposure of the female population of the chrysotile-mining agglomerations, the EPA's risk assessment on asbestos would predict a relative risk of 2.47 with a plausible interval of 1.16 to 7.69. These predictions were significantly higher than the observed SPMR and SMR. On the contrary, the lung cancer relative risk projected from a large cohort of chrysotile miners/millers from the same area was very close to the observed SPMR and SMR.

The discrepancy between the observed risk and that predicted by the EPA model suggests that point risk estimates produced by asbestos risk assessments using the EPA methodology are not universally valid and, in particular, that they overestimate the risk of lung cancer in populations exposed to chrysotile asbestos in the neighbourhoods of mines and mills. This overestimation is consistent with the accumulation of safety factors throughout the various steps of risk

assessments on asbestos and lung cancer which tend to produce conservative risk estimates. Furthermore, since there is evidence that the study's exposed population was exposed to asbestos fibres at least as carcinogenic as those to which general populations are exposed today, this study's results imply that conventional risk assessments on asbestos considerably overestimate the risk of lung cancer associated with environmental exposure to asbestos in the general population.

It is not clear which of the many assumptions of the EPA's risk assessment lead to the overestimated of risk of lung cancer in this study. However, the present study suggests that lung cancer risk estimations should at least take into account the mineralogical and physical characteristics of asbestos fibres to which the target population is exposed.

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et nombreuses révisions au cours de la dernière année m'ont beaucoup appris et m'ont mis sur la voie d'une rigueur intellectuelle que j'espère approcher un jour.

Je remercie Mme Bette Meek, toxicologue à l'emploi de Santé Canada pour son soutien indéfectible à cette recherche qui lui tenait à coeur, malgré les difficultés encourues. Je remercie également Santé Canada d'avoir financé cette recherche sous l'instigation de Mme Meek.

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Enfin, je ne voudrais pas oublier des gens dans le département d'Épidémiologie et de biostatistique de l'Université McGill qui m'ont soutenu et m'ont évité maints ennuis administratifs: les docteurs Christina Wolfson, Theresa Gyorkos, et John Bailar, ainsi que Mme Marlene Abrams. Pour la même raison, je remercie Mme Ginette Boulanger, du registrariat de l'Institut Armand-Frappier. Je remercie aussi l'Institut Armand-Frappier qui m'a

accueilli parmi ses "étudiants doctoraux libres en recherche" et m'a ainsi permis de travailler dans l'équipe du Dr Jack Siemiatycki.

Je remercie les auteurs du rapport de l'INSERM en France et ceux du Programme international sur la sécurité chimique (IPCS) de l'Organisation mondiale de la santé (W.H.O.) qui ont valorisé la présente étude et son originalité en la qualifiant de pièce-clé attendue dans l'évaluation des risques posés par les expositions environnementales à l'amiante. Merci aussi aux politiciens français qui ont pris la décision logique de bannir l'amiante-ciment pour éviter que les travailleurs du bâtiment ne continuent d'être menacés par des fibres isolantes floquées il y a plus de 20 ans. Ce bannissement a créé une ambiance qui a publicisé et mis à l'ordre du jour les résultats de la présente étude.

Enfin, je crois que la société ne doit rien à un "docteur" et qu'au contraire un "docteur" doit beaucoup à la société. Je m'en souviendrai et je tâcherai humblement d'assumer cette dette et d'être utile à la société. Je chercherai à sortir des tours d'ivoire universitaires et à éviter l'ambition individualiste, jalouse et schyzoïde des "chercheurs du temps perdu".

" Here comes the sun, and I say :
" It's alright"

(G. Harrison)

STATEMENT OF ORIGINALITY

Under Dr. Siemiatycki's supervision and in cooperation with him, I wrote all the protocols to obtain authorizations and funding for the different study components, and I directed or conducted every aspect of the study, data collection and data analyses. The literature review and analysis are also my own, although I used and completed past reviews which I acknowledged at the beginning of my review.

Dr. Jack Siemiatycki had the original idea to compare the estimated excess cancer mortality in the non-occupationally exposed female residents of Quebec's asbestos-mining towns with predictions based on the EPA's risk assessment model for asbestos. Dr. Siemiatycki thought of addressing this question with an ecological epidemiological study and obtained initial funding from Health Canada for me to conduct a feasibility study and develop a detailed research protocol in 1988.

Under Dr. Siemiatycki's supervision, I developed the research protocols for a mortality study, a mesothelioma incidence study and particularly a historical exposure assessment. Health Canada had the protocols reviewed by a panel of experts in the epidemiology of asbestos-related diseases and in the measurement of asbestos exposures. It was approved integrally except for a few minor cuts. In 1989, Dr. Siemiatycki and myself wrote grant applications and obtained funding from Health Canada and from the NHRDP. For practicality, it was decided to restrict my thesis project to the mortality study and the exposure assessment.

Regarding the mortality study, I determined how far back Quebec mortality records were available with identifiable municipality geocodes or names so as to obtain the largest statistical power possible. I determined the statistical unit of analysis (urban agglomerations defined by a governmental agency). I identified datasets on potential confounders available by

agglomeration and completed them by computing additional census-based variables by agglomeration. Faced with inadequacies of the mortality data supplied by the Quebec Statistics Bureau, I recoded some municipal codes of the female death certificate data for the Province of Quebec so as to match them over time to those of the Canadian population Censuses.

Regarding the representative population survey, I conceived the survey, sampling strategy and questionnaire which bore on 1) residential, household and occupational exposure histories, 2) chronological visible exposure recalls, and 3) health and socio-demographic descriptives to be compared with the 1987 Santé-Québec Survey. I supervised the data collection and data entry. I also determined how to weigh the results of the 1989 cross-sectional survey to represent the dynamic 1970-1989 study base. I designed and supervised the data analysis.

Regarding the exposure assessment, Dr. Siemiatycki wanted to conduct a residential history survey to estimate the cumulative exposure of the study population, but had not determined what exposure data was available nor how to come up with a town-year specific concentration matrix. For this purpose, I explored qualitative or quantitative sources of information on past asbestos exposures and designed multiple schemes and sub-studies to come up with various estimates and information. I used secondary datasets for annual asbestos production levels, ambient air and asbestos emission measurements (Asbestos Institute, Environment Quebec and Environment Canada) and lung burden measurements (Drs. B. Case, P. Sébastien and J.C. McDonald); however, I designed and conducted my own analyses of these datasets. I conceived that only a panel of experts could synthesize and weigh heterogeneous qualitative and quantitative data into a final subjective estimate of historical exposure levels. However, the final workings of the panel were developed in cooperation with Dr. Siemiatycki, who also helped me significantly in preparing a digest of the data to be presented to the panel.

Of course, I wrote the entire thesis myself. I will also be the first author of every article that will issue from the present mortality study and exposure assessment.

STATEMENT OF ETHICS

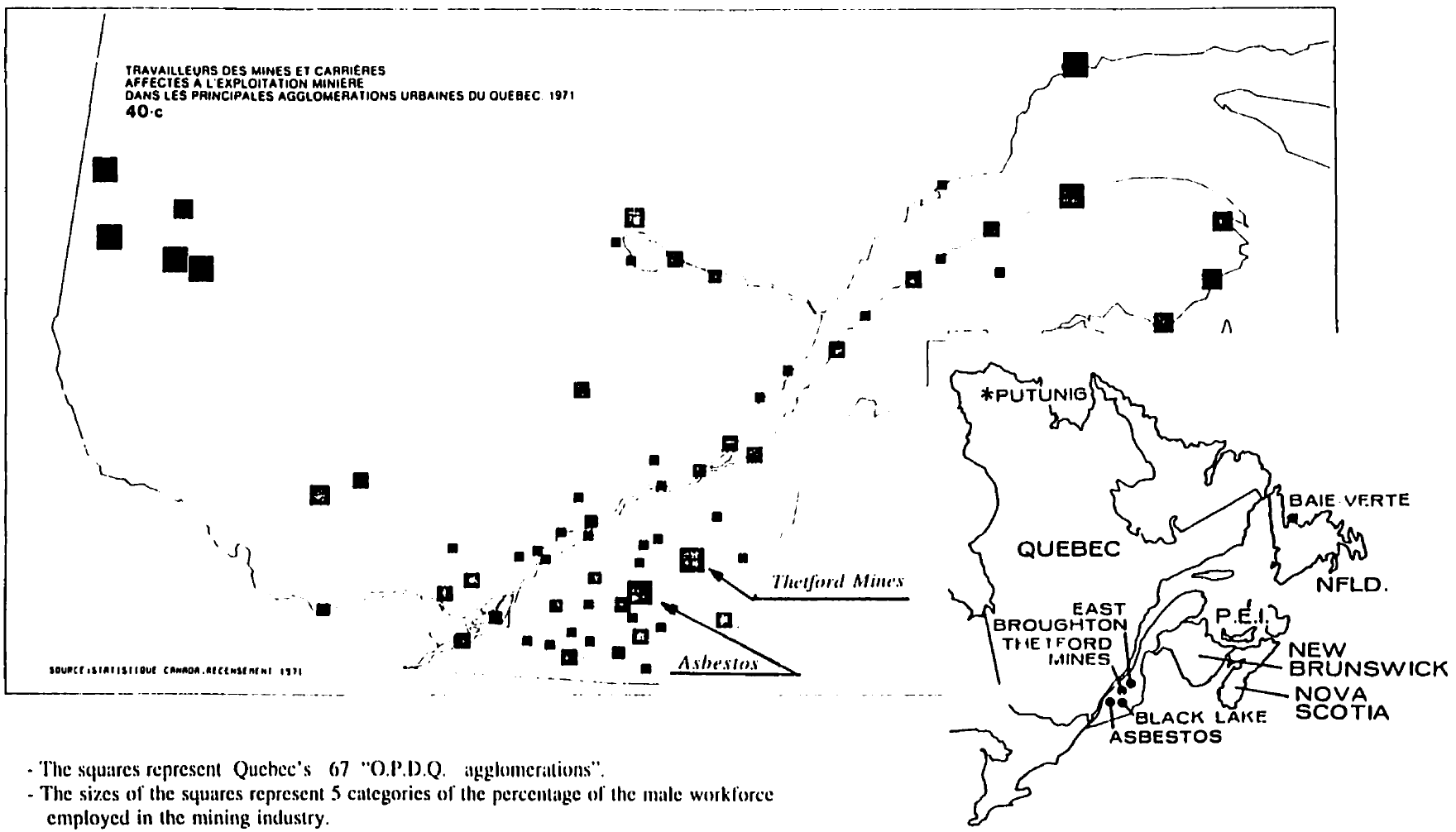
Most of the data has been collected from governmental records and computerized data bases. The mortality study and exposure assessment involved no hospital data. The death certificate data provided to us by the Bureau de la statistique du Québec were "denominalized", i.e. stripped from any potential identifier (name, social and health insurance numbers, personal address, etc.). In only one subproject were subjects contacted: the survey of women age 50 and over in the asbestos-mining regions. Part of the data were obtained by a self-administered questionnaire, and part by personal interviews with a female nurse with years of experience in such field work. The interviews focused on the subjects' residential, household and occupational exposure histories; although a few questions were asked about the subject's health status, there was no discussion of any disease history. All questionnaires have been be kept confidential and secure. All computerized data were denominalized.

The protocol was reviewed and approved by the Institut Armand-Frappier's Ethics Committee as well as by the Commission d'Accès aux renseignements nominatifs du Québec.

APPENDICES

Appendix B1	Map of Quebec's Agglomerations and Asbestos Mining Areas

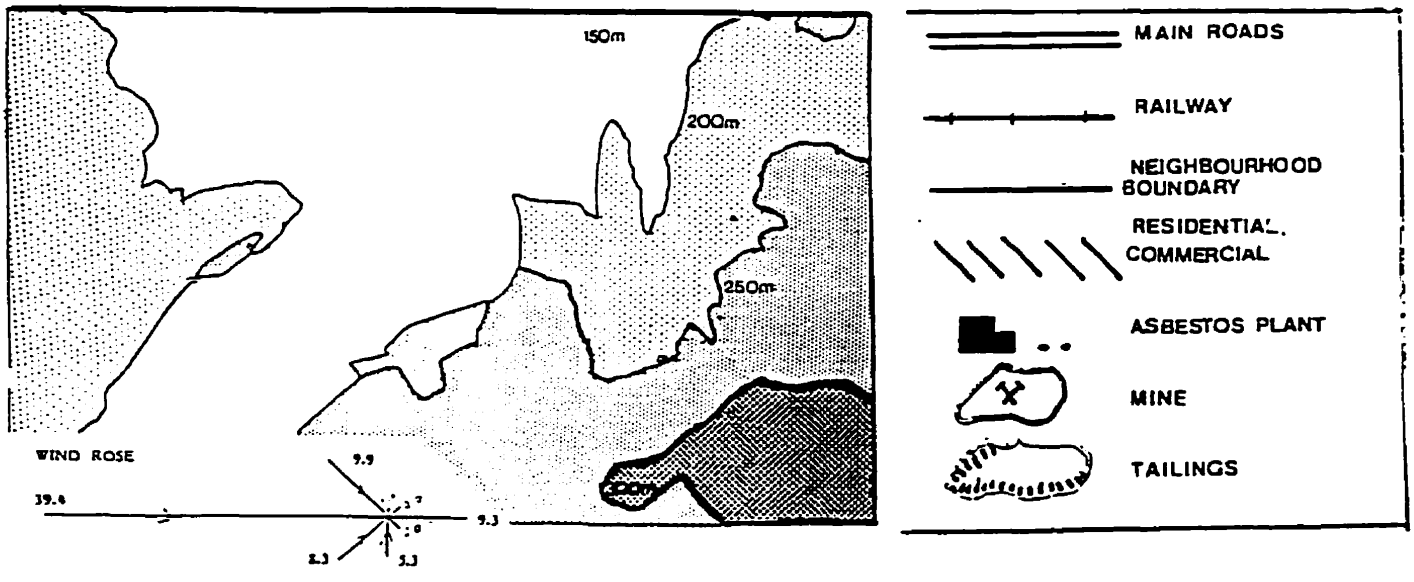
Figure B1-1 Map of Quebec's Agglomerations, and Asbestos Mining Areas



- The squares represent Quebec's 67 "O.P.D.Q. agglomerations".
- The sizes of the squares represent 5 categories of the percentage of the male workforce employed in the mining industry.

Appendix B2	Historical Maps of the Asbestos Mining Towns

The following maps represent the town of Asbestos in 1925, 1934, 1953, 1971 and 1985, and the towns of Black Lake and Thetford Mines in 1944, 1954, 1971 and 1983. The maps show relief, wind rose, mining pits, tailings piles, residential and commercial areas, and neighbourhoods. The latter were identified in panel interviews of local residents and in our residential history survey. Geographical and topographic maps were obtained from various archives in university and governmental libraries and archival depots, for roughly each 20-year period since 1900 to outline trends and patterns in the relation between residential areas and emission sources. The maps found had different scales, orientations and graphical informations. For comparability and use in the present exposure assessment, they were abstracted schematically in a standardized format.



The above legend on the left defines the different shades used to represent different altitude levels indicated in the following relief maps. The legend on the right defines the different symbols used in the maps. Wind roses are also shown on the maps to suggest the directional dispersion of asbestos emission fallouts.

Figure B2-1 Schematic Map of the Town of Asbestos in 1925

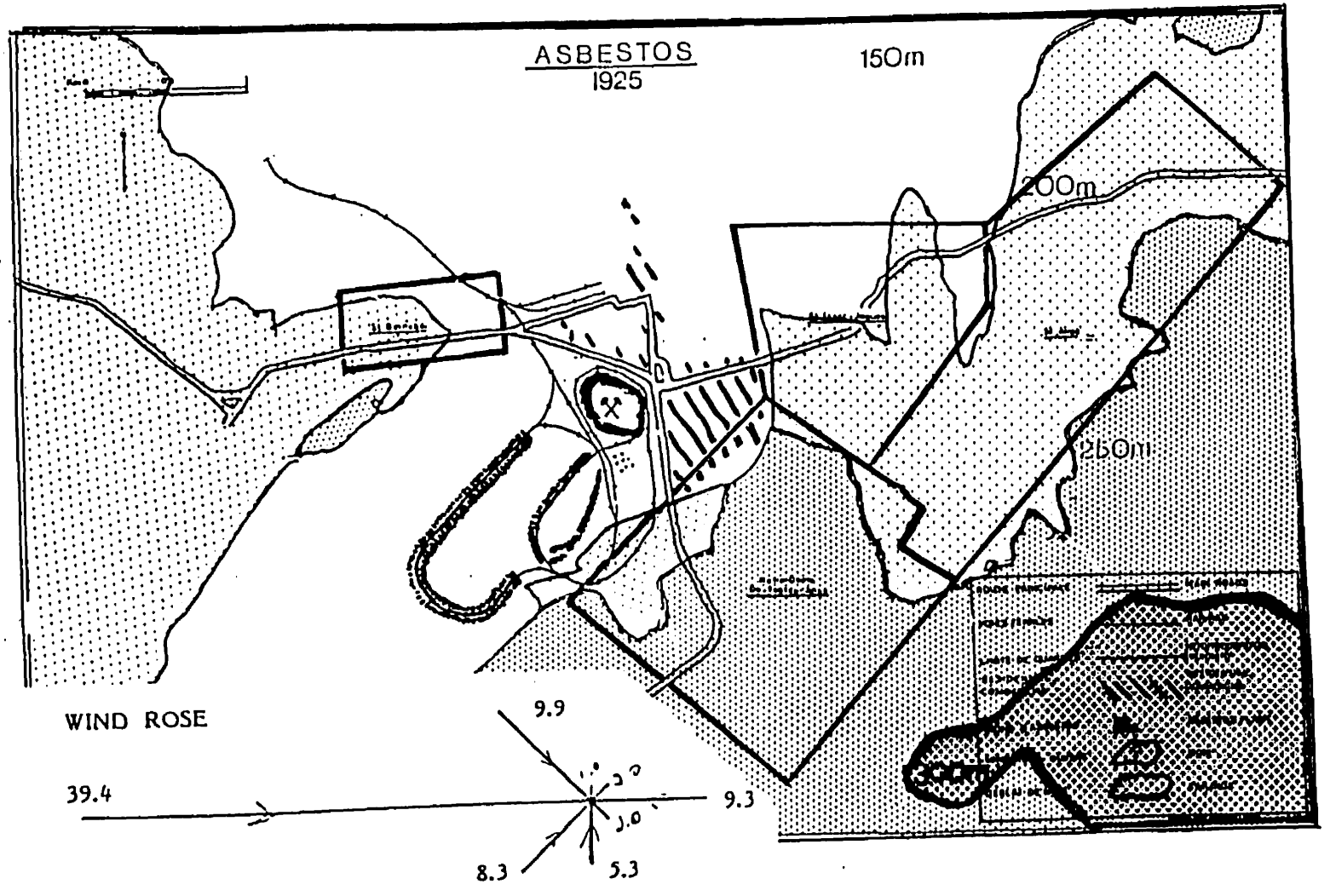


Figure B2-2 Schematic Map of the Town of Asbestos in 1934

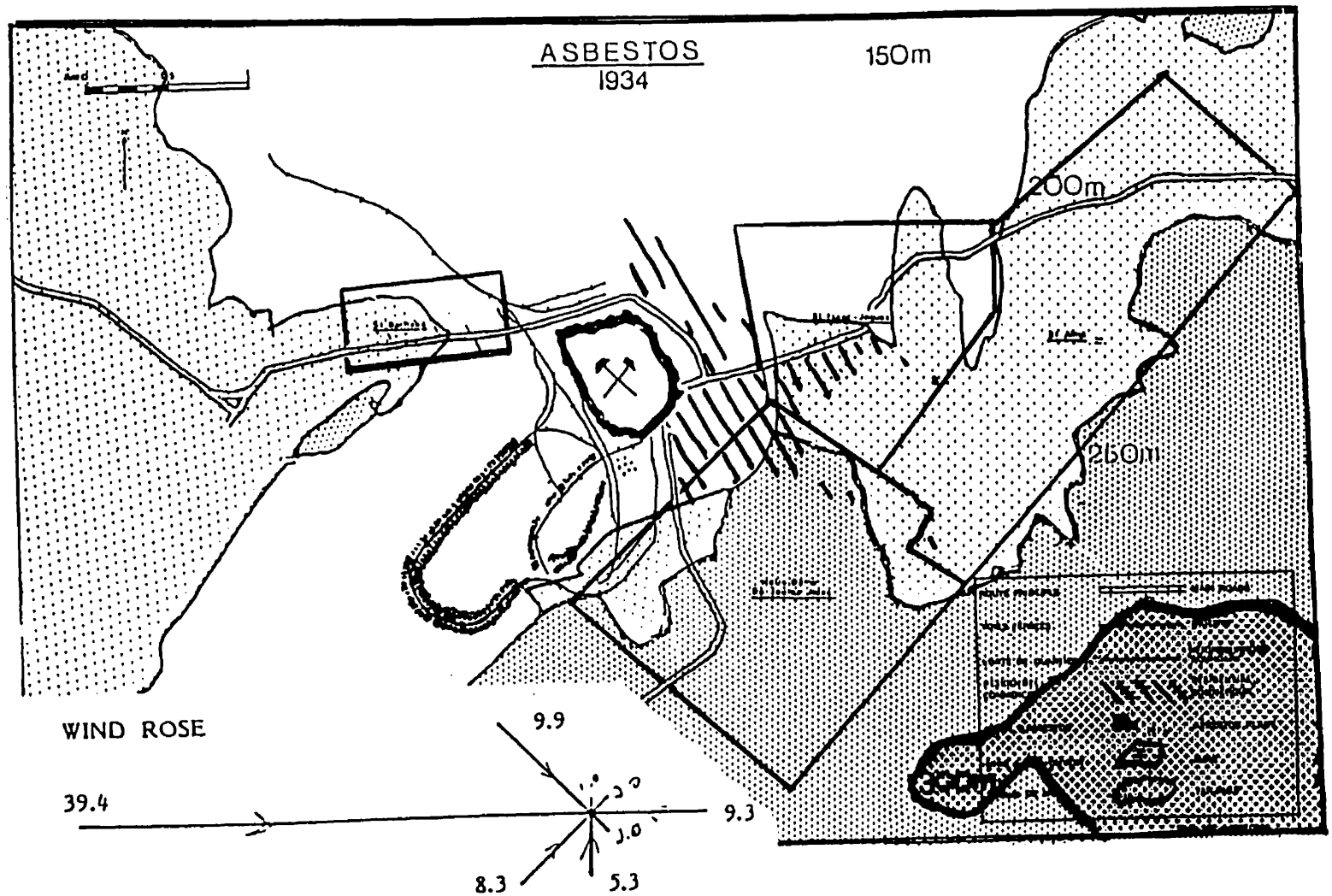


Figure B2-3 Schematic Map of the Town of Asbestos in 1953

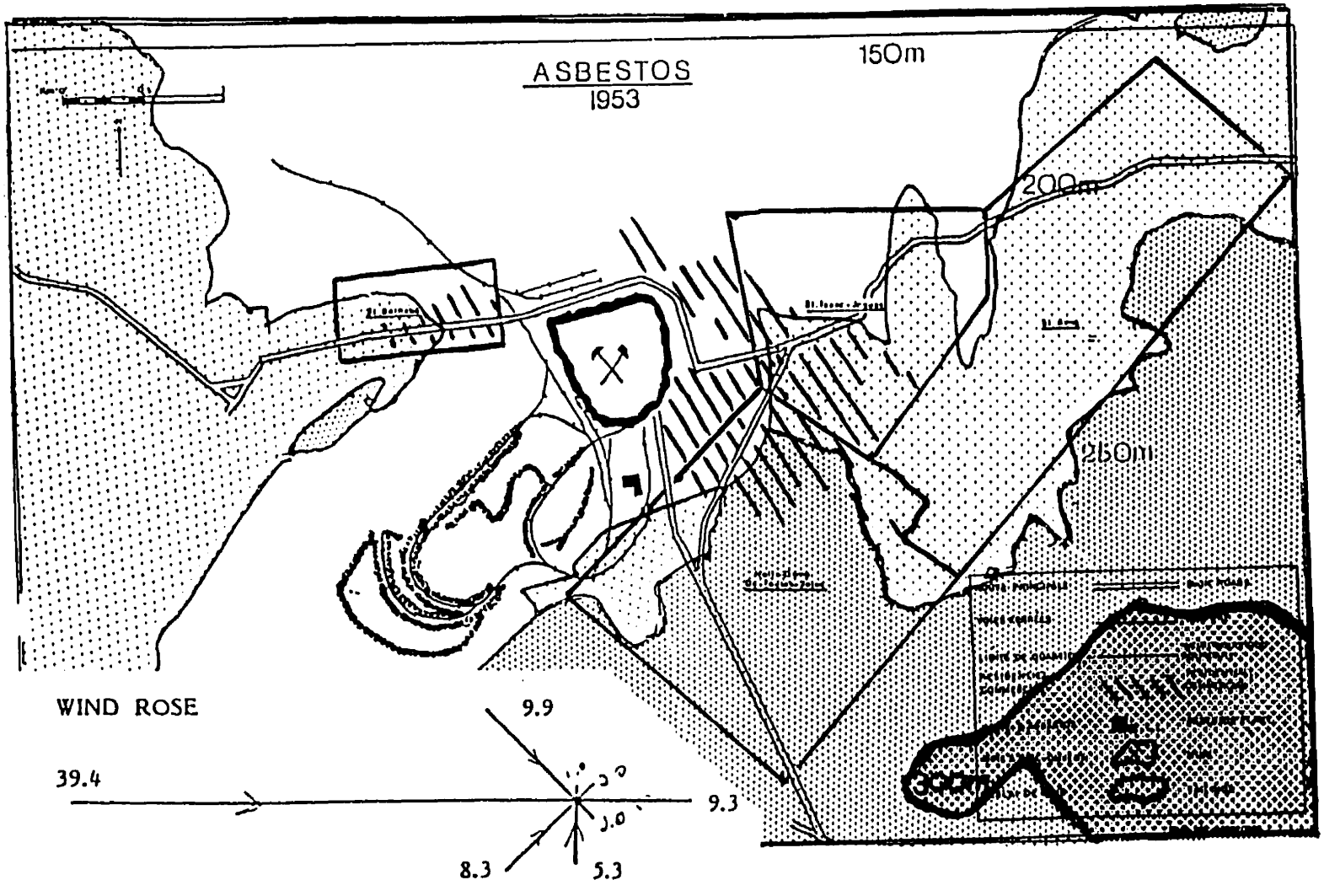


Figure B2-4 Schematic Map of the Town of Asbestos in 1971

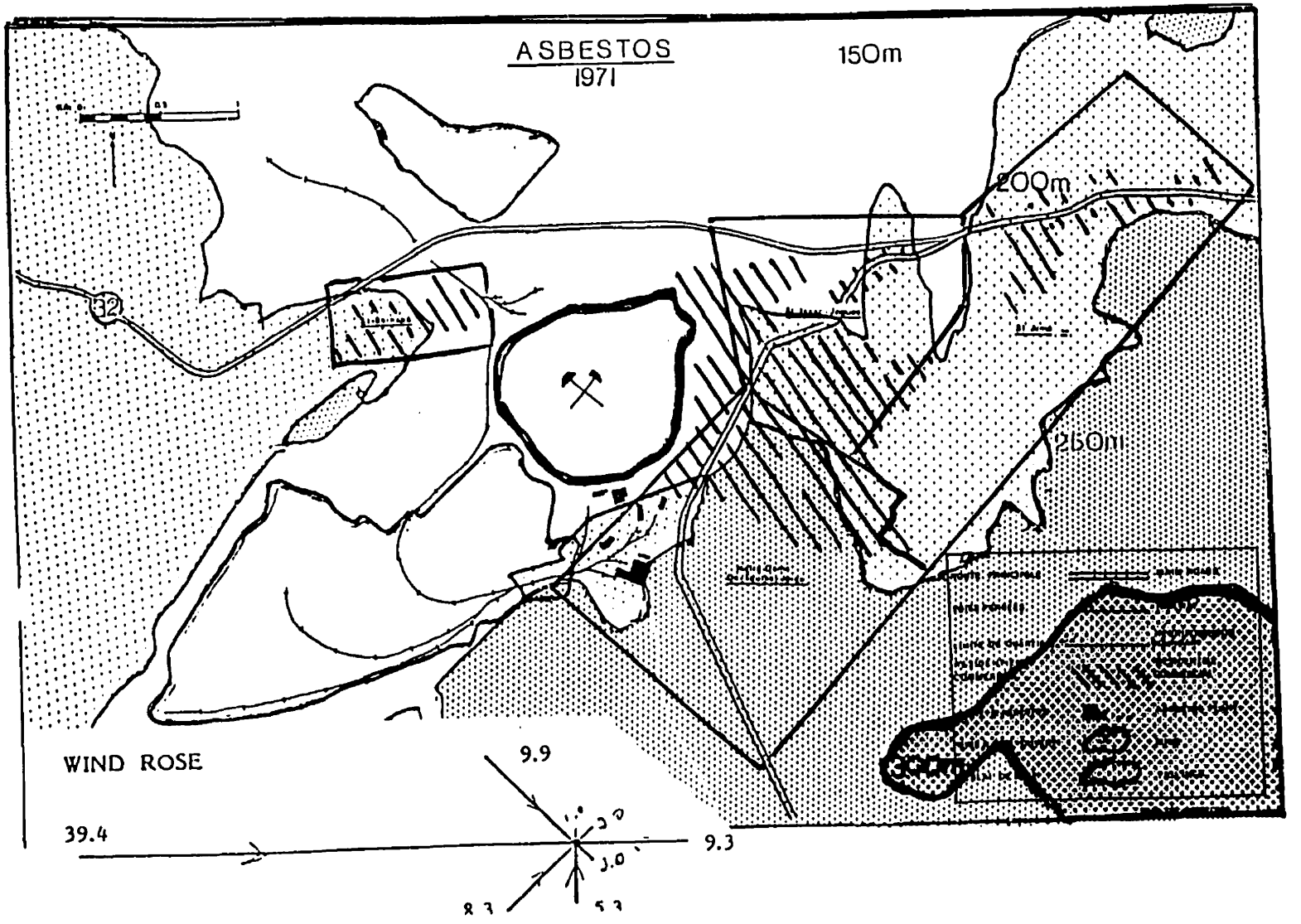


Figure B2-5 Schematic Map of the Town of Asbestos in 1985

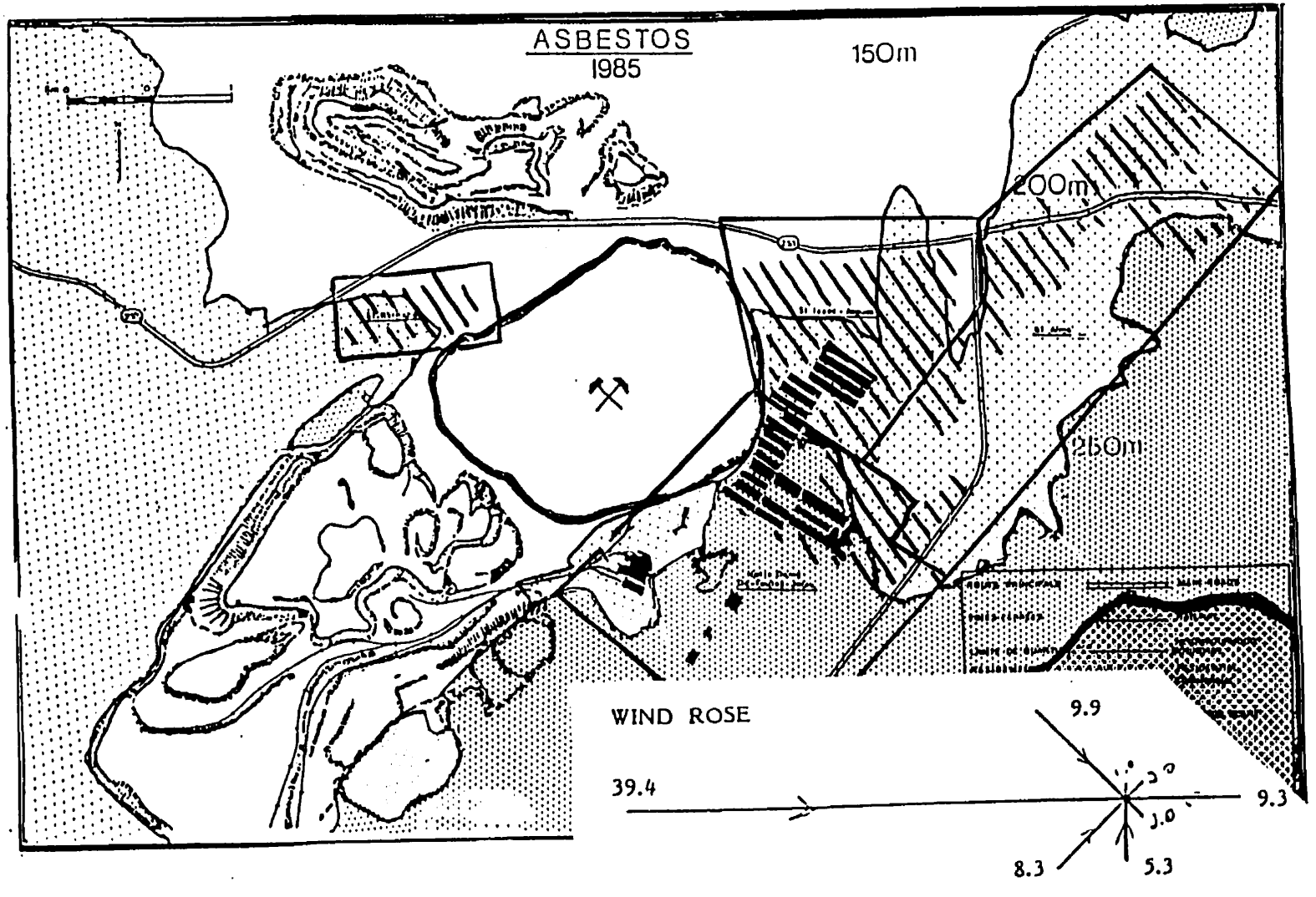


Figure B2-6 Schematic Map of Thetford Mines and Black Lake in 1944

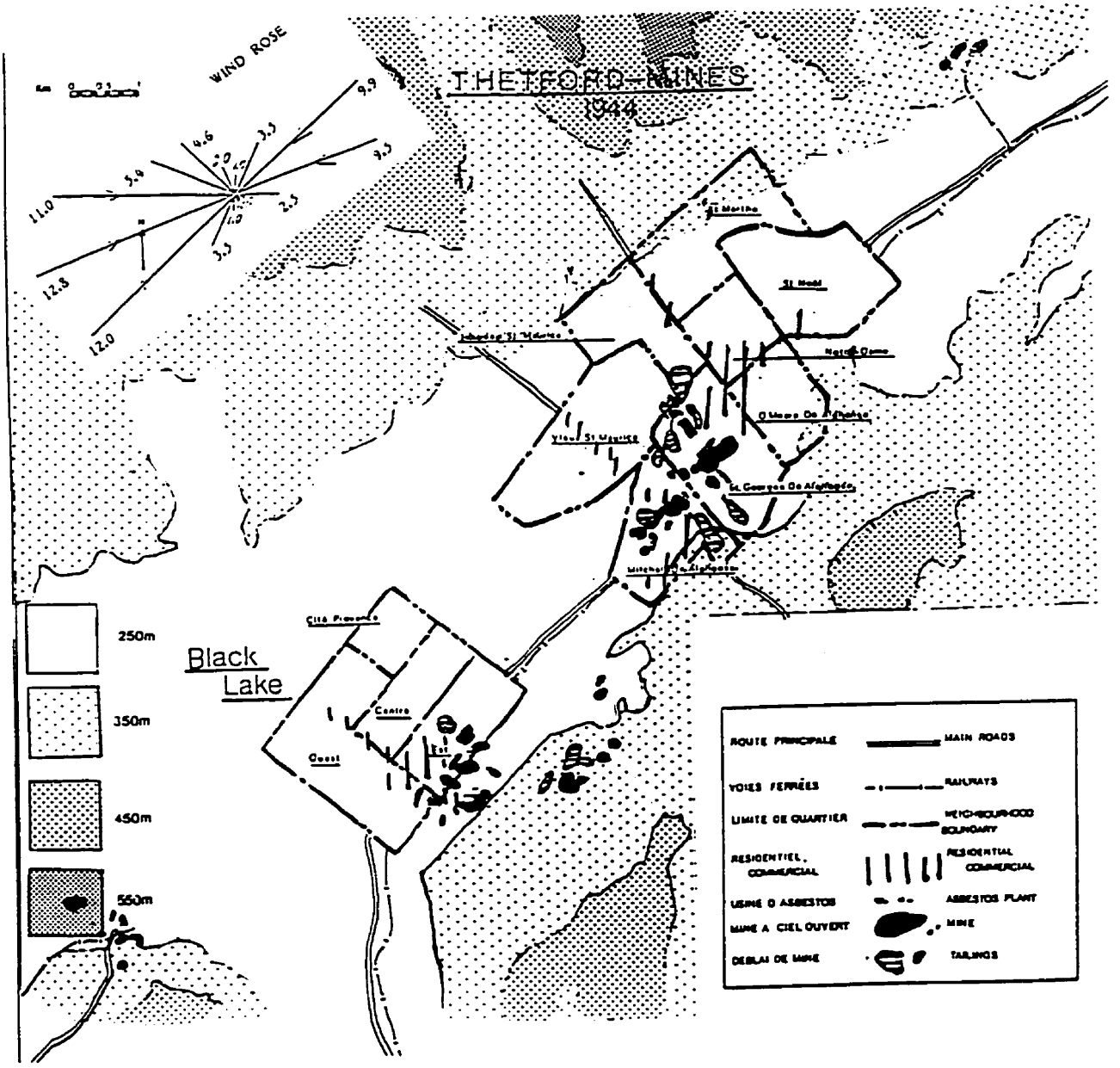


Figure B2-7 Schematic Map of Thetford Mines and Black Lake in 1954

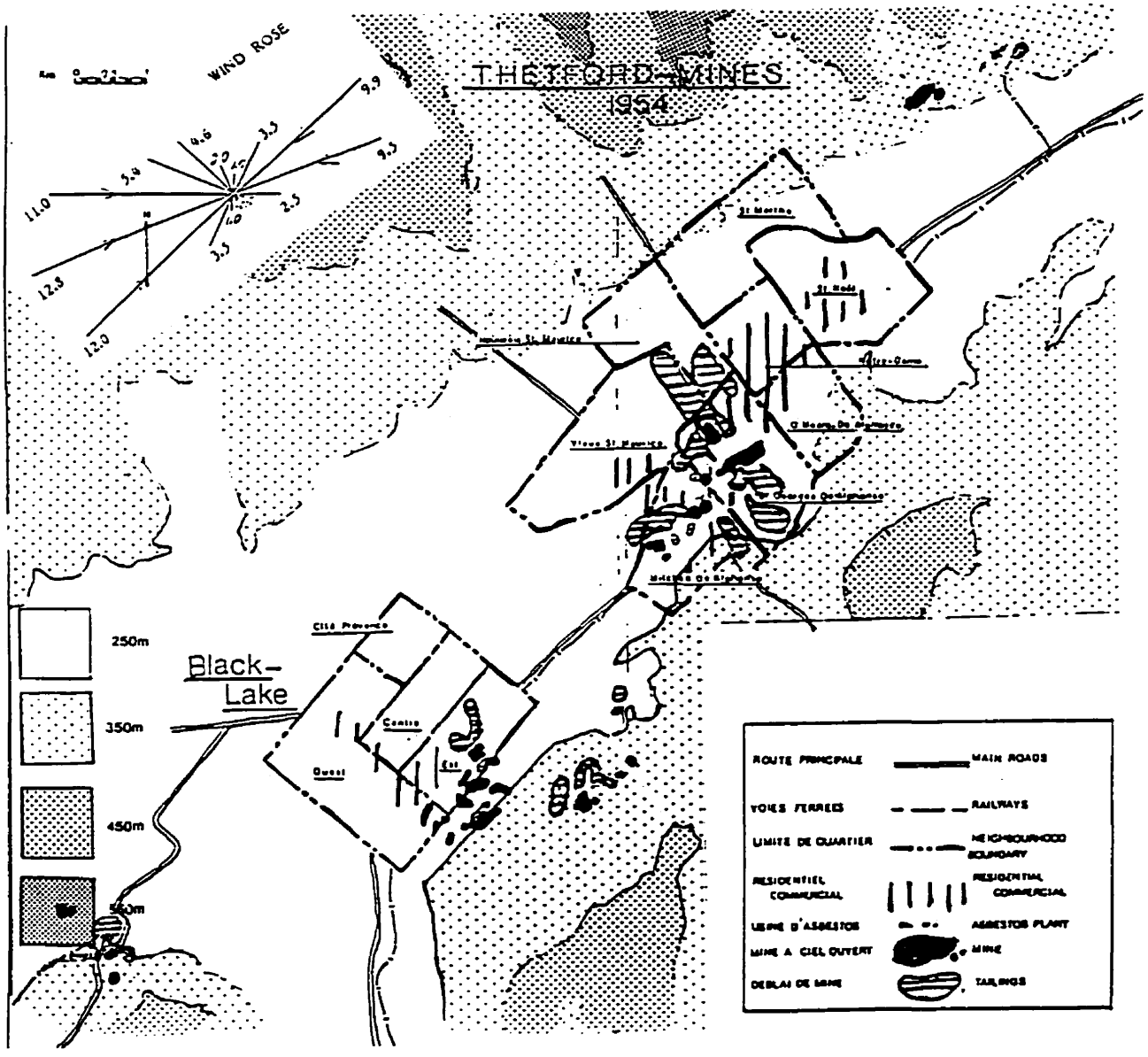


Figure B2-8 Schematic Map of Thetford Mines and Black Lake in 1971

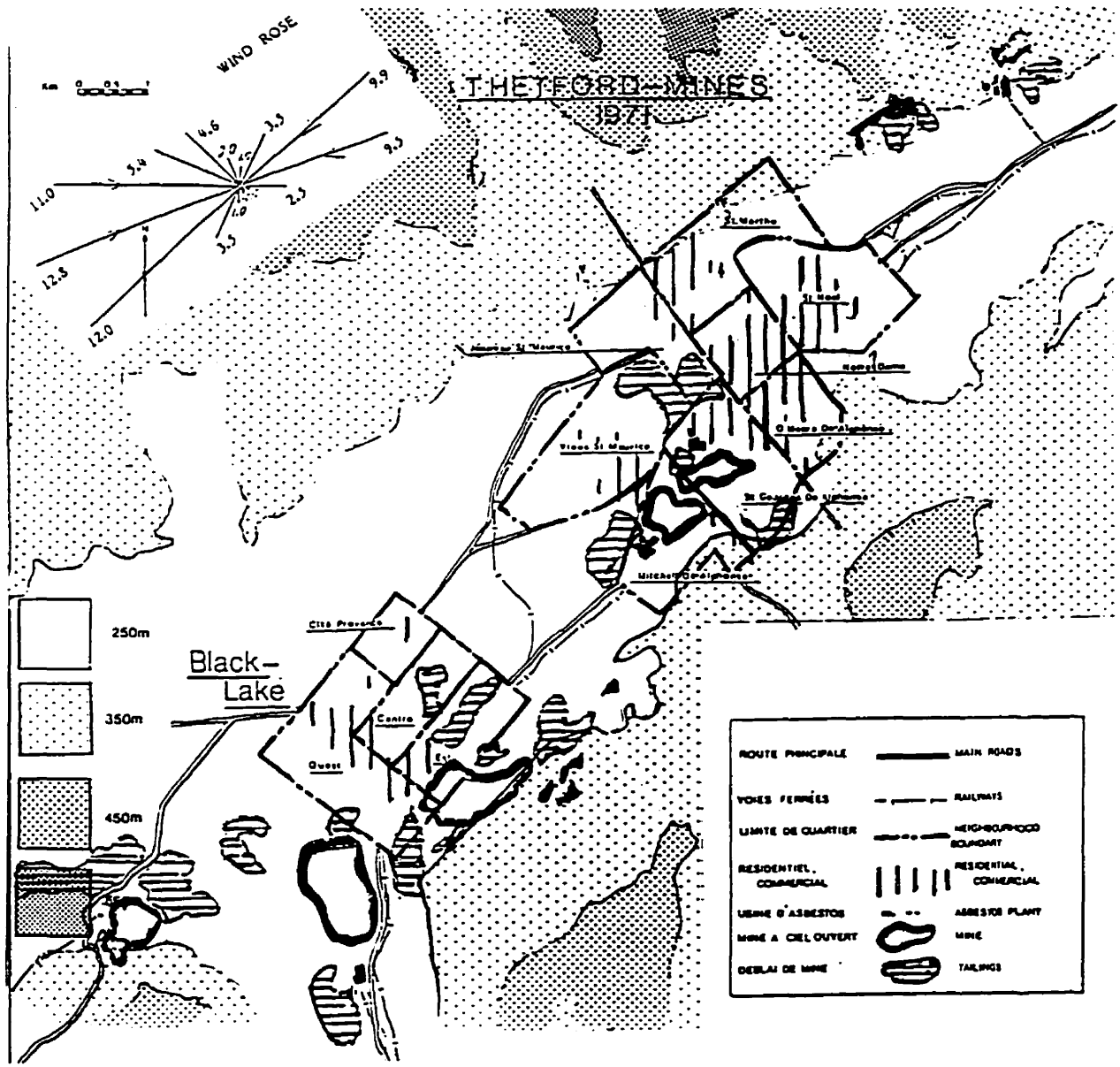
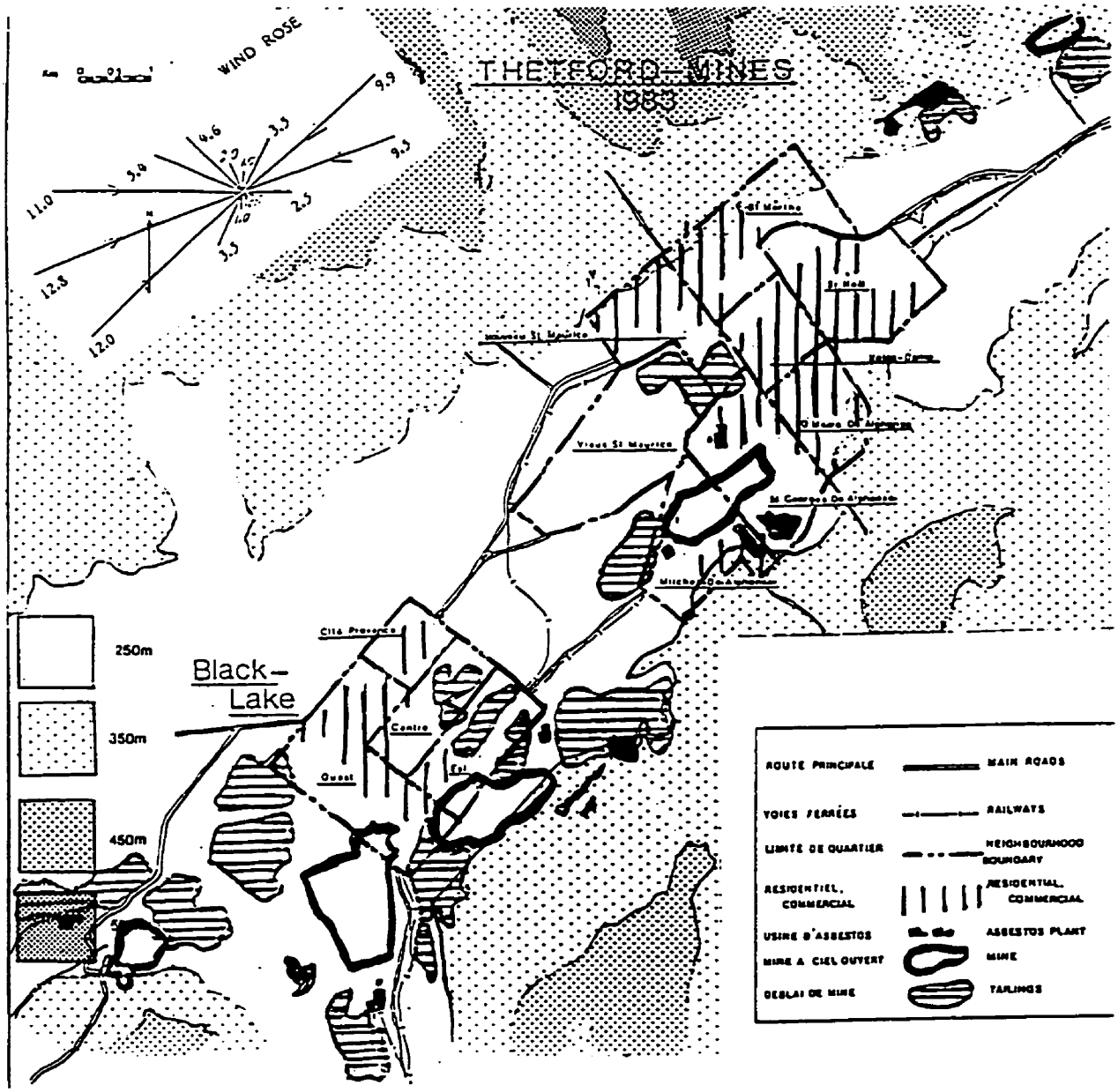


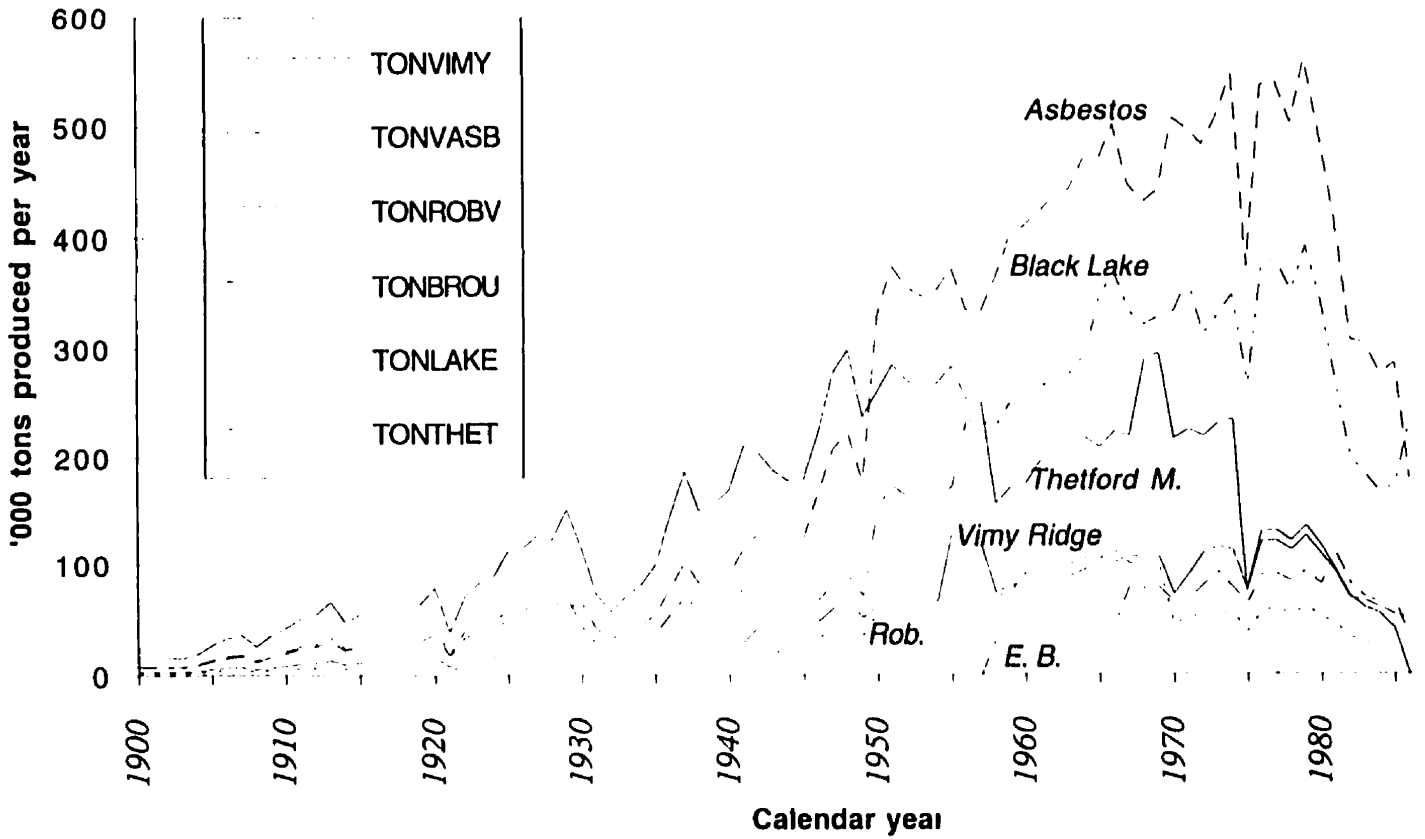
Figure B2-9 Schematic Map of Thetford Mines and Black Lake in 1983



Appendix B3	Annual Production Volume by Mining Town

Because production volume data were available by company only and because an individual company could comprise mines and mills in different towns, some town-year-specific data were based on partitioning an individual company's production volume according to the production capacity of its mills. Before 1950, Thetford Mines had the largest asbestos production volume. After 1950, Asbestos became the most important mining/milling centre, followed by Black Lake; from 1945 to 1974, Asbestos' production output increased 5-fold and Black Lake's 4-fold, while Thetford Mines' increased by less than 40%. After 1974, the asbestos slump hit Thetford Mines hardest, reducing further its relative importance as an asbestos producing centre. The interest in these town-specific production levels and trends is the assumption that air pollution levels in each asbestos-mining town should correlate somewhat with each town's production.

Figure B3-1 Trend in Asbestos Fibre Production in Quebec's Asbestos Mining Towns, 1900-1986



In Quebec's asbestos mining areas, mines supply the ore to asbestos mills which extract and classify the fibres into several grades. The fibres are then shipped to consumer industries for use in manufacturing and in construction. In Quebec's asbestos mining towns, very little secondary transformation of asbestos was done, except for a textile plant which operated in Asbestos. Some asbestos was transformed in the Montreal area (e.g. textile, friction materials, sealants, asbestos cement products, floor tile, etc.) and some was used for shipbuilding in the towns of Sorel and Tracy, but by far the largest part of the production (94.5% in 1970) was exported abroad to the head offices (e.g. Johns-Manville in the U.S.A.) of Canadian operations and to other consumer industries.

Asbestos fibres are classified and priced by "grade" based on fibre length from the longest to the shortest: "long" fibres ("crude" #1 and #2, and "spinning" grade #3) comprise fibre bundles 0.5 cm to more than 2 cm long and are used in transmission and conveyor belts, furnace and boiler insulation mats, dryer brakes, electric insulation tapes, ropes, theatre curtains, heat insulation tapes, etc.; "medium" fibres (grades #4, #5, #6) are 1-5 mm long and are used mostly in asbestos cement, roof shingles, paneling, stucco, paper and molded products, acoustic tiles, asbestos felts, pulley brakes, brake linings and clutch facings, electric insulators, etc.; "short" fibres (grades #7 also called "shorts", #8 and #9) are 0.1-1 mm long and are used in molded brake linings, in caulking, as fillers in paints and plastics, joint cements, stucco, floor tiles, etc.; and "floats" or "super-fines" (<200 µm in length) which are used as fillers in plastics. These asbestos fibre grades are further divided and comprise more than 30 regular grades in total, to which are added some 70 customized grades tailored to individual customer needs. The market value per ton depends on the length of fibres; for instance, the value of grade 3 fibres is one order of magnitude greater than that of grade 7 fibres. Grade numbers also indicate to some degree the chronological order in which these products were developed or marketed since the dawn of the asbestos industry.

Appendix B4 Production Process and Emission Sources

To quantify the relative importance of different emission sources in the past, the two earliest inventories of asbestos emissions in Canada were examined. Table B4-1 shows estimated emissions by source in the asbestos-mining-milling operations in Quebec in 1974. The data used to compute the table came from two surveys of all asbestos mining/milling operations in Canada and from available dust emission factors measured by the U.S. EPA for ore crushing and storage, for loading and unloading trucks and for other mining-related operations in American mines; no direct emission measurements or calculations were provided by Canadian asbestos producers. Aside from their very approximate nature, the asbestos emission estimates for year 1974 cannot be applied blindly to earlier periods because significant dust controls were enacted over the two preceding decades particularly in milling.

In 1974, asbestos emissions due to milling operations were by far the most important source (78%) of asbestos pollution in the mining towns. Moreover, since the major impact of dust controls between 1945 and 1974 has been in reducing pollution from mills, it is likely that milling operations represented an even greater proportion of environmental asbestos emissions in the past, before baghouses were installed on asbestos mills. Accordingly, emissions from milling will be the main piece of information in our deliberation on past asbestos pollution levels in the asbestos mining towns.

The other significant sources of asbestos pollution were tailings piles (11%) and dryer stacks (8%). Although the table shows asbestos emissions from tailings disposal to be more important than dryer emissions, the difference could be due to errors in the very simplified and approximate (crude assumptions based on U.S. EPA emission factors compiled in different mining settings) engineering computations. Another question was whether a given mass of airborne asbestos would be more polluting when emitted by dryer stacks or when emitted by tailings piles. The larger volume of air emitted by dryers, the greater height of tailings piles and the respirable fibre fraction of emitted dusts could weigh in different directions, and these relative weights could differ by town, season and era. For instance, in earlier decades, tailings piles were smaller and dryers had no filtration mechanisms to reduce stack emissions; this

gives more importance to dryers as past sources of asbestos pollution. Indeed, the first asbestos emission inventory by MacLaren in 1970 estimated emission factors of 0.1 and 0.7 kg/t (kilograms of emitted dusts per ton of asbestos fibre produced) for tailings and dryers respectively[J.F. MacLaren Ltd., 1973; Gagan, 1975]. The MacLaren report estimated 7 times more asbestos emissions from dryers than from tailings while a more comprehensive report by Gagan in 1977 gave slightly more importance to tailings emissions[Gagan, 1977]. It is difficult to reconcile this discrepancy as Gagan did not explain how his estimates were derived.

Regarding excavating and handling the ore, MacLaren estimated mining to emit 8%. Although the Gagan report does not mention the contribution of the mining process itself (excavating, blasting, drilling, etc.) to environmental air pollution for reasons mentioned above, it is unlikely that mining operations would have contributed significantly to the air pollution of the asbestos mining towns by respirable asbestos fibres.

Table B4-1 Contribution of Different Processing Operations to the Total Volume of Emissions, Estimated by Environment Canada, 1974

Source	Total dusts tons (%) (col.1)	Asbestos tons (%) (col.2)	Asbestos content in % of dusts emitted by source (col.2)÷ (col.1)	Emission factor[1]: grams of airborne asbestos per ton of fibre produced
Mining	---	---	---	---
Milling	17,219 (11.0%)	5,327 (78.3%)	30.9 %	3,500
Drying	11,416 (7.3%)	571 (8.4%)	5.0 %	380
Tailings	124,739 (80.0%)	746 (10.9%)	0.6 %	500
Crushing	2,354 (1.5%)	156 (2.3%)	6.6 %	100
Storage	103 (0.1%)	5 (0.1)	4.9 %	3
Total	155,831 (100%)	6,805 (100%)	4.4 %	4,500

[1] The emission factor (g/t) as referred to in the table is the mass of respirable dust expressed in grams emitted in outdoor air over a given time interval divided by the total asbestos fibre production volume expressed in tons over the same time interval.

The main conclusion on asbestos production and emission processes is that milling must have been the determining factor of asbestos pollution levels and trends in asbestos mining towns. This conclusion may not be fully in accord with recollections by many residents and visitors who have been impressed by visible emissions from dryers and from tailings piles in the early 1970s. However, dryer and tailings emissions could well have appeared more dense than they actually were; hot gas made dryer emissions more visible due to condensation in colder weather, and tailings piles emitted a large proportion of aerosols which were either not respirable or not asbestos fibres. Indeed, visible dustiness must have correlated better with total dust emissions than with finer respirable asbestos emissions, yet the distribution of total dust emissions by emission source, was utterly different from the distribution of asbestos emissions, as shown in Table B4-1.

Estimated asbestos emissions from tailings piles were somewhat larger than the dryer emissions estimated by Gagan for 1974, but uncertainty related to methods of estimation would nullify the apparent difference. It is hard to say if dust emission controls improved more for dryers than for tailings piles over the 1950-1974 period and thus it cannot be said which of these two sources was most important in the far past. Nevertheless, whether or not respirable asbestos emissions from tailings piles were more important than those from the dryers, the absence of data on tailings emissions or even on the volumes or surface areas of tailings piles and the natural assumption that dryer emissions depended on the easily accessible production volume, there was more basis to project dryer emissions than tailings piles emissions over different time periods. Hence, asbestos emissions from tailings piles were not addressed specifically in the present exposure assessment.

Appendix B5 Efficiency of Emission Controls, Regulations and Maintenance

In an internal report of the Environment Quebec Ministry[Brulotte, 1980], the responsible for environmental surveillance in the asbestos mining towns wrote in 1980: " The expression *It snows in July in Thetford* was still justified in the early 1970s; the expression evidently alluded to the asbestos dusts that many referred to as *cotton* . " Such visible dustiness, a rising environmental consciousness and knowledge about asbestos' toxicity, and the legal enforcement of ambient air standards led some citizens groups to make complaints to the Environment Ministry about sporadic local dust emissions or fallouts from 1974 on. These complaints are documented by some photographs shown in Figure B5-1 and show that dust emissions were still not well controlled in the Thetford Mines - Black Lake area before the 1980s, despite the controls in place since 1974. The problematic emissions came mostly from a few dryers still equipped with electrostatic precipitators or with automatic baghouse bypass systems (to avoid burning the bags) and from tailings disposal. Complaints decreased steadily and became rare in the 1980s.

The regulatory surveillance of emissions by Environment Quebec tells more objectively of the emission control problems and trends. Internal reports[Brulotte, 1976; Brulotte, 1980; Boisjoly, 1988] on the compliance of asbestos producers to governmental emission standards over the 1979-1990 period show a continuous improvement from 81%[1] compliance in 1979 to 95% in 1984 and 100% in the late 1980s[2]. 48% of infractions occurred in storage, 29% in drying and 21% in milling activities. Much of the improvement over the 1983-1986 period was due to the shutdown of delinquent operations.

Most infractions on baghouses were explained by poor maintenance (unreplaced torn bags). If such infractions were quite common after 1975 when governmental departments regulated the industry, it is reasonable to assume that emission levels above today's standards were even more common in the early days of baghouse controls in the 1950s and 1960s. Thus it is likely that the real efficiency of baghouses in the 1950s and 1960s was much lower than the

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[1] Proportion of mandatory samples not exceeding the standard values.

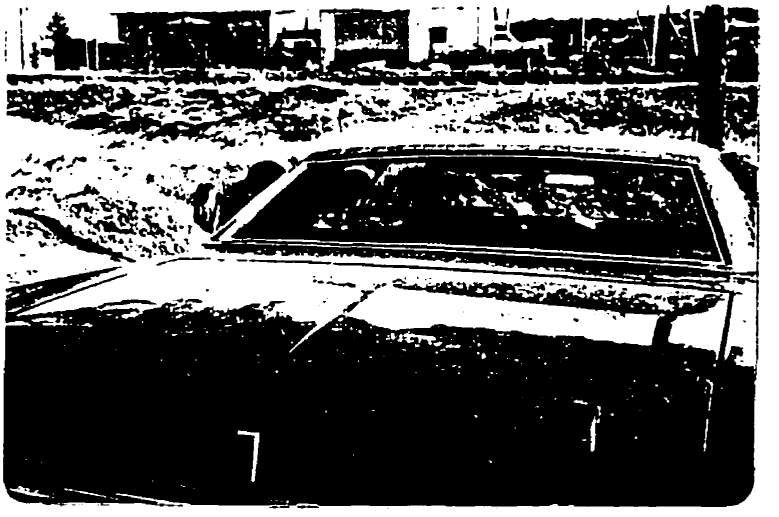
[2] The compliance rates reported here are slightly lower than the official statistics. The latter comprise the extra samples required to correct the detected problems.

theoretical 95% efficiency rating in those years. In effect, local experts and long-standing employees in different plants told us that maintenance improved tremendously over the last 20 years since the 'asbestos crisis' pushed the industry to improve its image and since the Federal and Provincial Environment Ministries started to impose emission standards. From recall, they believed also that the personnel, moneys and programs allocated to dust control maintenance increased and improved ever since the first baghouses were installed.

Maintenance and trends in maintenance must be taken into account to estimate past emissions.

Figure B5-1 Pictures of Stray Emissions in the 1970s
- Environnement Québec -

Visible emissions of dryers when the baghouse is in bypass due to excessive gas temperature in 1979



Dust deposits on a car after the breakage of a tailings conveying belt or of a slinger in 1977

Malfunction of an electrostatic precipitator in October 1979

Appendix B6 Characteristics of Dusts Emitted Before the Introduction of Controls

Although estimating the volumes of emissions before the introduction of dust controls is crucial for the exposure assessment, it does not tell if past emitted dusts were etiologically relevant, that is if they were respirable asbestos fibres and if they would fall in the neighbouring residential areas. The nature of past emissions can be determined by estimating: 1) the respirable proportion of dusts retained by the baghouses, 2) the asbestos fibre proportion of these dusts, 3) the dusts' aerodynamic diameter distribution to be used in an aerosol dispersion simulation model, 4) the contamination of chrysotile emissions by amphiboles, and 5) the fibre length and diameter distribution in different classes.

Most of the fibres that in the past would have been expelled from the mills into the town environment are today captured in the baghouses. Consequently some key characteristics of the dust that was expelled in the past can be measured by examining the retained by today's baghouses. We analyzed the dust retained in today's baghouses as a window on the past.

In November-December 1990, following our request, the plant managers of the remaining four mills and three dryers[1] in Quebec's asbestos mining region agreed to sample the dusts conveyed from the baghouses to the tailings piles at a point in the duct or on conveyor belt preceding the mixing of the rejects from the baghouses with other refuse from the rock circuit. A one-kilogram "grab sample" was taken at the beginning of each shift for 2 consecutive weeks by the leader of each team of workers, i.e. three times (at each 8-hour shift) per day, 6 days per week. This worker had been shown by the local industrial hygienist how to take the samples. At each site, one bag was used to accumulate all samples taken during a given week. There were 4 mills and 3 dryers, each sampled for 2 weeks, for a total of 14 samples or dust bags of roughly 18 kilograms each (3 shifts x 6 days). The sampling was conducted under the supervision of M. Camus (Institut Armand-Frappier) and Dr. A. Dufresne (McGill University's Occupational Health Microscopy Laboratory).

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[1] The plants involved were: J-M Asbestos, Bell (no dryer), B-C and Lac d'Amiante, the last three belonging to LAB-Chrysotile.

Dr. Dufresne took a representative 500 g sample from each bag. Each 500 g sample was then separated by the research division (CERAM) of the Société National de l'Amiante (SNA) into six D_{ae} particle size strata (Table B6-1), using a standard Ro-Tap sifter to classify the dusts gravimetrically for each sample. Transmission electron microscopy (TEM) was then performed at McGill University's Occupational Health Microscopy Laboratory to further characterize the particles in the respirable D_{ae} strata ($D_{ae} < 6.5$ µm), counting the number of fibres (aspect ratio > 3:1) and all particles and measuring the length and width of these particles. In addition, one sample was pooled from all samples from the Thetford Mines - Black Lake area, and one was pooled from the two samples from Asbestos, to determine the proportion of amphiboles in the dusts by EDXA (energy dispersion x-ray analyzer).

Table B6-1 Characterization of Dusts Captured in Baghouses: of Grab Sample Measurements by Particle Size

Aerodynamic equivalent diameter D_{ae}	Gravimetric Distribution (% Mass)		Approximate[1] Numeric Distribution	
	Dryer s	Mills	Dryers	Mills
< 2.5 µm	25.9	29.6	95.61	96.57
2.5 - 4.3 µm	20.1	17.7	3.69	2.87
4.3 - 6.5 µm	11.3	10.8	0.52	0.44
6.5 - 11.5 µm	17.0	13.0	0.17	0.11
11.5 - 33 µm	22.2	23.0	0.01	0.01
> 33 µm	3.5	6.0	0.00	0.00
TOTAL	100.0	100.0	100.00	100.00

1 The proportion of particles per D_{ae} stratum was estimated by assuming 1) that particulate mass was proportional to the volume of particles of identical densities and 2) that particulate volume was proportional to the stratum's mean D_{ae} elevated at the cubic power since $V = \frac{\pi}{3} \bullet \left(\frac{D_{ae}}{2} \right)^3$.

The dust retained by the baghouses was mostly respirable (Table B6-1), with nearly 60% of the total mass and 99.8% of the estimated particle count consisting of particles with $D_{ae} < 6.5$ µm. The D_{ae} distribution did not differ between mills and dryers, even though the fibres in the mills' baghouses would be expected to be smaller, being at the end of the fibre extraction process.

According to electron microscopy performed on the respirable dust fractions, 95% of the particles seen and counted by TEM were "short" fibres (< 5 µm length), 5% were so-called Stanton fibres (length > 8 µm, diameter < 0.25 µm) and 1% were "optical" fibres (length > 5 µm, diameter > 0.25 µm). Paradoxically, this distribution did not vary significantly from one D_{ae} stratum to another although length and diameter should be proportional to D_{ae}. This probably reflects a clumping phenomenon in the Ro-Tap sifting (respirable fibres adhering to or being entrained by larger particle) and an effect of the analytic technique (indirect method and high-resolution electronic microscopy).

1% of the fibres were estimated to be tremolite (4 in 380 fibres characterized by EDXA): 3/180 (1.5%) in Thetford Mines - Black Lake, and 1/200 (0.5%) in Asbestos.

Methodological limitations hamper the interpretation of these results. First, the Ro-Tap classification machine is too imprecise to determine the true size distribution of fine dust samples. Indeed, dry classification systems such as the Ro-Tap cannot completely separate fibres of different sizes; many short fibres stick to the longest fibres and end up erroneously in the larger D_{ae} strata. This "size overestimation" misclassification might not affect substantially the mass distribution given by the Ro-Tap process however, because small fibers would contribute little to the mass of the largest D_{ae} strata. This would partly explain that the TEM examination of dusts from different D_{ae} size strata would find mostly short fibres in all D_{ae} strata. For instance, since a 1.25 µm D_{ae} particle has a mass about 46,000 times smaller than that of a 45 µm D_{ae} particle, there could be thousands of the smallest fibres contaminating the largest strata without affecting materially the mass of these strata. Finally, the misclassification of fibres by the Ro-Tap gravimetric classification may more accurately reflect

flocculation which occurs in extremely dense aerosol clouds. In such clouds, falling particles take other floating particles down with them, growing as they fall as with a "rolling snowball effect"; as they fall they behave more and more as larger particles. Due to its sifting characteristic, the Ro-Tap classification might thus be more relevant than the true size distribution of the dust samples. However this interpretation is speculative.

Second, according to two experts on the exposure assessment panel (P. Sébastien and W. Nicholson, personal communications), TEM is probably too accurate to give a realistic picture of the fibre length and diameter distribution because the narrow field of view of a TEM cannot see particles larger than that field; thus TEM would have missed the largest particles just as if one tried to count trees in a forest through the eye of an optical microscope. However the proportions of short, Stanton and optical or PCOM fibres in the respirable D_{ae} strata must be relatively accurate because there should not be many large particles in these strata. Also, the TEM results were consistent with data reported by other investigators[Hwang, 1983] who characterized the size distribution of respirable fibres in the ambient air of asbestos mills' bagging departments at the end of the milling process where fibre aerosols might be as fine as those retained by the baghouses; 96% of the fibres were short, 1.2% were Stanton and 1.3% were optical (PCOM). Still, results of the present TEM analyses remain suspicious in the larger D_{ae} strata due to the limitations of the technique in detecting the largest fibres which should occur in these strata.

Appendix B7	Hypothetical Example of Historical Differences in Gravimetric and Numeric Size Distributions of Respirable Asbestos Emissions

The following hypothetical example was contrived to determine what numeric emission ratios could be consistent with the gravimetric ratios and with the size distribution of dusts retained by baghouses. With respect to the latter, it was assumed that non-respirable particles ($D_{ae} > 5$ µm) constituted 5% of post-cyclone dust particles and 40% of their mass and that respirable particles constituted 95% of the post-cyclone particles and 60% of their mass. Supposing that in 1945 float sheds stopped 25% (numeric proportion) of the post-cyclone non-respirable particles and 1% of the respirable particles, the gravimetric filtration efficiency of float sheds would have been 11%[1], and their numeric filtration efficiency 2%[2]. Supposing that in 1974 baghouses and other filtration systems stopped 99% of the non-respirable and 97% of the respirable particles, then the gravimetric filtration efficiency would have been 98%[3], and the numeric efficiency 97%[4]. Finally, supposing that generalized, improved and well maintained baghouses stopped 100% of the largest and 99.95% of the smallest particles in 1984, then the average gravimetric filtration efficiency would have been 99.97%[5], and the numeric filtration efficiency 99.95%[6].

The first part of Table B7-1 shows penetrance factors (= 1 - Efficiency) and the second part shows penetrance ratios derived in the example. "Penetrance" is the proportion of dusts escaping the controls. A ratio of penetrance factors is equivalent to an emission factor ratio. In the example, numeric emission ratios were about 20% smaller than gravimetric emission ratios between 1945 and 1974 and 20% smaller again between 1974 and 1984; between 1945 and 1984, at a constant production level, the proportionate reduction in the number of emitted particles would have been 2/3 of the proportionate reduction in the mass of emitted particles. Changes in numeric respirable emissions were practically identical to changes in numeric global emissions on a ratio scale because the assumed respirable fraction (95%) was very high.

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1 (25% x 40%) + (1% x 60%) = 11% gravimetric efficiency = 89% gravimetric penetrance.
2 (25% x 5%) + (1% x 95%) = 2% numeric efficiency = 98% numeric penetrance.
3 (99% x 40%) + (97% x 60%) = 97.8% gravimetric efficiency = 2.2% gravim. penetrance.
In his 1973 report[Denizeau, 1973], an Environment Quebec engineer mentioned a 99.7% gravimetric efficiency (p.12).
4 (99% x 5%) + (97% x 95%) = 97.1% numeric efficiency = 2.9% numeric penetrance.
5 (100% x 40%) + (99.95% x 60%) = 99.97% gravimetric efficiency = 0.03% gravim. penetrance.
6 (100% x 5%) + (99.95% x 95%) = 99.95% numeric efficiency = 0.05% numeric penetrance.

The example's differentials between gravimetric and numeric emission *factor* ratios were applied to the gravimetric respirable emission *volume* ratios in Table B6-1 (Appendix B6), and the resulting numeric respirable emission volume ratios are shown in the third part of Table B7-1. The most reliable result is the 1945-1974 comparison; hence, the number of respirable asbestos particles emitted in 1945 in the asbestos mining towns would have been about 7 times higher than that in 1974.

Table B7-1 Hypothetical Example of Changes in Gravimetric and Numeric Asbestos Emissions

Penetrance[1]	Pre-controls (pre-1945) float sheds penetrance	Penetrance of baghouses in 1974	Penetrance of properly maintained baghouses in 1984
Gravimetric	89%	2.2%	0.03%
Total Numeric	98%	2.9%	0.05%
Respirable Particles	99%	3.0%	0.05%

Chronological Comparisons of Penetrance Factors

Penetrance or Emission Factor Ratio[2]	1945 vs. 1974	1974 vs. 1984	1945 vs. 1984
Gravimetric Emission Factor Ratio	40	73	3,000
Numeric Emission Factor Ratio	34	60	2,000
Respirable Numeric Emission Factor Ratio	33	60	2,000
Numeric to Gravimetric Quotient	4/5	4/5	2/3

Corrected Comparisons of Respirable Emission Volumes

	1945 vs. 1974	1974 vs. 1984	1945 vs. 1984
Gravimetric Respirable Emissions Ratio	≈ 9	< 800	< 7,000
Numeric Respirable Emissions Ratio	≈ 7	< 650	< 4,500

1 Penetrance = 1 - efficiency. It is the proportion of particles passing through a dust control system or apparatus.

2 Although penetrance and emission factors are not identical, a penetrance ratio is identical to an emission factor ratio.

Appendix B8 The ISC-LT Aerosol Dispersion Model and Projections for 1972

Table B8-1 ISC-LT Aerosol Dispersion Model Formula

For a single stack, the mean seasonal concentration at a point ($r > 1$ m, θ) with respect to the stack is given by:

$$X_\ell = 2K\,(2\pi)^{-1/2}\,(r\Delta\theta')^{-1} \sum_{i,j,k} Qf_{SVD}(u\sigma_z)^{-1}$$

where

- Q = pollutant emission rate (mass per unit time), for the i^{th} wind-speed category, k^{th} stability category and ℓ^{th} season
- f = frequency of occurrence of the i^{th} wind-speed category, j^{th} wind-direction category and k^{th} stability category for the ℓ^{th} season
- $\Delta\theta'$ = the sector width in radians
- S = a smoothing function similar to that of the AQDM (see Section 2.5.1.3)
- u = mean wind speed (m/sec) at stack height for the i^{th} wind-speed category and k^{th} stability category
- σ_z = standard deviation of the vertical concentration distribution (m) for the k^{th} stability category
- V = the Vertical Term for the i^{th} wind-speed category, k^{th} stability category and ℓth season
- D = the Decay Term for the i^{th} wind speed category and k^{th} stability category
- ψ = the decay coefficient (sec^{-1})
- K = units scaling coefficient

The mean annual concentration at the point (r,θ) is calculated from the seasonal concentrations using the expression:

$$X_a = 0.25 \sum_{\ell=1}^{4} X_\ell$$

Figure B8-1 Representation of the Dust Plume Emitted by a Stack

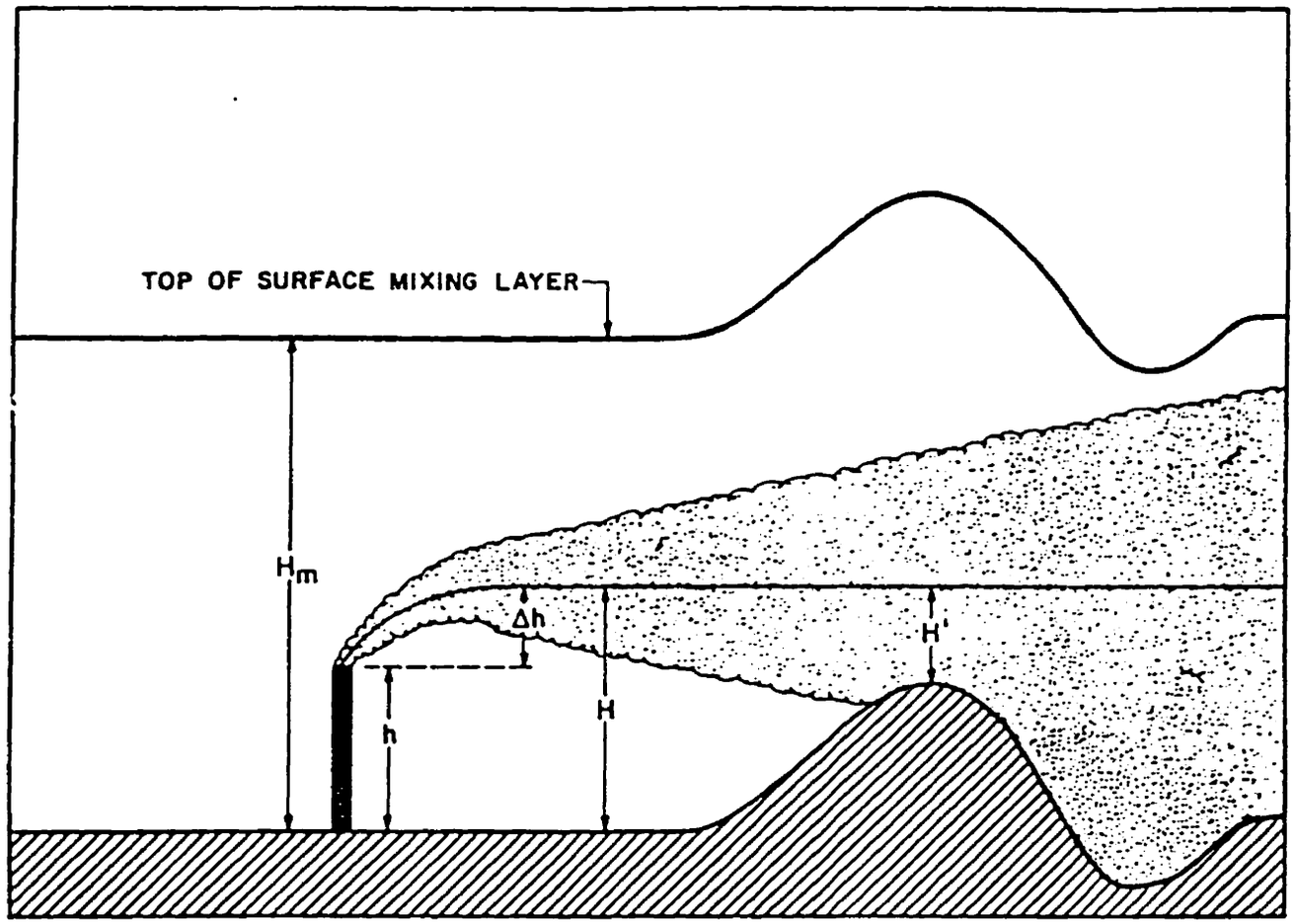


Figure Illustration of plume behavior in complex terrain assumed by the ISC Model.

Figure B8-2 Graphical Representation of the Vertical Gaussian Concentration Profiles Resulting From the "Reflection" of the Dust Plume Hitting the Ground

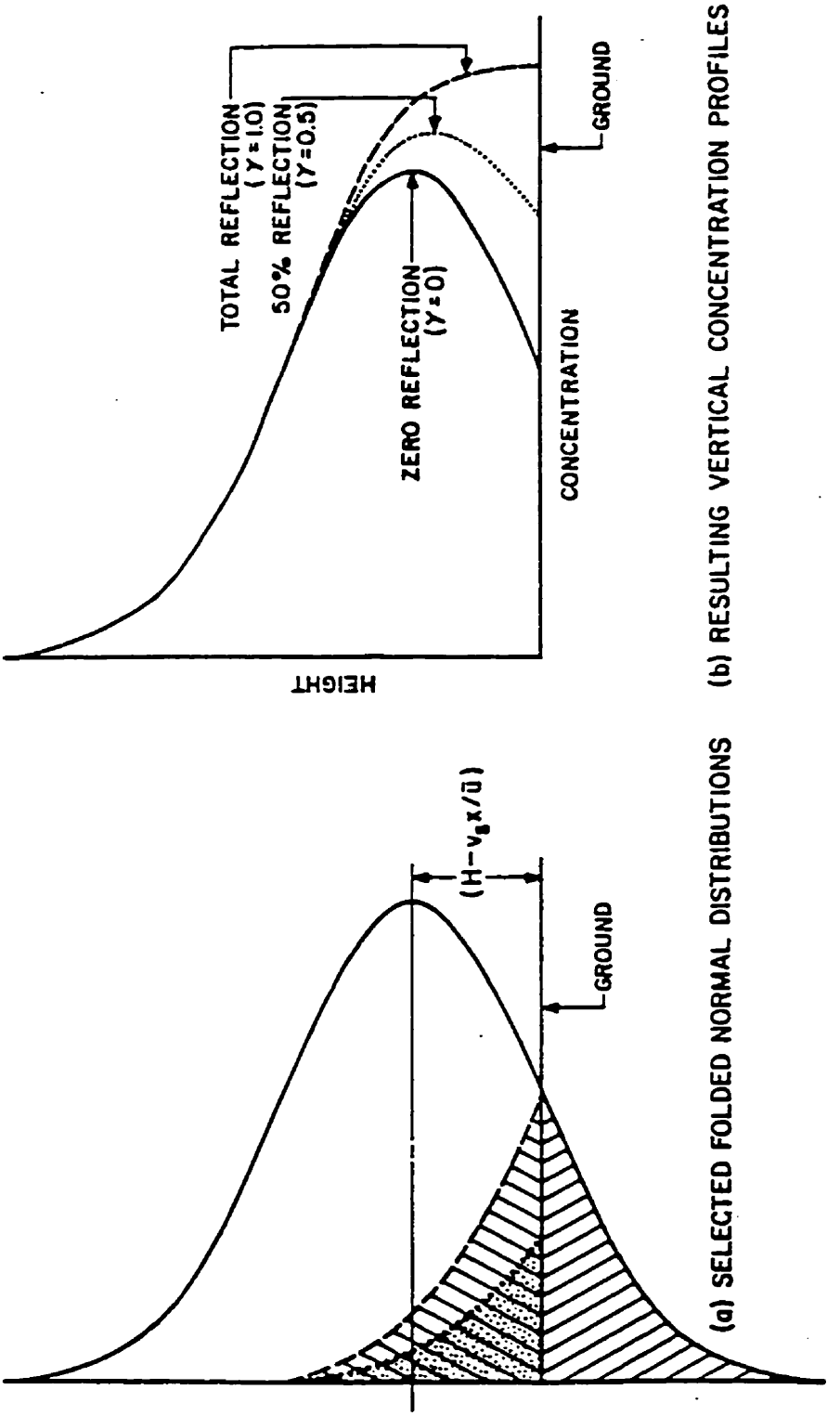


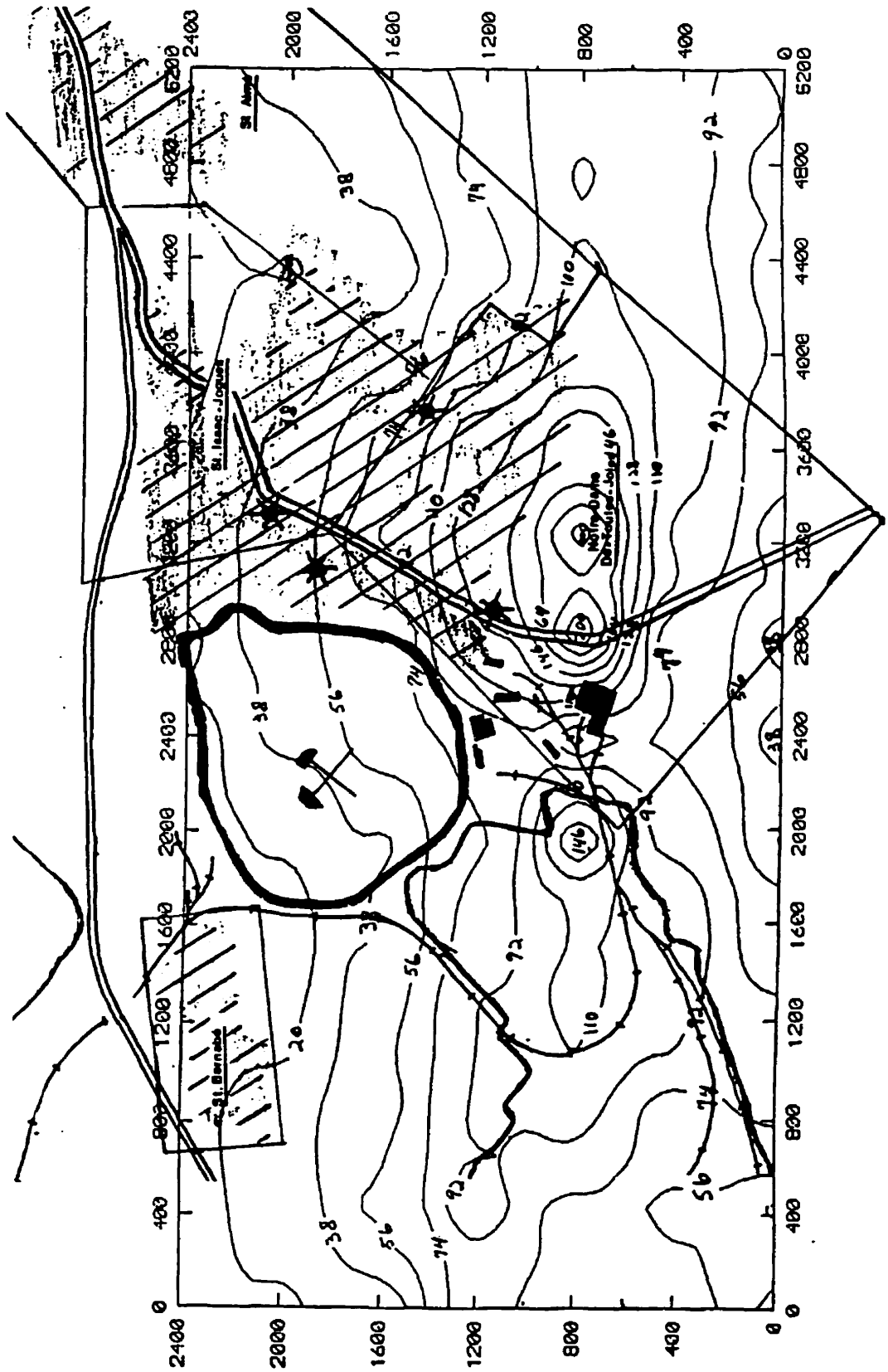
FIGURE Illustration of vertical concentration profiles for reflection coefficients of 0, 0.5 and 1.0.

Table B8-2 Source Inputs Required by the ISC Model Programs, With Definitions

Parameter	Definition
Stacks	
Q	Pollutant emission rate for concentration calculations (mass per unit time)
Qt	Total pollutant emissions during the time period t for which deposition is calculated (mass)
ψ	Pollutant decay coefficient ($seconds^{-1}$)
X, Y	X and Y coordinates of the stack (meters)
Z_s	Elevation of base of stack (meters above mean sea level)
h	Stack height (meters)
v_s	Stack exit velocity (meters per second)
d	Stack inner diameter (meters)
T_s	Stack exit temperature (degrees Kelvin)
ϕ_n	Mass fraction of particulates in the n^{th} settling-velocity category
V_{sn}	Gravitational settling velocity for particulates in the n^{th} settling-velocity category (meters per second)
γ_n	Surface reflection coefficient for particulates in the n^{th} settling-velocity category
h_b, BHj	Height of building adjacent to the stack (meters); direction specific building heights (meters) for the j^{th} wind direction category. The direction specific heights are required by the Schulman-Scire building wake effects method.
W, BWj	Width of building adjacent to the stack (meters); direction specific building widths (meters) for the j^{th} wind direction category. The direction specific heights are required by the Schulman-Scire building wake effects method.
L	Length of building adjacent to the stack (meters)

Figure B8-3 presents isolines of dust concentrations projected by the ISC-LT model for the town of Asbestos, for year 1972. The numbers on the isolines represent projected $\mu g/m^3$ levels of aerosols resulting from the mills' and dryer's dust emissions. The numbers on the axes represent meters. The Y-axis represent the north-south axis, the larger numbers representing the northern direction. The shaded areas represent the residential areas, and the starred black spots indicate the approximate location of the four Environnement Québec dust sampling stations.

Figure B8-3 ISC-LT Dust Concentration Isolines Projected in Asbestos, Dryer and Mills' Emissions Year 1972



Appendix B9 Past Visible Asbestos Pollution: Anecdotes and Photographs

Anecdotes Regarding Past Visible Asbestos Exposures

1. Residents of the asbestos mining towns have long seen fallouts of dust clouds emitted by tailings piles, dryers' stacks and mills' louvers. Anecdotes were abundant in our survey and in the few panel interviews that we conducted for our feasibility study.

2. Common saying: "In Thetford, it snows in the middle of July".

3. A novel[Langevin, 1950b; Langevin, 1950a] and a Canadian movie on Thetford Mines were named "Dust Over the City".

4. Dr. Clément Fortier, MD, Former director of the Hôpital général de la région de l'Amiante, wrote a history book, "Black Lake, lac d'amiante, 1882-1982" in which he recalls (p. 149)[Fortier, 1983]:

   "... After a day of westerly winds, dust would literally "fall" on the town. In less than 30 minutes, Brylcream and wave-set lost their shine and heads turned white starched by the night. In the morning, the observer could track and identify the footprints of those pious persons that he would soon see in church. Back home, he was greeted by his mother sweeping the front porch to prevent the asbestos dust from spreading over the house's linoleum floors. Dusting the furniture daily was bothersome enough."

5. Old women told us that toddlers and young children would lie, play and roll on lawns covered with white dust. If a bedroom window was left open overnight, fluffs would be found on the floor in the morning. During winter, windows were draught-proofed with dust collected in and around the house. Before the 1940s or 1950s, many women would card and spin asbestos wool and knit socks or centerpieces for dining tables.

6. A few old-timers told us that all cars had the same color - grey. Retailers had to dust the cars once or twice a day. Cars rusted more rapidly in the asbestos towns because asbestos fallouts on the roads would stick under car frames and maintain the humidity (and later the salt-calcium mixture).

7. J.J. Richard wrote in his novel[Richard , 1956] "Le feu dans l'amiante" (*"Burning asbestos"*):

"Dust had accumulated on the sidewalks during the night and it left behind the footprints of a young girl who did not know where to go other than walking on a sidewalk covered with asbestos snow. It did not take long for the dust to erase those footprints."

"The tops of these mountains [tailings piles] smoked continuously like volcanoes. Following the wind's will, the dust clouds sprinkled one side of the town or the other. One day will come when Johnsonville will be discovered like Pompeii was unearthed."

"(...) He finally reached the golf course, a gray lawn covered with fresh green footprints."

8. An ex-industrial hygienist [J. Lebel] from the asbestos industry told us:

"In the spring of 1974, I went to a sugar-loaf party in a maple grove 1 mile downwind from the Carey active tailings pile. Suddenly strong winds brought down on us so much asbestos dust - fresh from the tailings pile, - that we had to cancel the whole thing and leave immediately. It was unbearable. I think that I could have measured more than 15 f/cc in the cloud surrounding us, more than in the workplace. (...) Visible dust clouds extending 1\2 or 2 km downwind from the tailings were a normal thing and occurred on average once or twice a week, I would say".

9. Periodic complaints were made to Environment Quebec by citizens' groups regarding visible environmental asbestos pollution. These tailed off only around 1979. In fact, citizens and city-hall representatives had been complaining to the companies' at least since 1912 in Thetford Mines.

10. Friar Fabien wrote in his history of the town of Asbestos[Frère Fabien, 1977], p.18:

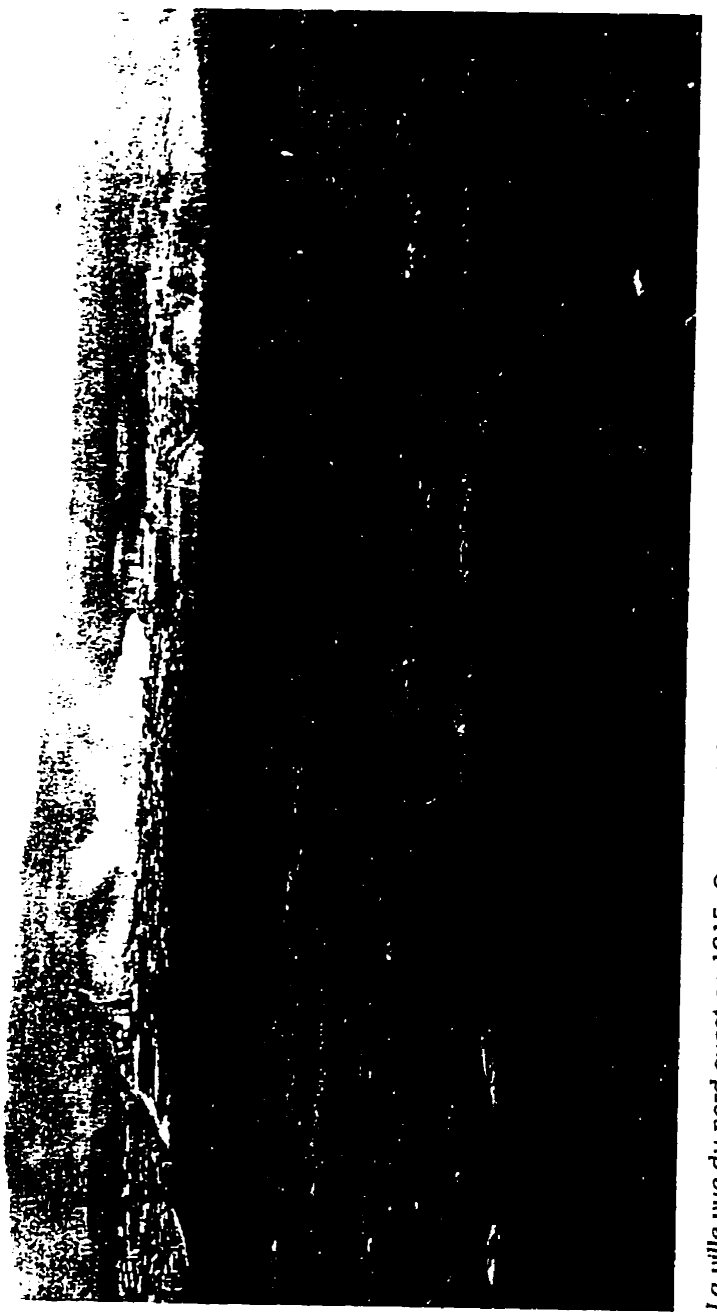
" What used to bother the citizens of Asbestos was the dust produced by the fiberizing process which, like la light snow, penetrated everywhere. However, the situation has improved a lot, and dust is now much less frequent. "

11. Elementary school teachers used to receive bags of asbestos from the companies to be used as molding paste by the children. This was a common leisure of children and even

of some adult residents: molding with asbestos dust. Ashtrays were often made in that way. Simple enough, only water was added. And then the mold solidified as it dried...

12. Kids often played with "asbestos balls". They would hold these "balls" in their hands and blow on them until they faded away by the dispersion of their constituting fibres and dusts. The game could consist to blow the fastest and the strongest so that your ball would the first to "vanish into thin air".

Figure B9-1 Looking South-East of Thetford Mines in 1915: Dust Emissions From Mining Operations South-West of the Municipality[1]

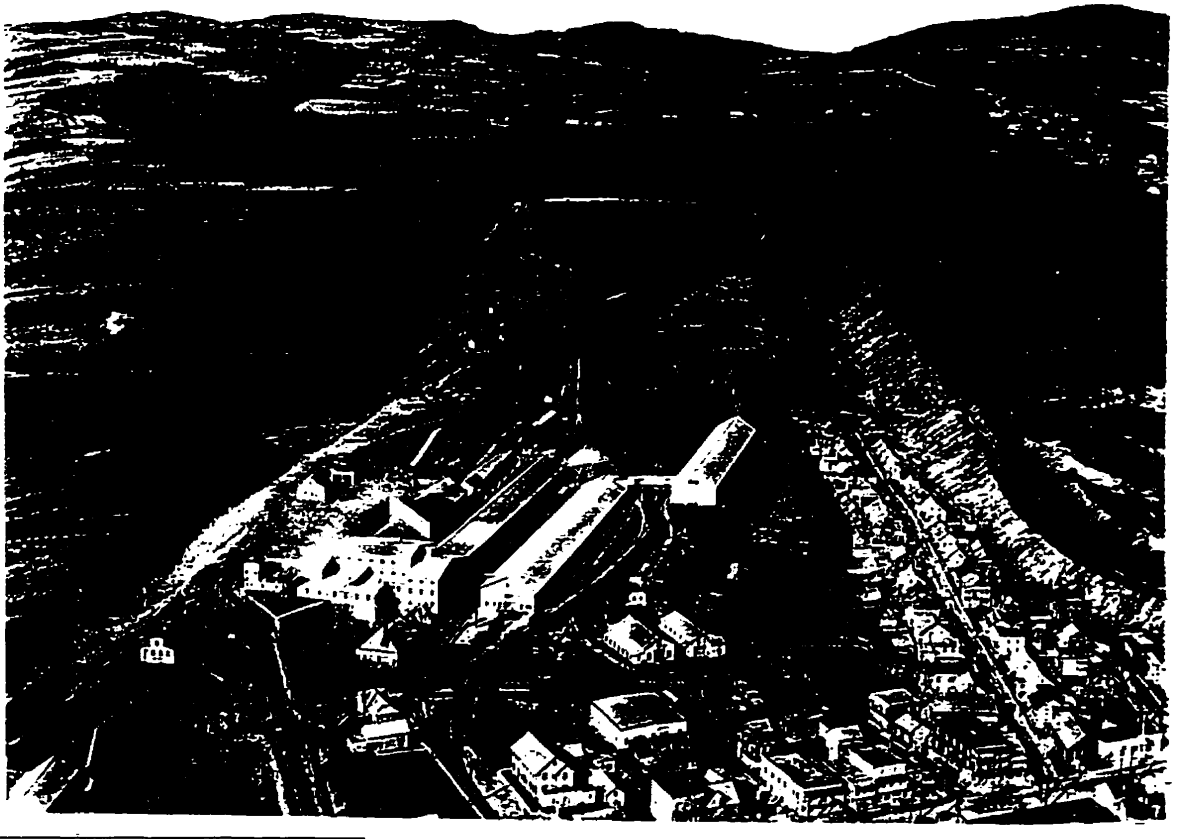


La ville vue du nord-ouest en 1915. On aperçoit les nuages de poussière dégagée par les opérations minières. SAHRA, Collection Musée minéralogique et minier de Thetford Mines. (Geological Survey of Canada nº GSC 34649) (Photo : R. Harvis).

[1] Reproduced from Cinq-Mars et al. [1994, p.200] with the permission of the municipality of Thetford Mines [Mr. Y. Faucher].

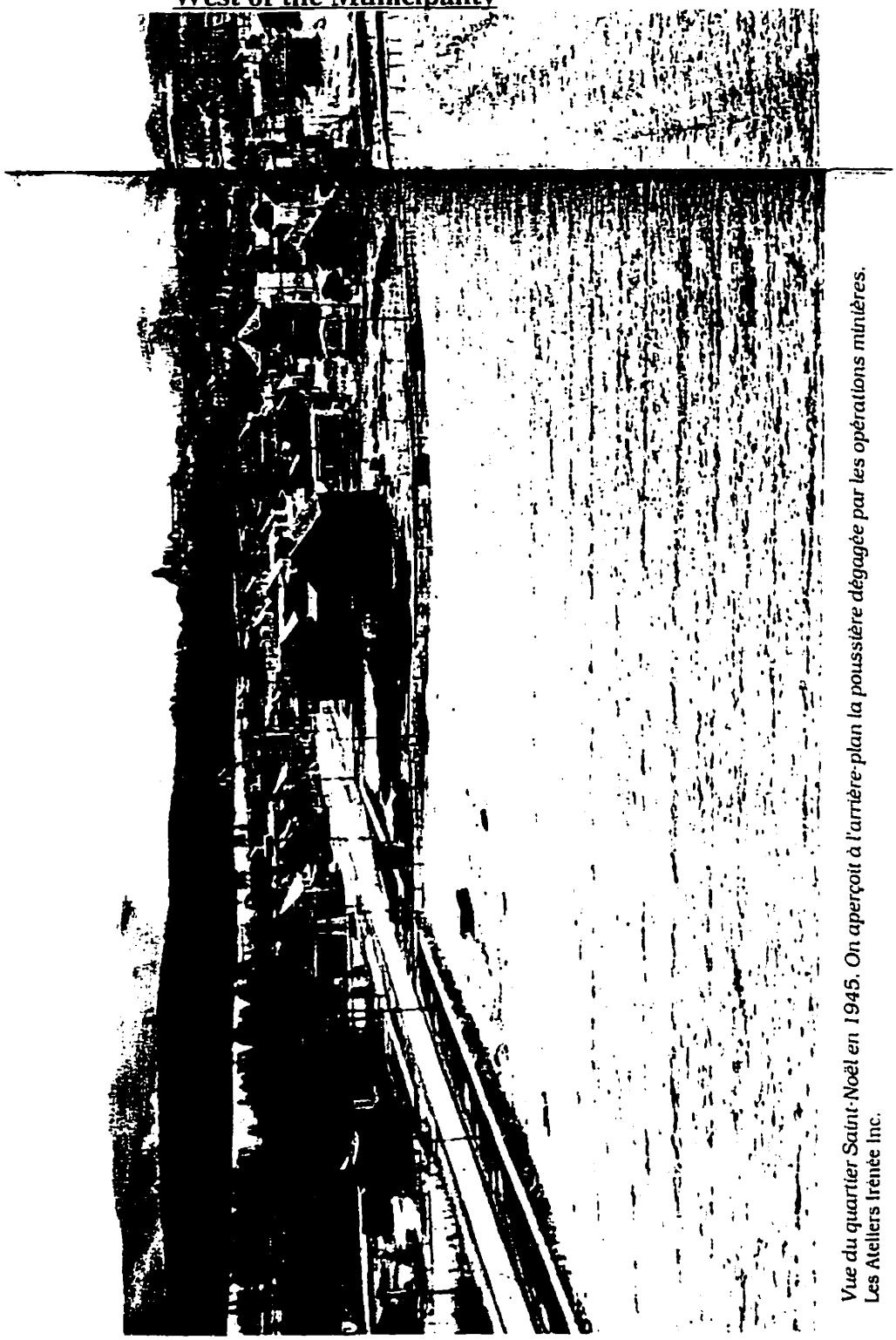
Figure B9-2 Asbestos Tailings Piles In the Backyards of Houses on Smith Street in 1910 and 1946 in Thetford Mines[1]

La rue Smith et le viaduc traversant la rivière Bécancour, vers 1910.
Geological Survey of Canada, GSC26218. Collection Musee minéralogique et minier de Thetford Mines.



[1] Reproduced from Cinq-Mars et al. [1994, pp.27, 263] with the permission of the municipality of Thetford Mines [Mr. Y. Faucher].

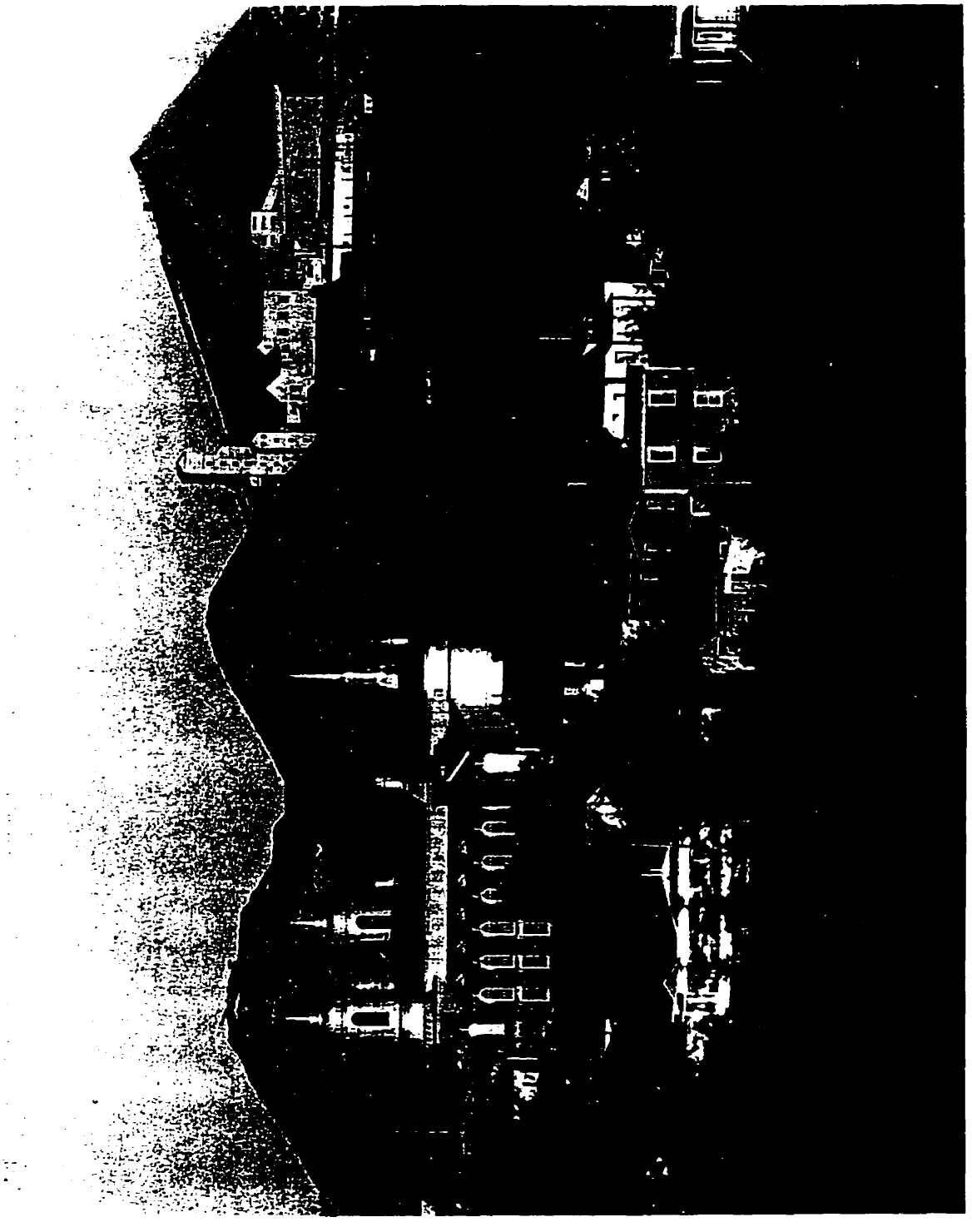
Figure B9-3 Thetford Mines' Eastern Saint-Noël Parish in 1945: Background Dust Emissions From Mining Operations West of the Municipality[1]



Vue du quartier Saint-Noël en 1945. On aperçoit à l'arrière-plan la poussière dégagée par les opérations minières.
Les Ateliers Irénée Inc.

[1] Reproduced from Cinq-Mars et al. [1994, p.212] with the permission of the municipality of Thetford Mines [Mr. Y. Faucher].

Figure B9-4 **Looking East of Thetford Mines in 1950: Old Saint-Maurice Parish, Tailings Piles, Dust Emissions From the Johnson Mine**[1]



Vue du quartier Saint-Maurice et de la mine Johnson vers 1950.
SAHRA, Collection Aline Visser. (Photo : Jacques Fugère.)

[1] Reproduced from Cinq-Mars et al. [1994, p.214] with the permission of the municipality of Thetford Mines [Mr. Y. Faucher].

Figure B9-5 Looking South-West of Thetford Mines in 1957: Dust Emissions From Mining Operations[1]

Vue de la ville de Thetford Mines vers le sud-ouest, en 1957.
Ville de Thetford Mines.

[1] Reproduced from Cinq-Mars et al. [1994, p.413] with the permission of the municipality of Thetford Mines [Mr. Y. Faucher].

Appendix B10 **Questionnaire on Residential and Household Exposure Histories and on Past Asbestos Pollution Sightings**

QUESTIONNAIRE SUR LA SANTÉ ET LES HABITUDES DE VIE

ÉTUDE ÉPIDÉMIOLOGIQUE
DES FEMMES
VIVANT DANS LES RÉGIONS
DE L'AMIANTE

INSTITUT ARMAND-FRAPPIER
Centre de recherche en épidémiologie
et médecine préventive

UNIVERSITÉ McGILL
Faculté de médecine

QUESTIONNAIRE SUR LA SANTÉ ET LES HABITUDES DE VIE

INSTRUCTIONS

Ce questionnaire comporte les sections suivantes:

- Renseignements généraux
- Histoire résidentielle
- Expérience de travail avec l'amiante
- Tabac
- Alcool
- Santé et utilisation des services de santé
- Histoire médicale familiale

Pour la majorité des questions, il suffit de cocher une case "❑" avec un "X" pour indiquer votre choix. Pour certaines questions, il faudra une réponse plus précise: l'âge, la date, l'adresse ou la profession, etc... Selon votre réponse à certaines questions, il vous faudra parfois sauter quelques questions et passer à une autre partie du questionnaire. Ceci vous fera épargner du temps en vous évitant la lecture de questions qui ne s'appliquent pas à vous.

Plusieurs questions demandent des renseignements précis sur des faits datant de plusieurs années. Si vous ne vous rappelez pas une information, sautez la question, peut-être que la réponse vous viendra plus tard. Par contre, une réponse approximative vaut mieux qu'aucun renseignement.

Si vous avez des questions concernant ce questionnaire, n'hésitez pas à contacter Mme Denise Bourbonnais à frais virés, au no. de téléphone:

(514) 687-5010 poste 201

NOUS VOUS REMERCIONS DE VOTRE PRÉCIEUSE COLLABORATION À CETTE ÉTUDE

RENSEIGNEMENTS GÉNÉRAUX

1. NOM _____

   PRÉNOM _____

2. NOM À LA NAISSANCE _____

3. TÉLÉPHONE _____

4. **Quelle est votre date de naissance?** _____/_____/_____
   Jour Mois Année

5. **a) Quel est votre lieu de naissance?**

   ☐ Canada ☐ autre _____ (précisez)

   → **Si vous êtes née au Canada, précisez la province:**

   ☐ Québec ☐ autre _____ (précisez)

   → **Si vous êtes née au Québec, précisez le comté:**

   ☐ Frontenac ☐ Richmond ☐ autre _____ (précisez)

   b) Dans quel village ou ville êtes-vous née? _____

   c) Nombre d'années vécues dans votre ville natale: _____

6. **Quel est votre état matrimonial?**

- ❑ Mariée ou vivant avec votre conjoint
- ❑ Séparée ou divorcée
- ❑ Veuve
- ❑ Célibataire

7. **Quelle est la langue que vous avez apprise en premier lieu dans votre enfance et que vous comprenez encore?**

❑ Français ❑ Anglais ❑ autre _____
(précisez)

8. **Dites-nous le plus haut niveau de scolarité que vous avez <u>complété</u>:**

- ❑ 3 années de scolarité ou moins
- ❑ 4 à 6 années de scolarité
- ❑ 7 à 9 années de scolarité
- ❑ Plus de 10 années de scolarité

9. **Travaillez-vous ou avez-vous déjà travaillé à l'extérieur de la maison?**

❑ Oui ❑ Non

↳ Si oui, précisez le nombre d'années: _____ ans

et votre principal emploi: _____

10. **Quelle a été le principal travail de votre conjoint au cours de sa vie?**

_____

HISTOIRE RÉSIDENTIELLE

11. Veuillez énumérer ci-dessous toutes les adresses où vous avez vécu pendant votre vie. Commencez par votre adresse à la naissance et remontez jusqu'à votre adresse actuelle.

Si vous ne vous souvenez pas d'une adresse exacte, donnez une indication approximative de l'endroit.

	LIEU DE RÉSIDENCE	ÂGE À L'ARRIVÉE	ÂGE AU DÉPART
1ère	Adresse _____ Paroisse ou quartier _____ Ville/Pays _____	de la naissance	à_____ans
2e	Adresse _____ Paroisse ou quartier _____ Ville/Pays _____	de_____ans	à_____ans
3e	Adresse _____ Paroisse ou quartier _____ Ville/Pays _____	de_____ans	à_____ans
4e	Adresse _____ Paroisse ou quartier _____ Ville/Pays _____	de_____ans	à_____ans
5e	Adresse _____ Paroisse ou quartier _____ Ville/Pays _____	de_____ans	à_____ans

	LIEU DE RÉSIDENCE	ÂGE À L'ARRIVÉE	ÂGE AU DÉPART
6e	Adresse _____	de_____ans	à_____ans
	Paroisse ou quartier _____		
	Ville/Pays _____		
7e	Adresse _____	de_____ans	à_____ans
	Paroisse ou quartier _____		
	Ville/Pays _____		
8e	Adresse _____	de_____ans	à_____ans
	Paroisse ou quartier _____		
	Ville/Pays _____		
9e	Adresse _____	de_____ans	à_____ans
	Paroisse ou quartier _____		
	Ville/Pays _____		
10e	Adresse _____	de_____ans	à_____ans
	Paroisse ou quartier _____		
	Ville/Pays _____		
11e.	Adresse _____	de_____ans	à_____ans
	Paroisse ou quartier _____		
	Ville/Pays _____		
12e	Adresse _____	de_____ans	à_____ans
	Paroisse ou quartier _____		
	Ville/Pays _____		

12. Nous aimerions avoir des renseignements sur la présence de poussière d'amiante près de chez vous. Veuillez répondre aux questions suivantes en indiquant la période pendant laquelle vous avez vu ces poussières d'amiante.

a) Avez-vous **déjà** vu de la poussière d'amiante par terre près de chez vous?

❑ OUI, de l'âge de _____ ans à l'âge de _____ ans. ❑ NON, JAMAIS

↳ SI OUI, pendant quelle période de votre vie en avez-vous vu **le plus**?
De l'âge de _____ ans à l'âge de _____ ans.

b) Y a-t-il eu une période de votre vie où vous voyiez de la poussière d'amiante par terre près de chez vous, **au printemps**, après la fonte des neiges?

❑ OUI, de l'âge de _____ ans à l'âge de _____ ans. ❑ NON, JAMAIS

c) Y a-t-il eu une période de votre vie où vous voyiez de la poussière d'amiante par terre près de chez vous **chaque semaine** ou presque?

❑ OUI, de l'âge de _____ ans à l'âge de _____ ans. ❑ NON, JAMAIS

d) Y a-t-il eu une période de votre vie où vous voyiez de la poussière d'amiante par terre près de chez vous **chaque jour** ou presque?

❑ OUI, de l'âge de _____ ans à l'âge de _____ ans. ❑ NON, JAMAIS

e) Y a-t-il eu une période de votre vie où vous pouviez voir, certains jours, les traces de pas dans la poussière d'amiante par terre devant la maison?

❑ OUI, de l'âge de _____ ans à l'âge de _____ ans. ❑ NON, JAMAIS

f) Y a-t-il eu une période de votre vie où il vous arrivait parfois de rentrer de promenade avec de la poussière d'amiante sur la tête, sur les épaules ou sur les vêtements?

❑ OUI, de l'âge de _____ ans à l'âge de _____ ans. ❑ NON, JAMAIS

g) Avez-vous déjà isolé vous-même des tuyaux, le fourneau ou les fenêtres du logement avec de l'amiante mouillée puis séchée?

❑ OUI, de l'âge de _____ ans à l'âge de _____ ans. ❑ NON, JAMAIS

h) Avez-vous déjà lavé ou épousseté des vêtements couverts de poussière d'amiante:

❑ OUI, de l'âge de _____ ans à l'âge de _____ ans. ❑ NON, JAMAIS

l) Avez-vous déjà demeuré à moins d'un mille d'un moulin produisant de l'amiante?

❑ OUI, de l'âge de _____ ans à l'âge de _____ ans. ❑ NON, JAMAIS

↳ Si OUI, quel était **le moulin le plus proche** de chez vous?

NOM du moulin: _____

Pouvez-vous préciser à quelle distance de cette mine vous demeuriez?

- ❑ 1 mille
- ❑ 1/2 mille (2,600 pieds)
- ❑ 1/4 mille (1,300 pieds)
- ❑ 1/10 mille (500 pieds)
- ❑ 1/20 mille (250 pieds)

Quel vent apportait le plus de poussière d'amiante dans ou près de **votre** maison?

- ❑ vent du nord
- ❑ vent d'ouest
- ❑ vent du sud
- ❑ vent d'est
- ❑ NE SAIS PAS

13. Est-ce que l'agrandissement du puits de la mine vous a déjà forcée à déménager?

oui ❑ non ❑

↳ Si oui, **quel âge aviez-vous lors de ce déménagement?** _____ans

EXPÉRIENCE DE TRAVAIL AVEC L'AMIANTE

14. Avez-vous déjà travaillé dans l'industrie de l'amiante ou manipulé de l'amiante dans votre travail?

oui ❑ non ❑

↳ Si oui, **quel était votre emploi?** _____

Pour quelle compagnie? _____

Pendant combien d'années? _____ ans

Âge au début: _____ ans

15. Avez-vous déjà réparé des sacs de jute ou effectué un autre travail à la maison pour une compagnie rattachée à l'industrie de l'amiante?

oui ❑ non ❑

↳ Si oui, **quel travail?** _____

Pendant combien d'années? _____ ans

Âge au début: _____ ans

16. Avez-vous déjà demeuré avec une personne (parent, conjoint, enfant ou autre) qui a travaillé pour l'industrie de l'amiante?

oui ❑ non ❑ ⟶ Si non, passez directement à la Question 17

⟶ Si oui: répondez aux questions suivantes pour chacune de ces personnes. Les questions sont répétées 8 fois au cas où vous auriez demeuré avec plusieurs personnes ayant travaillé dans l'amiante

1ère personne

Nom: _____ **Lien de parenté:** _____

Année de naissance _____

Pendant quelle période de votre vie avez-vous demeuré avec cette personne?

de: _____ ans à: _____ ans

Quel était son emploi? _____

À quel âge a-t-elle commencé? _____ans

Pendant combien d'années cette personne a-t-elle travaillé dans l'amiante?_____ans

Ses vêtements étaient-ils couverts de poussière d'amiante lorsqu'elle rentrait de son travail à la maison?

❑ Oui, régulièrement ❑ Oui, parfois ❑ Jamais

2e personne

Nom: _____ **Lien de parenté:** _____

Année de naissance _____

Pendant quelle période de votre vie avez-vous demeuré avec cette personne?

de: _____ ans à: _____ ans

Quel était son emploi? _____

À quel âge a-t-elle commencé? _____ans

Pendant combien d'années cette personne a-t-elle travaillé dans l'amiante?_____ans

Ses vêtements étaient-ils couverts de poussière d'amiante lorsqu'elle rentrait de son travail à la maison?

❑ Oui, régulièrement ❑ Oui, parfois ❑ Jamais

3e personne

Nom: _____ **Lien de parenté:** _____

Année de naissance _____

Pendant quelle période de votre vie avez-vous demeuré avec cette personne?
de: _____ ans à: _____ ans

Quel était son emploi? _____

À quel âge a-t-elle commencé? _____ans

Pendant combien d'années cette personne a-t-elle travaillé dans l'amiante?_____ans

Ses vêtements étaient-ils couverts de poussière d'amiante lorsqu'elle rentrait de son travail à la maison?

❑ Oui, régulièrement ❑ Oui, parfois ❑ Jamais

4e personne

Nom: _____ **Lien de parenté:** _____

Année de naissance _____

Pendant quelle période de votre vie avez-vous demeuré avec cette personne?
de: _____ ans à: _____ ans

Quel était son emploi? _____

À quel âge a-t-elle commencé? _____ans

Pendant combien d'années cette personne a-t-elle travaillé dans l'amiante?_____ans

Ses vêtements étaient-ils couverts de poussière d'amiante lorsqu'elle rentrait de son travail à la maison?

❑ Oui, régulièrement ❑ Oui, parfois ❑ Jamais

5e personne

Nom: _____ **Lien de parenté:** _____

Année de naissance _____

Pendant quelle période de votre vie avez-vous demeuré avec cette personne?
de: _____ ans à: _____ ans

Quel était son emploi? _____

À quel âge a-t-elle commencé? _____ans

Pendant combien d'années cette personne a-t-elle travaillé dans l'amiante?____ans

Ses vêtements étaient-ils couverts de poussière d'amiante lorsqu'elle rentrait de son travail à la maison?

❑ Oui, régulièrement ❑ Oui, parfois ❑ Jamais

6e personne

Nom: _____ **Lien de parenté:** _____

Année de naissance _____

Pendant quelle période de votre vie avez-vous demeuré avec cette personne?
de: _____ ans à: _____ ans

Quel était son emploi? _____

À quel âge a-t-elle commencé? _____ans

Pendant combien d'années cette personne a-t-elle travaillé dans l'amiante?____ans

Ses vêtements étaient-ils couverts de poussière d'amiante lorsqu'elle rentrait de son travail à la maison?

❑ Oui, régulièrement ❑ Oui, parfois ❑ Jamais

7e personne

Nom: _____ **Lien de parenté:** _____

Année de naissance _____

Pendant quelle période de votre vie avez-vous demeuré avec cette personne?
de: _____ ans à: _____ ans

Quel était son emploi? _____

À quel âge a-t-elle commencé? _____ans

Pendant combien d'années cette personne a-t-elle travaillé dans l'amiante?_____ans

Ses vêtements étaient-ils couverts de poussière d'amiante lorsqu'elle rentrait de son travail à la maison?

❑ Oui, régulièrement ❑ Oui, parfois ❑ Jamais

8e personne

Nom: _____ **Lien de parenté:** _____

Année de naissance _____

Pendant quelle période de votre vie avez-vous demeuré avec cette personne?
de: _____ ans à: _____ ans

Quel était son emploi? _____

À quel âge a-t-elle commencé? _____ans

Pendant combien d'années cette personne a-t-elle travaillé dans l'amiante?_____ans

Ses vêtements étaient-ils couverts de poussière d'amiante lorsqu'elle rentrait de son travail à la maison?

❑ Oui, régulièrement ❑ Oui, parfois ❑ Jamais

TABAC

17. Présentement faites-vous usage de:

	OUI régulièrement	OUI à l'occasion	NON jamais
a) la cigarette	❑	❑	❑
b) la pipe ou le cigare	❑	❑	❑
c) le tabac à priser ou à chiquer	❑	❑	❑

Si vous avez répondu non aux questions ci-dessus, passez à la question 21

18. Fumez-vous la cigarette tous les jours?

oui ❑ non ❑ ⟶ Si non, passez directement à la Question 21.

19. A quel âge avez-vous commencé à fumer la cigarette tous les jours?

_____ ans

20. Actuellement, environ combien de cigarettes fumez-vous par jour?

_____ cigarettes

SI VOUS NE FUMEZ PAS LA CIGARETTE TOUS LES JOURS

21. Comment décrivez-vous votre expérience de la cigarette?

- ❑ Je n'ai jamais fumé
- ❑ J'ai déjà fumé à l'occasion
- ❑ J'ai déjà fumé tous les jours

SI VOUS AVEZ DÉJÀ FUMÉ TOUS LES JOURS

22. À quel âge avez-vous commencé à fumer la cigarette tous les jours?

_____ ans

À quel âge avez-vous cessé de fumer la cigarette tous les jours?

_____ ans

Environ combien de cigarettes fumiez-vous habituellement par jour?

_____ cigarettes

ALCOOL

23. Avez-vous déjà consommé de la bière, du vin, des liqueurs fortes ou d'autres boissons alcoolisées?

oui ❑ non ❑ ⟶ Si non, passez directement à la Question 27.

24. Sans compter les fois où vous avez seulement goûté, à quel âge avez-vous commencé à consommer des boissons alcoolisées?

_____ ans

25. Au cours des 12 derniers mois avez-vous consommé de la bière, du vin, des liqueurs fortes ou d'autres boissons alcoolisées?

oui ❑ non ❑ ⟶ Si non, passez directement à la Question 27.

26. Au cours des 12 derniers mois, quelle a été la fréquence moyenne de votre consommation de boissons alcoolisées?

- ❑ 4 fois ou plus par semaine
- ❑ 1 à 3 fois par semaine
- ❑ Une ou deux fois par mois
- ❑ Moins d'une fois par mois

SANTÉ ET UTILISATION DES SERVICES DE SANTÉ

27. Quel est votre grandeur? ____ pi ____ po OU _____ m ____ cm

Quel est votre poids? ____ livres OU _____ kg

28. Au cours des 12 derniers mois, avez-vous été gravement malade?

oui ❑ non ❑

29. Au cours des 12 derniers mois, avez-vous été hospitalisée?

oui ❑ non ❑

↳ Si oui, **à quel hôpital?**_____

dans quelle ville? _____

30. Au cours des 12 derniers mois, vous êtes-vous adressée aux personnes suivantes au sujet de votre santé?

médecin généraliste oui ❑ non ❑ ville _____

médecin spécialiste oui ❑ non ❑ ville _____

31. Au cours des 2 dernières semaines, vous êtes-vous adressée aux personnes suivantes au sujet de votre santé?

médecin généraliste oui ❑ non ❑ ville _____

médecin spécialiste oui ❑ non ❑ ville _____

32. Au cours des 2 dernières semaines, avez-vous dû limiter vos activités pour des raisons de santé?

oui ❑ non ❑

└► Si oui, **pendant combien de jours?**_____ jours

33. Quand avez-vous eu un prélèvement vaginal ("Pap Test") la dernière fois?

- ❑ Il y a moins de 12 mois
- ❑ Il y a de 1 à 2 ans
- ❑ Il y a plus de 2 ans
- ❑ Jamais
- ❑ Ne sais pas

34. Avez-vous déjà eu un cancer du sein?

oui ❑ non ❑

└► Si oui, **à quel hôpital avez-vous été traitée?** _____

dans quelle ville? _____

HISTOIRE MÉDICALE FAMILIALE

35. Avez-vous déjà eu les problèmes de santé suivants?

	Oui	Non	Ne sais pas
Trouble cardiaque	❑	❑	❑
Hypertension (haute pression)	❑	❑	❑
Diabète	❑	❑	❑
Problèmes de poumons	❑	❑	❑
Cancer de poumon	❑	❑	❑
Cancer: autre que poumon	❑	❑	❑
Problèmes d'articulations (jointures)	❑	❑	❑

Si vous avez éprouvé un ou plusieurs de ces problèmes, à quels hôpitaux êtes-vous allée?

Hôpital _____ Ville _____

Hôpital _____ Ville _____

Hôpital _____ Ville _____

Hôpital _____ Ville _____

Hôpital _____ Ville _____

36. Est-ce que votre <u>conjoint</u> a déjà eu les problèmes de santé suivants? (Nous avons besoin de ces informations même si votre mari est décédé.)

	Oui	Non	Ne sais pas
Trouble cardiaque	❑	❑	❑
Hypertension (haute pression)	❑	❑	❑
Diabète	❑	❑	❑
Problèmes de poumons	❑	❑	❑
Cancer du poumon	❑	❑	❑
Cancer: autre que poumon	❑	❑	❑
Problèmes d'articulations (jointures)	❑	❑	❑

Si votre conjoint a éprouvé un ou plusieurs de ces problèmes, à quels hôpitaux est-il allé?

Hôpital _____ Ville _____

Hôpital _____ Ville _____

Hôpital _____ Ville _____

Hôpital _____ Ville _____

Hôpital _____ Ville _____

Le questionnaire que vous venez de compléter concerne:

Mme _____

et la personne qui l'a rempli se nomme:

Mme ou M. _____

Auriez-vous l'obligeance de nous laisser votre numéro de téléphone au cas où nous aurions besoin de clarifier certaines questions.

(____)_____
Numéro de téléphone

Nous vous remercions de votre précieuse collaboration.
Si vous avez des questions, n'hésitez pas à contacter
Mme Denise Bourbonnais à frais virés au numéro de téléphone suivant
(514) 687-5010 poste 201

Appendix B11	Past Visible Asbestos Pollution Recalled by Residents: PY-Percentage Frequency of Responses, by Town and Era

Table B11-1 Percent of Person-Years for Which Respondents Reported Seeing Asbestos Depositions, by Town and Era

	Era			
Type of Deposit / Town	**1920-49**	**1950-69**	**1970-79**	**1980-89**
Some[1] Deposits	**73**	**39**	**7**	**2**
Asbestos	82	35	3	2
Black Lake	73	52	12	3
Thetford Mines	69	38	7	2
Weekly deposits	**68**	**34**	**4**	**0**
Asbestos	74	29	2	0
Black Lake	69	44	6	0
Thetford Mines	65	34	4	0
Daily deposits	**60**	**30**	**4**	**0**
Asbestos	70	25	2	0
Black Lake	70	40	5	0
Thetford Mines	53	30	4	0
Footprints	**43**	**19**	**1**	**0**
Asbestos	52	18	0	0
Black Lake	53	26	0	0
Thetford Mines	37	18	2	0
Head+shoulders	**39**	**14**	**0**	**0**
Asbestos	39	14	0	0
Black Lake	54	22	0	0
Thetford Mines	36	13	0	0

[1] "Some deposits" is a short-cut formulation for "ever seen dust deposits".

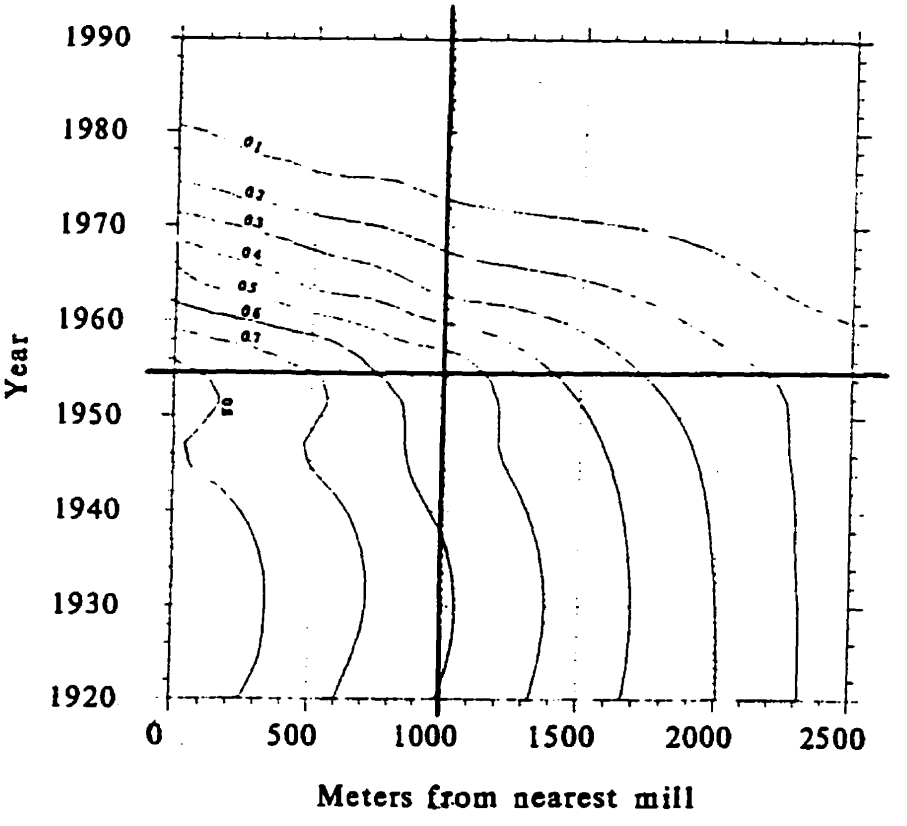
Table B11-2 Cumulative Frequency Distribution of Distance From Nearest Mill by Town Before 1950 (N=202, PY=5982)

Distance [1]	Asbestos (cum. %)	Black Lake (cum. %)	Thetford Mines (cum. %)	TOTAL (cum. %)	Denomin. PY (N)
> 1600 m	14.6	16.0	33.6	25.1	409 (50)
≈ 1600 m	22.2	21.4	44.4	34.1	599 (19)
≈ 800 m	42.3	48.1	61.9	53.8	1757 (39)
≈ 400 m	86.9	85.9	80.3	83.2	1180 (59)
≈ 150 m	97.2	91.7	91.4	93.2	537 (20)
≈ 75 m	100.0	100.0	100.0	100.0	1500 (15)
Average distance [2]	649 m	676 m	1063 m	908 m	

1 Respondents' subjective appreciations of distance between home and the nearest mill. Neighbourhood by neighbourhood comparisons showed very good agreement between the respondents' subjective appreciations and map measurements of distances between neighbourhoods and nearest mills.

2 To compute PY-weighted averages, the "> 1600 m" category was assigned an arbitrary value of 2400 m. The last value may be overestimated by 400 m, in which case average distances would be overestimated by some 60 m in Asbestos and Black Lake and 130 m in Thetford.

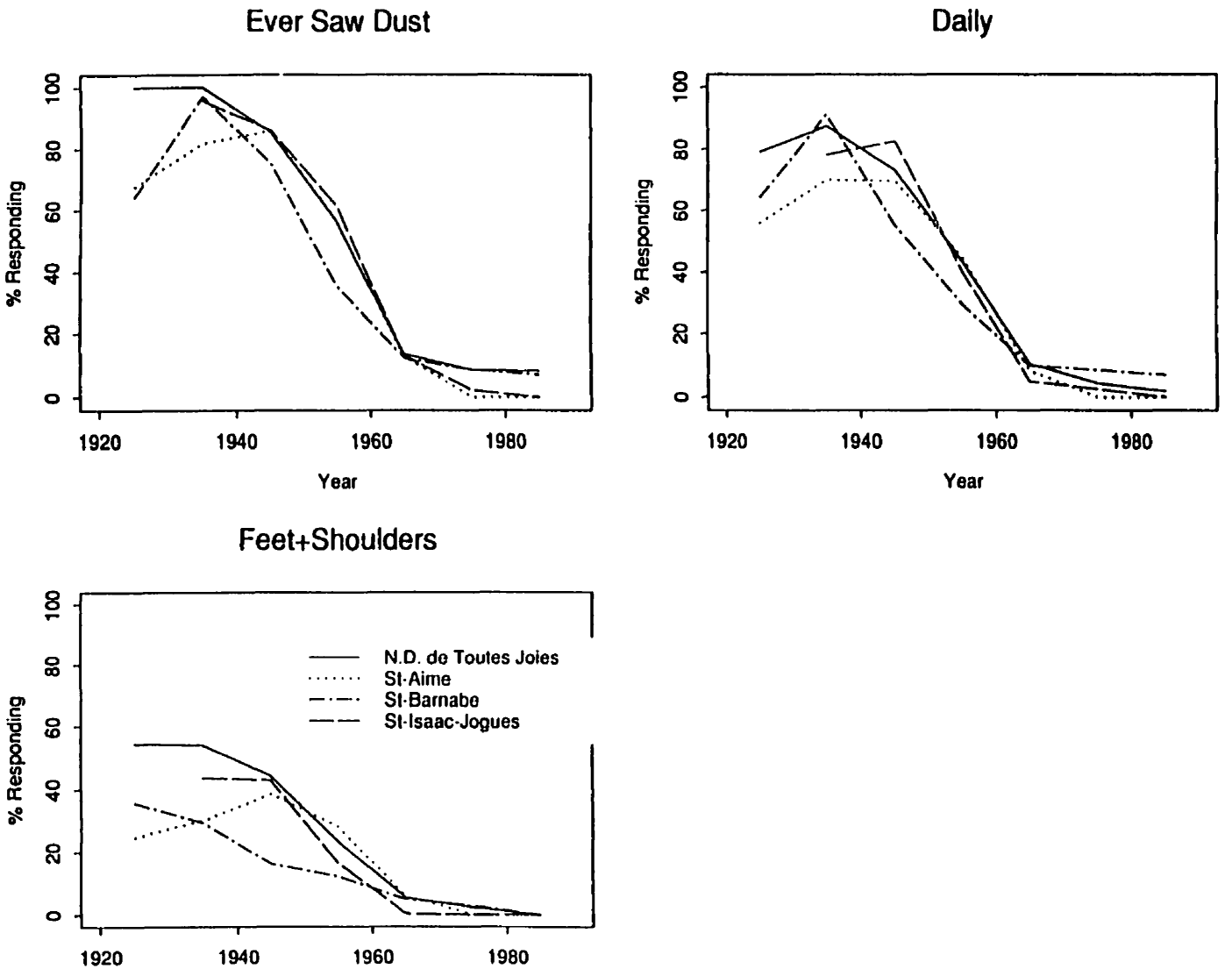
Figure B11-1 PY-Proportion[1] of Weekly Deposit Sightings, by Year and Reported Distance From Nearest Mill, for all Respondents Combined



[1] These proportions are represented by the isolines and are indicated as "0.1" to "0.8" on this graph.

Appendix B12 Dustiness Indices by Neighbourhood and Year in Asbestos

Figure B12-1 Dustiness Indices by Neighbourhood and Year in Asbestos



Appendix B13 Correlation Between Past Exposures and Lung Burden Biomarkers

In the 89 Thetford miners and millers, certain lung burden variables correlated better with exposure determinants than others and logarithmic transformations improved the correlations, as indicated in Table B13-1. This correlation table is composed of two separate triangular correlation matrices. The lower triangle pertains to the untransformed variables and the upper triangle to the exposure logarithms. Bold characters represent correlations with log(tremolite burden), and italics represent correlations with log(asbestos bodies).

On the natural untransformed scale, no lung burden variable correlated well with duration of exposure (years exposed), exposure intensity (mpcf), cumulative exposure (mpcf.y) duration or cessation. The fact that the three biomarkers were correlated on the logarithmic scale but not on the natural scale suggests proportional or geometric measurement errors of lung burden. Of the three lung burden biomarkers, only tremolite correlated meaningfully on the logarithmic scale with past exposure variables; it was correlated positively with *mpcf, mpcf.y* and *exposure duration*, and negatively albeit weakly with *cessation* (time since last exposure). Hence tremolite burden appears to be the most reliable and congruous biomarker on the individual observation level, and logarithmic transformations are necessary to attenuate the effect of geometric measurement error and to bring out underlying data patterns and relations.

Table B13-1 Pearson Correlations Between Lung Burden and Various Exposure Variables

	log(trem) (n=88)	log(chry) (n=89)	*log(AB) (n=79)*	log(mpcf)	log(mpcf.y)	exposure duration	exposure cessation	age
tremolite		**.40**	***.51***	**.42**	**.50**	**.41**	**-.20**	**.07**
chrysotile asbestos bodies	.23		*.35*	.01	.11	.14	-.21	-.13
mpcf	.04	.24		*.15*	*.14*	*.07*	*-.08*	*-.10*
mpcf.y	.07	.00	-.04		.87	**.33**	.02	-.50
years exposed	.10	-.03	-.04	.89		**.74**	**-.51**	.10
years since last exposure	.13	-.03	-.03	.22	.47		-.72	.19
age	-.07	-.07	-.14	-.03	-.22	-.72		.33
	.08	-.14	-.29	.12	.23	.19	.33	

Appendix B14	Construct Validity of the Biokinetic Model With Respect to Two Stratified Occupational Datasets

Methods

The biokinetic model has not been used by other epidemiologists or toxicologists except for Berry et al.[Berry et al., 1989], and nonlinear regression is not a familiar technique in epidemiology. For those reasons, and because the biokinetic model was fitted to a small and heterogeneous dataset (large inter-individual variability) of 72 asbestos workers, some reassurance about the validity of the model and regression was sought. It was not possible to evaluate the predictive validity of the biokinetic modeling in the absence of other estimates of past exposure levels. However, on the basis of accepted knowledge and notions (constructs), the biokinetic model should obtain predictable differences when fitted on chrysotile vs. tremolite data and when comparing occupational groups with very different non-occupational exposures.

Two such validations of accepted ideas were sought to validate the biokinetic model and the lung burden data. First, when the biokinetic model is applied to chrysotile burden, the estimated clearance rate K_e should be slower than for tremolite, while the "deposition rate" K_d relative to mpcf exposure intensity measurements should be higher for chrysotile than for tremolite since there was much more airborne chrysotile than tremolite in Thetford and Black Lake chrysotile mines and mills. Second, exposure and burden data stratified on exposure duration and cessation[1] were used to compare two cohorts of asbestos workers (Thetford and Charleston)[Sébastien et al., 1989] to see if the differences between chrysotile and tremolite deposition and clearance rates were observable in a different occupational cohort (Charleston) and if a substantial non-occupational exposure estimated by the model would be observed only in the cohort which lived in a visibly asbestos polluted area (Thetford). Simultaneously, stratification should reduce data-fitting problems and biases introduced by individual random

[1] Cessation is defined as the time since last occupational exposure to asbestos.

measurement errors, since stratified data points are average values of individual observations in each given stratum.

Results

Analyses were conducted on the published[Sébastien et al., 1989] aggregate data from Charleston textile workers (n=66) and Thetford Mines miners and millers (n=70) which were stratified in 14 exposure cessation time and duration strata (100-months categories) which included at least one (1 to 17) representative of each cohort. As reported by Sébastien and coworkers, both cohorts worked with asbestos from the Thetford area but the distributions of exposure variables and covariates differed significantly between the two cohorts. Thetford necropsied cases were 10 years older at death (68 years of age), had been employed 30 years more (36 years total) than Charleston workers (6.4 years), and had been exposed to median dust exposure intensities (10.9 mpcf) 8 to 9 times higher than Charleston workers (1.4 mpcf). Conversely, Charleston workers had ceased working in the asbestos textile plant (cessation) much longer before death (20 years) than Thetford workers had (8 years median).

Table B14-1 shows the results of the application of the biokinetic model to the stratified data for the two cohorts. The observation strata were weighted by the number of workers in each cell. The regression sums of squares and the precision of parameter estimates were evidently inflated by the ecological nature of the data and the weighting. The table suggests that the biokinetic model provides results consistent with accepted knowledge in both cohorts, since the estimated clearance rate for chrysotile was important whereas it was null that for tremolite, and chrysotile burden related less (R^2) to exposure measurements and circumstances (duration, cessation) than did tremolite. In both cohorts, the K_d concentration-deposition-conversion factor was much higher for chrysotile than for tremolite, reflecting the much larger proportion of chrysotile (>100:1) in the respirable aerosols. Only in Thetford where past air pollution was visible was a significant non-occupational exposure intensity estimated (3 mpcf).

Table B14-1 Fitted Biokinetic Models on the Thetford and Charleston Stratified Data

Fibre type	Parameter / Statistic	Thetford Cohort Estimate (n=70)	Charleston Cohort Estimate (n=65)
Tremolite	R^2	.89	.93
	K_{dt}	0.05	4.10
	K_{et} (%)	0.0	0.0
	Non-occ. mpcf	3.0	0.0
Chrysotile	R^2	.42	.81
	K_{dc}	2.2	71.5
	K_{ec} (%)	7.2	49.0
	Non-occ. mpcf	0.0	0.0

Appendix B15	Lung Burden Data: Description and Statistics

Occupational Dataset

Sébastien et al.[Sébastien et al., 1989] analyzed the retained fibres in lung tissue specimens from two cohorts of chrysotile workers: 89 miners and millers from Thetford Mines and 72 textile workers from Charleston, South Carolina. The latter handled chrysotile asbestos originating from the Thetford area. The 161 lung tissue specimens (74 formalin fixed and 87 paraffin blocks) were analyzed by transmission electron microscopy and energy dispersive spectrometry. Only fibres longer than 5 μm were counted with an electronic microscope (TEM). Asbestos bodies were counted with a phase-contrast optical microscope (PCOM). Individual lifetime average *occupational* total dust exposures[1], age at death, duration and cessation periods were available from two previous cohort studies[Dement et al., 1983; McDonald et al., 1993a; McDonald et al., 1993b; Dement et al., 1994]. Although stratified aggregate data[Sébastien et al., 1989] of both occupational datasets were used to partially validate the biokinetic model , only the individual Thetford Mines occupational data were used in the three aforementioned methods of estimating the relation between lung burden and past average exposure intensity.

Extrapolating the biokinetic relation from the occupational to the target non-occupational groups was moot because the groups were extremely different in crucial respects. The asbestos workers were male smokers exposed intermittently after the age of 20 to extremely high asbestos levels, whereas the non-occupational target groups were essentially female non-smokers exposed continuously since birth to asbestos levels 10-1000 times lower than in the workplace. Smoking, asbestos exposure, exposure time pattern, and the level of exposure all interact among them and affect pulmonary deposition and clearance. Such complex interactions reduce external validity and generalizability of a statistically fitted relation, even

[1] In the past, total respirable dusts were sampled by midget samplers and counted with an optical microscope.

more so when exposure measurement errors[Armstrong, 1983; Doll and Peto, 1986; Armstrong, 1990] and biokinetics are nonlinear[Vincent and Donaldson, 1990; Vacek and McDonald, 1991]. Moreover, the occupational data comprised outliers on all variables used in the model, making the model-fitting very unsteady and strongly influenced by individuals least comparable with the target group. For example, the median tremolite burden of workers was 16 times higher than the median of the non-occupational group, but the highest occupational value was 600 times higher than the non-occupational median.

To reduce the gap between the occupational "study" group and the non-occupational "target" group (female residents), 14 observations were excluded *a priori*[1]. Regarding tremolite burden, one (1) missing value and the four (4) highest and very extreme values were excluded. Six (6) workers who were exposed less than two years and one (1) other who had ceased his work in the asbestos industry 47 years before death were excluded. One (1) worker with the highest chrysotile burden and one (1) with the highest AB burden were also excluded. In addition, three (3) other workers were excluded *a posteriori*. With their extreme mpcf/tremolite-burden ratios (9, 29 and 47 vs. a median of 0.44) and mpcf.y/tremolite ratios (220, 1110 and 1589 vs. a median of 12), these three outliers had very heavy leverages since their lung burdens were orders of magnitude lower than expected relative to their exposures. After exclusions, only 72 Thetford workers were thus selected to estimate past ambient exposure levels in the Thetford Mines area.

[1] All 14 excluded subjects had very high leverage values in linear and log-linear regressions when they were included.

Non-Occupational Dataset

The *non-occupational lung burden* dataset analyzed hereafter consisted of 51 cases collected and analyzed by Dr. B. Case in a previous study[Case and Sébastien, 1989] designed to compare the lung burdens of neighbourhood, household-contact and background exposed persons. For the three groups to be comparable, autopsies were selected in the same pathology department of the *Hôpital général de la Région de l'Amiante* in Thetford Mines from January 1976 to December 1981. Detailed occupational histories were obtained using company records and the hospital record. Fifty-one (51) cases remained after excluding individuals with any history of work in asbestos mines or mills or any related industry. Occupational and socio-demographic data, and residential and household exposure history were obtained from a next-of-kin. There were 22 neighbourhood-only exposed subjects: residents of Thetford Mines or Black Lake having lived less than 10 km from an asbestos mine or mill for more than 20 of the last 30 years of their life. There were 10 household contacts of asbestos workers: residents who ever lived with a father, mother or spouse who worked in the mines or mills for more than one year. Finally, there were 18 "referents": persons who had lived more than 10 km from all mines and mills for more than 20 of the last 30 years of their life and had never lived with an asbestos worker. One of the 51 subjects available did not quite satisfy the eligibility criteria for any group and was dropped from all analyses.

Comparing the three non-occupational exposure groups, the investigators found that lung burden for household contacts was on average 5 to 10 times higher than in environmentally exposed subjects without any household exposure, and they inferred that the cumulative exposure to asbestos must have been similarly 5 to 10 times higher among subjects who had experienced both household-contact and neighbourhood exposure than among those who had

experienced only the latter. I re-analyzed the data with the objective of estimating absolute as well as relative exposure levels.

Descriptive Data

Table B15-1 compares the occupational, household-contact, neighbourhood-only and referent groups on the main exposure variables and cofactors. Median values were used rather than geometric means because the distributions were not all skewed positively.

The asbestos workers were male (100%) and mostly smokers (85%) whereas the non-occupational groups and the referents were mostly female (80% and 56% respectively) and non-smokers (75%). Lung burdens were highest in the occupational group, then in the household-contact group, followed by the neighbourhood-only group, and were lowest in the referent group. Asbestos bodies had the largest and most regular gradient from one group to the next whereas chrysotile had the smallest discriminant power. The within-group variations (ranges) relative to the medians were highest for chrysotile and asbestos bodies. The household-contact group lived in the area for a longer period than the neighbourhood-only exposed group, and these non-occupational exposure durations exceeded the occupational exposure period of workers. The cessation period did not differ between groups, but there was much more inter-individual variation among workers than among other residents.

Table B15-1 Occupational and Non-Occupational Datasets

Lung Burden, Exposure and Other Variables	Occupational[1] Study Groups			Non-Occupational[2] Target Groups		
	Unselected Workers	Selected Workers	"Least Exposed" Workers [3]	Household Contacts [4]	Neighbour-hood	Unexposed "Referents"
n =	89	72	10	10	22	18
Tremolite f/µg						
median	26.2	26.9	2.8	1.6	0.3	0.03
range	2732.9	105.4	4.6	4.8	4.2	0.3
Chrysotile f/µg						
median	6.5	6.9	2.7	1.6	0.3	0.03
range	1,075.0	84.0	19.6	12.6	2.0	0.3
AB/mg						
median	213	218	22	4.6	2.0	0.08
range	9421	7920	166	21.3	6.2	6.0
mpcf						
median	10.9	10.9	2.0			
range	123	123	5.2			
mpcf.y						
median	342	393	59.2			
range	3606	3604	211.9			
Duration (years)	...in industry			...in asbestos area		
median	36.0	37.3	28.7	61.5	40.0	0.0
range	52	49	44	47	64	6
Cessation (years)	...last work in industry			...last residence in area		
median	8.0	7.0	6.5	< 15	< 9	70
range	61.0	35.0	30.0	15	9	6
Age at death						
median	68	69	66	62	73	70
range	29	28	21	42	57	55
Smokers %	69%	74%	75%	? %	? %	? %
Female %	0%	0%	0%	83%	78%	56%
Distance from mine (median)	---	---	---	1.6 km	4.2 km	>10 km

1 Data previously reported in [Sébastien et al., 1989]

2 Data previously reported in [Case and Sébastien, 1989; Case et al., 1993].

3 Workers with tremolite< 6 f/µg were selected as most comparable with non-occupational subjects.

4 According to our survey, female household contacts were so exposed for about half of the years lived in the area. However this was not recorded in the lung burden data, so "exposure duration" here is the number of years *lived in the area.*

Appendix B16 Intrinsically Linear Regression Models

The various models are shown in Table 16-1 with the explained variance proportions (R^2) of the logarithmic dependent variables in the occupational datasets, and in the neighbourhood-only, household-contact and referent groups.

The eleven "intrinsically linear" models fitted by stepwise multiple regression and based on tremolite burden gave very different exposure estimates in the small non-occupational samples. The linear model predicted negative exposure values and was thus discarded. The models (#7-#10) that respected the causal time sequence from exposure to lung burden projected unreasonably low exposures and there estimates were not heeded; for instance the historical estimates for the environmental group were one order of magnitude lower than levels measured even as late as in 1984. The "predictive" models (#1-#6) provided estimates that spread over a range one order of magnitude wide. Projections based on the whole group of 72 workers were somewhat higher than those based on the 36 least exposed workers.

Table B16-1 Intrinsically Linear Regression Models

Non-Occupational Groups

Note: In the last three columns, italics represent exposure intensity estimates (*mpcf*) whereas plain characters represent cumulative exposure estimates (mpcf.y).

#	Fitted Model	N (sample on which the model was fitted)	R^2 log. scale	Household-contact Estimated mpcf.y / *mpcf* (n=10)	Neighbour.-only Estimated mpcf.y / *mpcf* (n=22)	Referent Estimated mpcf.y / *mpcf* (n=18)
1:	$mpcf.y = 21.3 * (trem^{.89})$	72	.37	33.1 *0.15*	7.8 *0.04*	.046 *.0003*
2:	$mpcf.y = 14.1 * (trem^{.73})$	36 (mpcf<10.9)	.38	20.2 *0.10*	6.2 *0.03*	.091 *.0007*
3:	$mpcf.y = 4.74 * (trem^{.62}) * (1.07^{duration})$	72	.57	305.8 *1.29*	111.6 *0.41*	.082 *.0001*
4:	$mpcf.y = 4.86 * (trem^{.47}) * (1.06^{duration})$	36 (mpcf<10.9)	.61	175.7 *0.72*	73.4 *0.31*	.239 *.0014*
5:	$mpcf = 0.706 * (trem^{.61}) * (1.02^{duration})$	72	.38	160.0 *0.71*	47.8 *0.26*	.85 *.0026*
6:	$mpcf = 0.782 * (trem^{.44}) * (1.01^{duration})$	36 (mpcf<10.9)	.34	94.2 *0.43*	39.4 *0.20*	3.05 *.0093*
7:	$trem = 1.85 * (mpcf.y)^{.42}$	72	.37	.8 *.003*	.02 *.000*	.0000 *.00000*
8:	$trem = 1.21 * (mpcf.y)^{.52}$	36 (mpcf<10.9)	.38	1.8 *.007*	.08 *.000*	.0000 *.00000*
9:	$trem = 4.35 * (mpcf)^{.43} * (1.02^{duration})$	72	.37	.4 *.002*	.007 *.000*	.0000 *.00000*
10:	$trem = 3.26 * (mpcf)^{.56} * (1.02^{duration})$	36 (mpcf<10.9)	.38	2.1 *.009*	.10 *.000*	.0000 *.00000*
11:	trem = 4.08 + .054 * mpcf.y [Sébastien et al., 1986]	39 (cessation <75 months)	.21	-45.2 *-.18*	-69.5 *-0.27*	-75.5 *-0.25*

The simplest models (#1 and #2) gave the lowest estimates and were the only credible models. Models #3 to #6 gave much higher estimates; however, models #3 and #4 were not reliable despite their elevated R^2 (0.57-0.61) because this R^2 was an artifact due to the inclusion of duration on both sides of the equation, and because this oddity may well have biased the parameter estimates. Although models #5 and #6 should be more reliable because they sought to explain the variance of the main unknown component of the study population's cumulative exposure, past exposure intensity, their form was illogical and their application to non-occupationally exposed groups was inappropriate. In effect, these models implied that lung burden should be inversely proportional to exposure duration for a given past exposure intensity, an unacceptable incongruity.

Finally, the predicted values of models #1 and #2 were retained as the best log-linear estimates: 0.10-0.15 mpcf for household-contact exposed residents and 0.03-0.04 mpcf for neighbourhood-only exposed residents. Still, their statistical fit (R^2<40%) was weak and their geometric 95%CI had a five-fold range. No other log-linear regression model had higher adjusted R^2 than the above models without running into serious multicolinearity.

Appendix B17 Nonlinear Biokinetic Lung Retention Model

Methods

Given the lack of time-specific exposure data for individual workers, the large inter-individual lung retention variability[Sébastien et al., 1990] and the small sample size, only a simplified one-compartment model could be fitted to the data. Four variables were included in the model: exposure intensity *(I)* , estimated by the average respirable dust level (mpcf) to which each subject was exposed while he worked in the asbestos industry; exposure duration *(D)* , the number of years that this occupational exposure lasted; cessation time *(C)* , the number of years between the end of this occupational exposure to asbestos and time of death; age *(A)* , a proxy for the number of years that a worker lived in an asbestos-mining town and was exposed to non-occupational asbestos pollution. Three parameters were to be estimated by regression: K_d , the asbestos fibre fraction of the respirable dust concentration actually inhaled, deposited and remaining in the deep lung a few months after inhalation (when biokinetic conditions have stabilized); K_e , the clearance rate or fraction of retained asbestos fibres cleared yearly from the deep lung (by whatever mechanisms); *envir*, the unknown fibre level in the non-occupational milieu of the worker (home and other). The multiple compartments inferred by other workers[Vincent et al., 1985; Vincent and Donaldson, 1990] were not accounted for in the present modeling; such detailed mechanisms had little relevance in a long-term perspective and with imprecise exposure data, and could be conceived as included in a broader definition of K_d comprise the exposure fraction deposited in the lung *and remaining* after the early fibrillation and the fast clearance which occur in the first months or year after exposure. As well, the interpretation of K_e in the following analyses must be restricted to the long-term clearance rate.

If K_e is the proportion of retained asbestos fibres cleared at the end of one year, then the retention fraction of fibres in the lung at one point in time will be $(1-K_e)$ after one year, and

$(1-K_e)^y$ after y years. If a fraction K_d of a given instantaneous asbestos fibre exposure is retained in the deep lung, then after y years the contribution to lung burden resulting from this single exposure would be:

$$lung\ burden = I \bullet K_d \bullet (1-K_e)^y$$

Assuming constant exposure intensity, constant "deposition" rate and constant clearance rate over a continuous exposure period, lung burden at the end of this period can be estimated by integrating the marginal contribution to lung burden by each instantaneous new dose. If the lung is dust-free at the beginning of a period, then lung burden at the end of a continuous exposure period is:

$$\begin{aligned}
\int_0^D I \bullet k_d \bullet (1-k_e)^y \bullet \partial y &= I \bullet k_d \bullet \int_0^D (1-k_e)^y \bullet \partial y \\
&= I \bullet k_d \bullet \left| \frac{(1-k_e)^y}{\ln(1-k_e)} \right|^D - \left| \frac{(1-k_e)^y}{\ln(1-k_e)} \right|^0 \\
&= I \bullet k_d \bullet \left[\frac{(1-k_e)^D}{\ln(1-k_e)} \right] - \left[\frac{1}{\ln(1-k_e)} \right] \\
&= I \bullet k_d \bullet \left[\frac{(1-k_e)^D - 1}{\ln(1-k_e)} \right]
\end{aligned}$$

where I is the exposure intensity (i.e. exposure level), K_d is the deposition rate, K_e is the yearly clearance rate, and D is the number of years of continuous exposure (duration).

If a cessation period C has occurred between last year of exposure and year of death, then only a fraction $(1 - Ke)^C$ of the lung burden at the end of the exposure period will remain in the lung at time of death. So the complete equation becomes:

$$lung\ burden = I \bullet K_d \bullet \left[\frac{(1-K_e)^D - 1}{\ln(1-K_e)} \right] \bullet (1-K_e)^C$$

Nota: Berry et al.[Berry et al., 1989] derived the same formula from the same modeling assumptions. However, this similarity is not evident at first sight due to the differences in algebraic form and symbology and due also to a slight typographical error.

$$A_O = k \cdot d_o^{\lambda-1} \cdot \left[e^{-\lambda(t-t_2)} - e^{-\lambda(t-t_1)} \right], \text{ which should have read as:}$$

$$= k \cdot d_o \lambda^{-1} \cdot \left[e^{-\lambda(t-t_2)} - e^{-\lambda(t-t_1)} \right]$$

Translating the symbols to those used here:

$$lung\ burden = K_d \cdot I \cdot K_e^{-1} \cdot \left[e^{-K_e C} - e^{-K_e (D+C)} \right]$$

Making the same approximations as used here:

Since $K_e^{-1} \equiv -1/\ln(1-K_e)$

and $e^{-K_e} \equiv (1-K_e)$ when K_e is small,

$$\text{then } lung\ burden = K_d \cdot I \cdot \frac{-1}{\ln(1-K_e)} \cdot \left[(1-K_e)^C - (1-K_e)^{(D+C)} \right]$$

$$= I \cdot K_d \cdot \frac{-1}{\ln(1-K_e)} \cdot \left[(1-(1-K_e)^D) \right] \bullet (1-K_e)^C$$

$$= I \cdot K_d \cdot \left[\frac{(1-K_e)^D - 1}{\ln(1-K_e)} \right] \bullet (1-K_e)^C$$

Since workers are not exposed occupationally outside working hours, the above formula must be divided by 4.2, the ratio of week-time to work-time (168 h. / 40 h.). Although the working week was longer in the past, vacations and shut-downs should also be taken into account and therefore this figure, used by other workers [Nicholson, 1986; HEI-AR, 1991], was retained. Finally, lifetime average non-occupational exposure *(envir)* was accounted for and estimated in the model by adding an appropriate expression, resulting in the following the model:

$$lung\ burden = \frac{1}{4.2} \bullet I_{occ} \bullet K_d \bullet \left[\frac{(1-K_e)^D - 1)}{\ln(1-K_e)} \right] \bullet (1-K_e)^C + \frac{3.2}{4.2} \bullet I_{nonocc} \bullet K_d \bullet \left[\frac{(1-K_e)^{Age} - 1)}{\ln(1-K_e)} \right]$$

This model makes theoretical sense but it is "intrinsically" nonlinear; its equation cannot be transformed using logs or other mathematical operators to a linear form to be fitted with a linear regression program. Instead, a form of regression called "nonlinear iterative regression" (Systat 5 "Nonlin")[1] was applied to estimate the unknown biokinetic constants K_d and K_e..

---

[1] The Systat non-linear regression program software was also tested on different datasets published by other investigators who used different models: weighted linear[Nicholson, 1986], biokinetic[Greco and et al., 1982], logistic[Cox, 1970; Hosmer and Lemeshow, 1989], and Poisson[Scotto et al., 1974; Kleinbaum et al., 1988] with both maximum likelihood and iterative reweighted least squares loss functions; the results concurred with those published.

The nonlinear iterative regression statistical method[Dennis and Schnabel, 1983; Scales, 1985; Bates and Watts, 1988; SYSTAT and Wilkinson, 1990] is not very different from multivariate linear regression but it offers more flexibility. The parameters of the regression model are estimated so as to minimize a loss function, often the least sum of squares function. However, when the model is intrinsically nonlinear or when there are more than one local minimum of the loss function, there is no simple mathematical formula to minimize the loss function. Instead, parameter estimates must be obtained empirically by trial and error following some algorithm to direct and stop the iterative estimation process such as the Quasi-Newton and Simplex minimization methods. Local singularities and irregularities may hamper the optimization of the minimization method. To ensure that maximum convergence and true minimization were achieved, various starting values for the parameter estimates were tried in different runs, and the more exhaustive and robust Simplex minimization method was used. The parameter estimates and statistics presented hereafter were stable in the different runs of the program.

As for the loss function used for fitting the biokinetic model, a simple least squares fitting criteria was not satisfactory because the distributions of both lung burden and past exposure variables were log-normal and because more weight ought to be given to observations in the lower exposure and lung burden ranges closer to non-occupational exposure levels to which the results were to be extrapolated. As for the above log-linear regressions, the "loss function" to be minimized in the nonlinear regressions was chosen to be the square of the difference between the logarithms of lung burden and of the regression estimate ("y-hat").

$$LOSS = \sum_{n=1}^{72} [\ln(burden) - \ln(estimate)]^2$$

Another loss function was also used, weighing the residual sum of squares of the regression by the inverse of each worker's average exposure intensity (*mpcf*) to mitigate the influence of workers with extreme lifetime average exposure intensities relative to the less exposed target non-occupationally exposed population. Moreover, the variance of the error term seemed to

increase with *mpcf*, so that weighing by *1/mpcf* was indicated to obtain homoscedasticity of the errors with respect to the exposure variable of interest[Johnston, 1984; Armitage and Berry, 1994]. Thus the following loss function was applied to the same nonlinear regression model as the logarithmic loss function.

$$LOSS = \sum_{n=1}^{72} w^2 \bullet (burden - estimate)^2$$

$$= \sum_{n=1}^{72} \left(\frac{burden - estimate}{mpcf} \right)^2$$

i.e. $w = 1/mpcf$

Both loss functions were used and compared in the analyses.

Results of the Nonlinear Biokinetic Modeling

The results (Table B17-1) of the biokinetic models applied to the 72 workers must be interpreted carefully given the low explanatory power (R^2) of three models, the large confidence intervals, the lack of statistical significance of most parameter estimates and the strong correlations between parameter estimates in all models (0.8 to 0.9 between K_d and K_e) suggesting a multicolinearity problem. Applying the biokinetic model to the data set of Thetford asbestos workers, the weighted least squares loss function had much more explanatory power (R^2=.83 and .41) than did the logarithmic loss function (R^2=.24 and .08), and tremolite had much more explanatory power than chrysotile independently of the loss function.

With the logarithmic loss function, the clearance rate estimates (K_e) were 0.9%/year for tremolite and 43.2%/year for chrysotile, and their confidence intervals did not overlap. With the weighted least squares loss function, a 3.8% clearance rate estimated for tremolite whereas a 19% clearance rate was estimated for chrysotile; the confidence interval of the chrysotile estimate included the confidence limits of tremolite's clearance estimate. The 0.55 mpcf non-occupational lifetime average exposure intensity estimated with the weighted least squares tremolite model had a 95%CI of 0.26-0.84 mpcf. When the parameter estimates were reentered as fixed values in the model and the model was fitted to non-occupationally exposed groups, average exposure intensity was estimated at 0.22 mpcf for the household-contact group and 0.07 for the environmental group. The values estimated with the tremolite log-loss model were about twice as high.

Table B17-1 Fitted Biokinetic Models on the Thetford Cohort Data (n=72)

		- Parameter Estimates -		*- Projections -*	
Fibre type	**Parameter / Statistic**	**Estimate**	**95% CI**	**Domest. mpcf**	**Envir. mpcf**
Loss function:	Logarithmic least squares				
Tremolite	R^2	.243			
	K_{dt}	0.131	-0.07, +0.33		
	K_{et}	0.009	-0.04, +0.06		
	Non-occ. mpcf	1.113	-0.15, +2.38	0.485	0.151
Chrysotile	R^2	.082			
	K_{dc}	2.490	-2.91, +7.89		
	K_{ec}	0.432	+0.16, +0.70		
	Non-occ. mpcf	0.847	-0.60, +2.29	0.815	51.91
Loss function:	Weighted least squares (w=1/mpcf)				
Tremolite weight.lst.sq.	R^2	.831			
	K_{dt}	0.511	-0.08, +1.11		
	K_{et}	0.038	-0.02, +0.09		
	Non-occ. mpcf	0.549	+0.26, +0.84	0.217	0.068
Chrysotile	R^2	.405			
	K_{dc}	2.684	-5.69, +11.05		
	K_{ec}	0.190	-0.13, +0.51		
	Non-occ. mpcf	0.488	-0.49, +1.47	0.282	0.065

Appendix B18 Discussion on the Validity of the Fitted Biokinetic Model

The biokinetic model was validated by obtaining expectedly much higher yearly clearance rate estimates for chrysotile (19%-43%) than for tremolite (1%-4%), by reproducing this pattern in aggregate data of a cohort of Charleston asbestos textile workers, and by obtaining a specific non-occupational asbestos exposure estimate for Thetford workers and not for Charleston workers.

The faster removal of chrysotile from human lung has been confirmed in many studies[Pooley, 1976; Rowlands et al., 1982; Gylseth et al., 1983]. in addition to those reported by Sébastien, Case, Churg, Wright and others. Berry et al.[Berry et al., 1989] combined a biokinetic model with the classical asbestos-mesothelioma exposure-time-effect model[Peto, 1984] and applied it to the lung burden of gas-mask workers[Jones et al., 1980]; with more indirect and less lung burden data (n=14) than used here, the investigators estimated that Peto's exposure-effect model[Peto, 1984] implied a 15% yearly clearance rate for amphiboles in gas-mask workers. Rodent studies have also observed faster clearance of chrysotile relative to amphiboles[Wagner and Skidmore, 1965; Wagner et al., 1973; Middleton et al., 1977]. Wagner[Wagner et al., 1974] estimated a 20% yearly amphibole clearance rate in rats. In rats exposed 12 months to airborne asbestos and sacrificed either at time of exposure cessation or six months after, Davis[Davis, 1989] observed yearly clearance rates of 36% for short (< 5 μm) and 26% for long amosite fibres, and 99% for short and 80% for long chrysotile fibres. Davis argued that clearance rates should be similar in rats to those in humans because of the similar chemical removal process (leaching or dissolution). From data obtained with a sheep model[Bégin et al., 1983], chrysotile clearance estimated from alveolar lavage was about 77%/year for the first two years, and 38%/year afterwards, very close to our own estimates.

Our clearance estimates were lower than those obtained by Berry et al. for humans and by Davis et al. and Wagner et al. for rodents. The different rates are not irreconcilable however; all the estimates are based on small samples and are measurement-error prone, and most bear on different types of asbestos fibres. Moreover, clearance rates in rats could well be faster than those in workers who smoked[McFadden et al., 1986b; McFadden et al., 1986a; Churg et al.,

1987; Tron et al., 1987] and whose lung burdens were measured many years after exposure cessation (long-term clearance being slower than short-term clearance).

The main limitations of the data were the very small sample sizes, the large inter-individual variability of bio-accumulation, non-differential geometric measurement errors, the absence of smoking data on non-occupational cases and the inevitable selection biases of autopsy series. Most of these problems reduced the reliability of data analyses and projections, and induced "regression-dilution"[Smith and Phillips, 1990; Brenner, 1992; Brenner et al., 1992] bias which lowers the R^2 and tends to obfuscate the relations between independent and dependent variables. In the biokinetic analyses, this dilution bias might have contributed to inflate the estimated non-occupational exposure of workers since this corresponded in a way to that part of occupational exposures which could not be explained by the other parameters. In addition to regression biases, extrapolations from typically smoking workers to mostly non-smoking residents (female and children particularly) of the same area would also underestimate environmental and household-contact asbestos exposures because any given level of lung burden generally represents less cumulative exposure in a smoker than in a non-smoker, probably due to slower pulmonary clearance[McFadden et al., 1986a; McFadden et al., 1986b; Churg et al., 1987; Tron et al., 1987; Churg and Stevens, 1995] in smokers. On the other hand, it has been suggested that clearance could be faster at higher exposure levels or doses[Sébastien et al., 1986]; since workers had higher exposures, they would tend for that reason to have faster clearance. The latter bias due to smoking being less documented and thus more speculative, it seems more likely that the obtained estimates were underestimated.

Appendix B19 The Panel's Estimation Process for Each of 4 Key Years

i. Year 1984

The panel felt that the best source of data for 1984 was Sébastien's full-year continuous measurement survey[Sébastien et al., 1986]. Because of possible unrepresentativeness of the Environment Quebec sampling stations used by Sébastien et al., the panel roughly reweighted the sampling stations so as to better reflect the population centroids. This reduced the averages based on Sébastien's data and, interestingly, closed the gap between Sébastien's measurements and QAMA's data for 1984. The PCOM estimates for 1984 based on reweighted and converted TEM counts from Sébastien et al. appear in row A of Table B19-1.

ii. Year 1974

For 1974, the panel used their estimates for 1984 as reference levels to which they applied town-specific asbestos dust concentration ratios: the 1974 asbestos dust levels divided by the 1984 dust levels. To estimate these ratios, Environment Quebec's year-round sampling measurements of respirable dusts were used instead of QAMA's one-day-per-year measurements of PCOM f/mL; the latter showed extreme year-to-year random variation within each town and panelists had no confidence in PCOM measurements below 0.05 f/mL. The panelists applied the fitted town-specific pollution-production models (Table B-14 in Section B.1.6.e) to their 1984 estimates to estimate levels in 1974. These calculations are summarized in the top part (rows A, B and C) of Table B19-1.

I am not sure if the panel finally considered changes in production levels even though it intended do so. Maybe it did but that this effect was canceled out by other factors. If it did not however, then the panel would have underestimated levels in 1974 by a factor of 2.0-2.7. However, this would not have significantly changed their estimate for 1945. Therefore, the final effect of this suspected error might be surmised as underestimating the overall cumulative exposure estimate by about 20-30%.

Table B19-1 Summary of the Derivation of Estimates for 1974

	Factor	Asbestos	Thetford Mines	Black Lake
A	1984 estimates by the Expert Panel	10 f/L	7 f/L	47 f/L
B	'74/'84 emission or penetrance ratio	3.4	6.9	3.0
C	1974 estimates by Expert Panel = **A x B**	**35 f/L**	**49 f/L**	**141 f/L**
D	Production '74 /Production '84 ratio	2.0	2.1	2.7
E	Asb. dust '74 /Asb. dust '84 ratio = **B x D**	6.8	14.6	8.1
F	1974 estimates with the town-specific models = **A x E**	68 f/L	102 f/L	381 f/L

A = the panel's town-specific estimate for 1984;

B = the ratio obtained from Table B-14 in Section B.1.6 as $gr_t^{('74-'84)}$:

C = **A** x **B** = panel's estimates.

D = Ratio of dust levels after removing the town-specific fixed portion (non-production related background) of 28, 29 and 22 f/L estimated in the pollution-production models (Table B-14);

E = **A** x **D**;

F = **A** x **E** = my own estimate.

iii. Year 1945

The panel then jumped to year 1945 rather than to 1960 because the level of dust controls was more precisely known for 1945 than for 1960 and because the 1945 estimates would be the fulcrum in the extrapolation of yearly values over a long period from 1900 to 1959. The dustiness ratio between 1945 and 1974 estimates could be estimated from the visible-deposit-recall survey, the production-based projections, the engineering-based calculations and, to a lesser degree, the lung burden-based estimations.

The group did not agree immediately on a best estimation method, although production-based projections were at the center of their reflection and even though the different datasets converged toward similar estimates. The panelists were concerned with the extremely high range and the large uncertainty of the exposure estimates. So some panelists figured out their own estimates independently to convince themselves or to seek some external validation.

One panelist used a macroscopic analogy with major emission controls introduced in Pittsburgh between 1930 and 1980 to see how much reduction in particulate concentrations could be expected by passing from an uncontrolled to a controlled environment. The calculated dust reduction ratios were in the 17-50 range for total suspended particulate matter ($\mu g/m^3$) and 27-100 for SO_2. Applying the two maximum factors (50 and 100) to QAMA's 1984 data and adjusting for changes in production between 1945 and 1984, he obtained estimates of roughly 200-400 f/L in Asbestos, 700-1400 f/L in Thetford Mines, 1120-2250 f/L in Black Lake, and 740-1480 f/L for the three-town average in 1945. In a second approach, he used the production-based projections in Table B-15 in Section B.1.6.e and different assumptions about changes in dust filtration efficiency of baghouses and other filtration

systems, and arrived at somewhat similar estimates Accordingly, this panelist proposed a first approximation of 1000 f/L for all three towns.

Another panelist did some calculations on the basis of the 1974-1984 concentration-production-based projections adjusted by the visible dust deposit recall and lung burden data and came up with rough estimates of 1000-1500 f/L for the three towns.

There was some initial disagreement about relative exposure concentrations in the three mining towns. However, prompted by the recalls of visible dust deposits which were similarly important in all three towns, it was finally decided that there was no credible basis for estimating different levels.

The panel agreed on a ballpark figure of 1000 f/L for each town with in 1945. However, they acknowledged their uncertainty by indicating a plausible range of estimates. While the best estimate of 1000 f/L was fragile, the panel was confident that the true values could not likely have been less than one third of their best estimate, nor could the true values likely have been more than three times greater than the best estimate. Thus they arrived at what could be termed a "plausibility range" of 333-3000 f/L.

iv. Year 1960

With various assumptions and interpolations between and 1945 and 1974, the panel estimated a low level of 100 f/L for Asbestos because of the full installation of baghouses on the dryer and mills, an intermediate level of 250-500 f/L for Thetford where emissions were uncontrolled but production volume was low, and 500-700 f/L for Black Lake where production and tailings piles grew rapidly bringing the population closer to various emission sources. The survey data was most useful for this step of the exposure assessment.

Appendix B20	Miscellaneous Industrial Hygiene Measurements Which Could Hint About Past Neighbourhood and Household Exposure Levels

Site or Subject	Date	N	Method	Units	Mean	Range or S.D.
SAMPLES in 1970:						
Gate	1970	1	Personal	f/mL	3.1	n.a.
Gate	1970	3	Fixed	f/mL	7.0	2.1 - 5.
Surveyor	1970	1	Personal	f/mL	4.4	n.a.
Mill baghouse	1970	3	Fixed	f/mL	0.7	0.1 - 1.7
Management offices	1970	4	Fixed	f/mL	1.4	0.3 - 2.6
Offices	~1972	12	Fixed	f/mL	1.7	±1.3
Offices	~1972	12	Fixed	mpcf	0.48	±0.10
Canteens & offices	~1972	18	Fixed	f/mL	3.4	±3.8
Canteens & offices	~1972	18	Fixed	mpcf	0.45	±0.21
Cafeteria	1970	6	Fixed	f/mL	3.7	1.0 - 5.4
Entrance gate	1970	3	Fixed	f/mL	7.0	2.1 - 16.2
Mill Baghouse	1970	3	Fixed	f/mL	0.7	0.1 - 1.7
SAMPLES in 1990:						
Mill baghouse	1990	16	Fixed	f/mL	0.05	0.03 - 0.08
Supervisors' offices	1990	3	Fixed	f/mL	0.2	0.1 - 0.34
Cafeteria	1990	6	Fixed	f/mL	0.15	0.01 - 0.32
Mill baghouse	1990	16	Fixed	f/mL	0.05	0.03 - 0.08

Note: These measurements were made available to us by a plant manager.

Appendix B21	Three Scenarios on Past Outdoor and Indoor Levels Compatible With the Lung Burden Analyses

The cells in the tables for the three scenarios were deduced by 1° calculating the neighbourhood-only indoor, outdoor and daily average exposures. 2° Then a distance effect which differed between the three scenarios was applied to estimate the exposure level outside the homes of household contacts. 3° According to the assumption or not of a house-filter effect or of housework effect, I calculated the indoor level and then the daily average exposure of household contacts when they did not live with an asbestos worker. 4° The daily average exposure of household contacts when they lived with an asbestos worker was estimated on the assumption that the t.w.a. 0.76 f/mL estimate obtained over the years lived in the area was the average of the daily average exposure levels when household contacts lived with and when they did not live with an asbestos worker. 5° Finally, indoor exposure of household contacts when they lived with an asbestos worker was the level to which they were exposed 19 hours per day given their daily average and outdoor exposure levels. For comparison with other data sources, ratios were computed.

Scenario #1. In Table B21-1, it was assumed that 1° the neighbourhood-only group was exposed at 0.23 f/mL daily, indoor and outdoor. 2° The asbestos levels outdoor of houses of "household contacts" would have been 1.73 ($=2.6^{-0.57}$) times higher: 0.40 f/mL. 3° No house-filter effect was assumed in this scenario, so that indoor and daily average exposure of household contacts were also 0.40 f/mL when they did not live with an asbestos worker. 4° The daily average exposure of household contacts when they lived with an asbestos worker was estimated as (0.76 - .5*0.40)/.5 = 1.12 f/mL. 5° The corresponding indoor level was estimated at (1.12 - .2*0.40)/.8 = 1.30 f/mL.

Scenario #2. In Table B21-2, 1° the neighbourhood-only group was exposed at 0.23 f/mL daily, 0.223 f/mL indoor and 0.255 f/mL outdoor for a 12.5% house-filter effect. 2° The asbestos levels outdoor of houses of "household contacts" would have been 2.6 ($=2.6^{-1.0}$) times higher: 0.664 f/mL. 3° Indoor and daily average exposure of household contacts when they did not live with an asbestos worker were simply 2.6 times higher than for the neighbourhood-only exposed. 4° The daily average exposure of household contacts when they

lived with an asbestos worker was estimated as (0.76 - .5*0.60)/.5 = 0.92 f/mL. 5° The corresponding indoor exposure level was estimated at (0.92 - .2*0.66)/.8 = 1.00 f/mL.

Scenario #3. In Table B21-3, 1° the neighbourhood-only group was exposed at 0.23 f/mL daily, 0.24 f/mL indoor and 0.19 f/mL outdoor for a +25% indoor housework effect over outdoor level. 2° The asbestos levels outdoor of houses of "household contacts" would have been 1.6 ($=1.6^{1.0}$) times higher: 0.307 f/mL. 3° Indoor and daily average exposure of household contacts when they did not live with an asbestos worker were simply 1.6 times higher than for the neighbourhood-only exposed. 4° The daily average exposure of household contacts when they lived with an asbestos worker was estimated as (0.61 - .5*0.37)/.5 = 0.85 f/mL. 5° The corresponding indoor exposure level was estimated at (0.85 - .2*0.31)/.8 = 1.00 f/mL.

Table B21-1 Scenario #1: Analysis of Past Indoor and Outdoor Exposures of Household Contacts and of Neighbourhood-Only Exposed Residents Based on Lung Burden Data

Thetford Mines	**Neighbourh. -Only Exposed Residents**	**Household-Contact Exposed Residents**		**Ratios Used to Build the Table**	
sample size	n = 22	n = 10			
km from mines	2.6* x	x		*distance effect* = $2.6^{0.57} = 1.73$	
(column #)	(1)	(2)	(3)	(3) vs. (1)	(3) vs. (2)
	Never lived with asbestos worker	While NOT living with asbestos worker	While living with asbestos worker	Crude Ratio	Distance-Adjusted Ratio
average lung burden	0.3 f/µg	1.6 f/µg			
t.w.a. exposure over total years of residence	0.23 f/mL	0.76 f/mL (=0.5 x 0.40 + 0.5 x 1.12)			
average daily exposure level of female residents	0.23 f/mL	0.40 f/mL	1.12 f/mL	4.9	2.8
indoor exposure (B)	0.23 f/mL	0.40 f/mL	1.30 f/mL	5.7	3.3
outdoor exposure (A)	0.23 f/mL	0.40 f/mL	0.40 f/mL	1.7	1.0
Indoor vs. Outdoor Ratio (B/A)	1.0	1.0	3.3		

Table B21-2 Scenario #2: Analysis of Past Indoor and Outdoor Exposures of Household Contacts and of Neighbourhood-Only Exposed Residents Based on Lung Burden Data

Thetford Mines	**Neighbourh. -Only Exposed Residents**	**Household-Contact Exposed Residents**		**Ratios Used to Build the Table**	
sample size	n = 22	n = 10			
km from mines	2.6* x	x		*distance effect* = $2.6^{1.0} = 2.6$	
(column #)	(1)	(2)	(3)	(3) vs. (1)	(3) vs. (2)
	Never lived with asbestos worker	While NOT living with asbestos worker	While living with asbestos worker	Crude Ratio	Distance-Adjusted Ratio
average lung burden	0.3 f/µg	1.6 f/µg			
t.w.a. exposure over total years of residence	0.23 f/mL	0.76 f/mL (=0.5 x 0.60 + 0.5 x 0.92)			
average daily exposure level of female residents	0.23 f/mL	0.60 f/mL	0.92 f/mL	4.0	1.5
indoor exposure (B)	0.22 f/mL	0.58 f/mL	1.00 f/mL	4.5	1.7
outdoor exposure (A)	0.26 f/mL	0.66 f/mL	0.66 f/mL	2.6	1.0
Indoor vs. Outdoor Ratio (B/A)	0.88	0.88	1.5		

Table B21-3 Scenario #3: Analysis of Past Indoor and Outdoor Exposures of Household Contacts and of Neighbourhood-Only Exposed Residents Based on Lung Burden Data

Thetford Mines	**Neighbourh. -Only Exposed Residents**	**Household-Contact Exposed Residents**		**Ratios Used to Build the Table**	
sample size	n = 22	n = 10			
km from mines	1.6* x	x		*distance effect* = $1.6^{1.0} = 1.6$	
(column #)	(1)	(2)	(3)	(3) vs. (1)	(3) vs. (2)
	Never lived with asbestos worker	While NOT living with asbestos worker	While living with asbestos worker	Crude Ratio	Distance-Adjusted Ratio
average lung burden	0.3 f/µg	1.21 f/µg[1]			
t.w.a. exposure over total years of residence	0.23 f/mL	0.61 f/mL[2] (=0.5 x 0.37 + 0.5 x 0.85)			
average daily exposure level of female residents	0.23 f/mL	0.37 f/mL	0.85 f/mL	3.7	2.3
indoor exposure (B)	0.24 f/mL	0.38 f/mL	1.00 f/mL	4.1	2.6
outdoor exposure (A)	0.19 f/mL	0.31 f/mL	0.31 f/mL	1.6	1.0
Indoor vs. Outdoor Ratio (B/A)	1.25	1.25	3.2		

[1] The lung burden and overall t.w.a. exposure were adjusted here as if the sample of 10 household contacts had lived 1.6 times rather than 2.6 times closer to the mines. The true figures were thus divided by $(2.6/1.6)^{0.57}$.

[2] See previous footnote.

Appendix B22	Sampling Fractions by Agglomeration, Age Group and Follow-Up Decade

Table B22-1 NY:PY Sampling Fractions by Agglomeration, Age Group and Follow-Up Decade

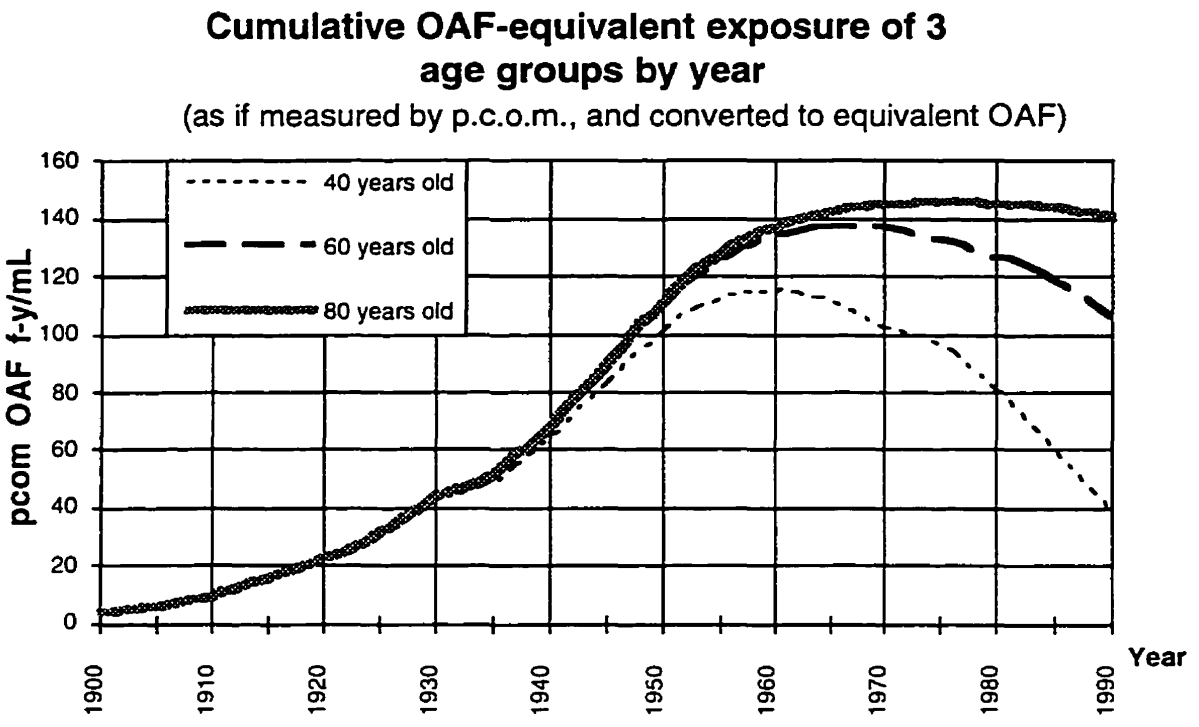
Agglomeration of Asbestos

Age group	1950-1959	1960-1969	1970-1979	1980-1989	Total by age group
30-44	4.66 %	4.40 %	2.96 %	0.23 %	3.20 %
45-54	3.53 %	7.31 %	11.63 %	5.53 %	7.62 %
55-69	2.03 %	5.80 %	10.97 %	11.55 %	8.94 %
70+	0.00 %	1.19 %	5.56 %	14.12 %	7.28 %
Total by decade	3.60 %	4.84 %	7.36 %	7.02 %	5.88 %

Agglomeration of Thetford Mines

30-44	4.13 %	3.52 %	1.94 %	0.08 %	2.33 %
45-54	2.16 %	5.88 %	5.83 %	2.62 %	4.27 %
55-69	0.65 %	3.47 %	7.98 %	5.98 %	5.32 %
70+	0.00 %	0.71 %	4.22 %	8.59 %	5.00 %
Total by decade	2.81 %	3.72 %	4.36 %	3.60 %	3.69 %

Appendix B23 **Cumulative Exposure If Residents Spent Their Whole Lifetime in Area.**



"OAF" means "Occupational Asbestos Fibre"; in fact, it refers to the exposure circumstances of an asbestos worker exposed 40 hours per week equivalent to the exposure of a continuously exposed resident that would result in the same cumulative asbestos exposure.

The above graphic means that an 80-year old person in 1980 had twice the lifetime cumulative exposure of an 80-year old person in 1940. This age-group had a relatively constant cumulative exposure from 1960 to 1990. However, a 40-year old person in 1980 had about the same lifetime cumulative exposure of a 40-year old person in 1940. This age group had a decreasing cumulative exposure from 1960 to 1990. Age-specific asbestos-attributable excess risks should follow the patterns of these curves.

A fist law applies throughout the whole century: in any given year, the older age groups have higher cumulative lifetime exposures than the younger ones. The second law is that the cumulative exposure of older age groups is less sensitive than that of younger age groups to changes in exposure intensity levels.

Appendix C1 Correspondence Table of ICD and LCDC codes, 1950-1989

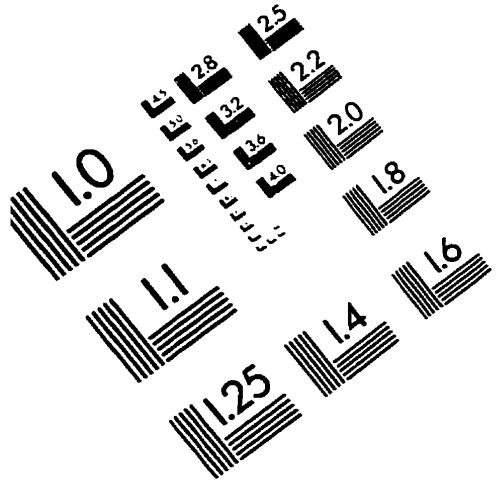
Table C1-1 Causes of Death and Corresponding ICD and LCDC Codes (Part 1)

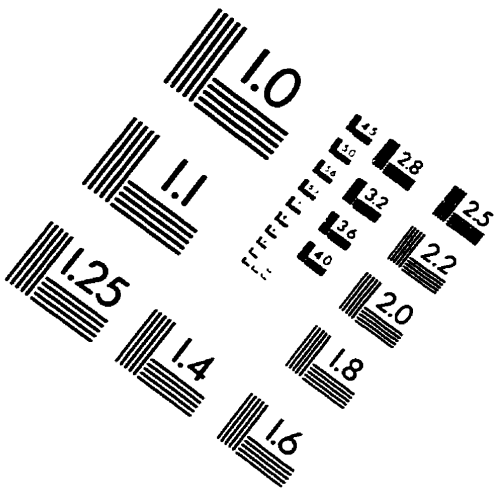
NON-NEOPLASTIC DISEASES	LCDC	ICD-9 1979-88	ICD-8 1969-78	ICD-7 1958-68	ICD-6 1950-57
Infective and Parasitic NOT(pulmon. tuberc. pneumoc.)	2 NOT(129.2)	001-009, 013-139	001-010, 013-136	003.1-138, 571, 696, 697, 764, 767, 768, 785.6	003.1-138, 571, 696, 697, 764, 767, 768, 785.6
Blood, Endocrine, Metabolic	4, 5	240-289	240-289	250-291, 292.0-292.7, 298.1, 299, 468.0-468.2, 772	250-291, 292.0-292.7, 298.1, 299, 468.0-468.2, 772
Mental and Nervous Disorders	6, 7	290-389	290-315, 340-398	300-326, 688.1, 781.5, 744.1, 765, 780.3	300-326, 688.1, 781.5, 744.1, 765, 780.3
Circulatory Diseases					
Acute Myocardial Infarction	124	410	410	NE	NE
Other Ischemic Heart Disease	117 NOT(124)	411-414	411-414	420	420
Hypertensive Dis.	121	401-405	400-404	NE	NE
Other Circulatory	8 NOT(117, 121)	390-400, 406-409, 415-459	390-399, 405-409, 415-458	330-334, 400-419, 421-467, 468.3, 570.2	330-334, 400-419, 421-467, 468.3, 570.2
RESPIRATORY DISEASES					
Asbestosis	142	501	515.2	523.2	523.2
Other Pneumoconioses	140 NOT(142)	011.4, 495, 500, 502-505	010, 515.0, 515.1, 515.3-516	001, 523.0, 523.1, 523.3-524	001, 523.0, 523.1, 523.3-524
Chronic Bronchitis, Emphysema, Asthma	132	490-493, 496	490-493, 519.3	241, 501, 502, 527.1	241, 501, 502, 527.1
Tuberculoses Pulmon. et Respir.	129.2	010-012	011-012	001, 001, 003.0	001, 001, 003.0
Other Respiratory	9 NOT(132, 142)	460-489, 494, 497-499, 506-519	460-489, 494-514, 515.4-519.2, 519.4-519.9	240, 470-500, 503-522, 525, 526, 527.2-527.9, 763	240, 470-500, 503-522, 525, 526, 527.2-527.9, 763
Digestive Diseases	10	520-579	520-577	530-561, 570.0, .1, .3, .4, .5, 572-587, 784.6	530-561, 570.0, .1, .3, .4, .5, 572-587, 784.6
Accidents, Poisonings, Violence, Unnatural	207	800-999	800-999	365, 800-999	365, 800-999
Ill-Defined Symptoms	17	780-799	780-796	780-795	780-795
Other Non-Neoplastic Diseases	11-16	580-779	580-779	?	?

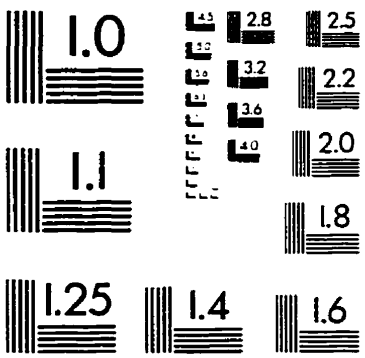
Table C1-1 Causes of Death and Corresponding ICD and LCDC Codes (Part 2)

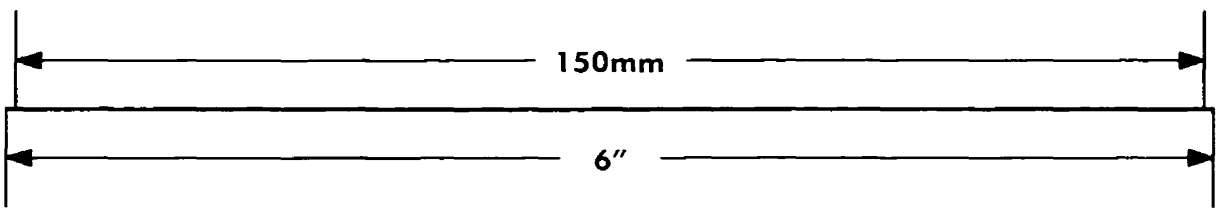
NEOPLASTIC DISEASES	LCDC	ICD-9 1979-88	ICD-8 1969-78	ICD-7 1958-68	ICD-6 1950-57
ORAL					
Pharynx	28	146-148, 149.0	146-149	145-148	145-148
Buccal Cavity, Lip, Other Oral	29 NOT(28)	140-145	140-145	140-144	140-144
DIGESTIVE					
Oesophagus	31	150	150	150	150
Stomach	32	151	151	151	151
Small Intestine	33	152	152	152	152
Large Intestine (Colon), Rectum	36	153, 154, 159.0	153, 154	153, 154	153, 154
Peritoneum	41	158	158	158	158
Other Digestive: Pancreas, Liver, Biliary, ...	39, 40, 42	155-157, 159.1-159.9	155-157, 159	155, 157, 159	155, 157, 159
RESPIRATORY					
Larynx	45	161	161	161	161
Lung, Bronchus	46	162.2, .3, .4, .5, .8, .9	162.1	162.1	NE
Pleura	47	163	163.0	162.2	NE
Trachea, Nose, Sinus, Lower Resp. Tract	49 NOT(45-47)	160, 164.2, .3, .8, .9, 165	160-164, NOT(162.1, 163.0)	160-164, NOT(162.1, 163.0)	160, 162-164
Breast	64	174, 175	174	170	170
Genital	74	179-184	180-184	171-176	171-176
Urinary					
Kidney	78	189.0, .1, .2	189.0, .1, .2	180	180
Bladder	79	188	188	181.0	NE
Other Urinary	80 NOT(79)	189.3-189.9	189.3-189.9	181.1-181.9	181
Bone Tissue, Skin	50, 51, 52, 63	164.1, 170-173	164.1, 170-173	179.1, 190-191, 196-197	190-191, 196-197
Eye, Brain, C.N.S.	82,85	190-192, 225, 237.5, .6, 239.6	190-192, 225, 238	192-193, 223, 237	192-193, 223, 237
Endocrine Glands	86, 87	193-194, 164.0	193-194	194-195	194-195
Lymphatic And Hematopoietic	105	200-208, 238.4	200-209	200-205, 292.3, 294	200-205, 292.3, 294
Other, Ill-Defined, Unspecified Sites And Neoplasms	108	195-199, 210-239	195-199, 210-239	156, 163, 165, 198, 199, 210-239	156, 163, 165, 198, 199, 210-239

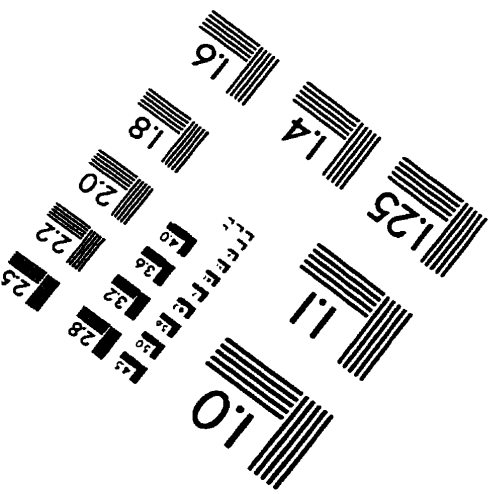
IMAGE EVALUATION TEST TARGET (QA-3)

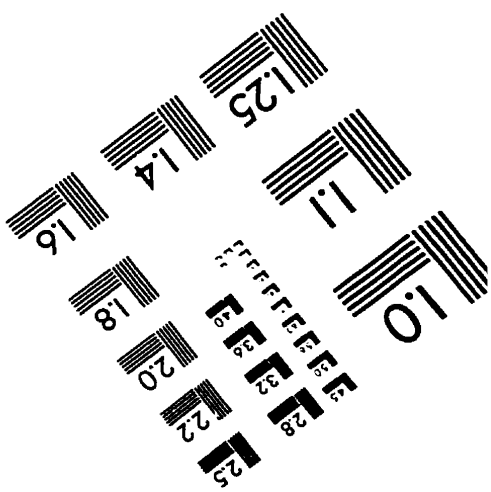












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