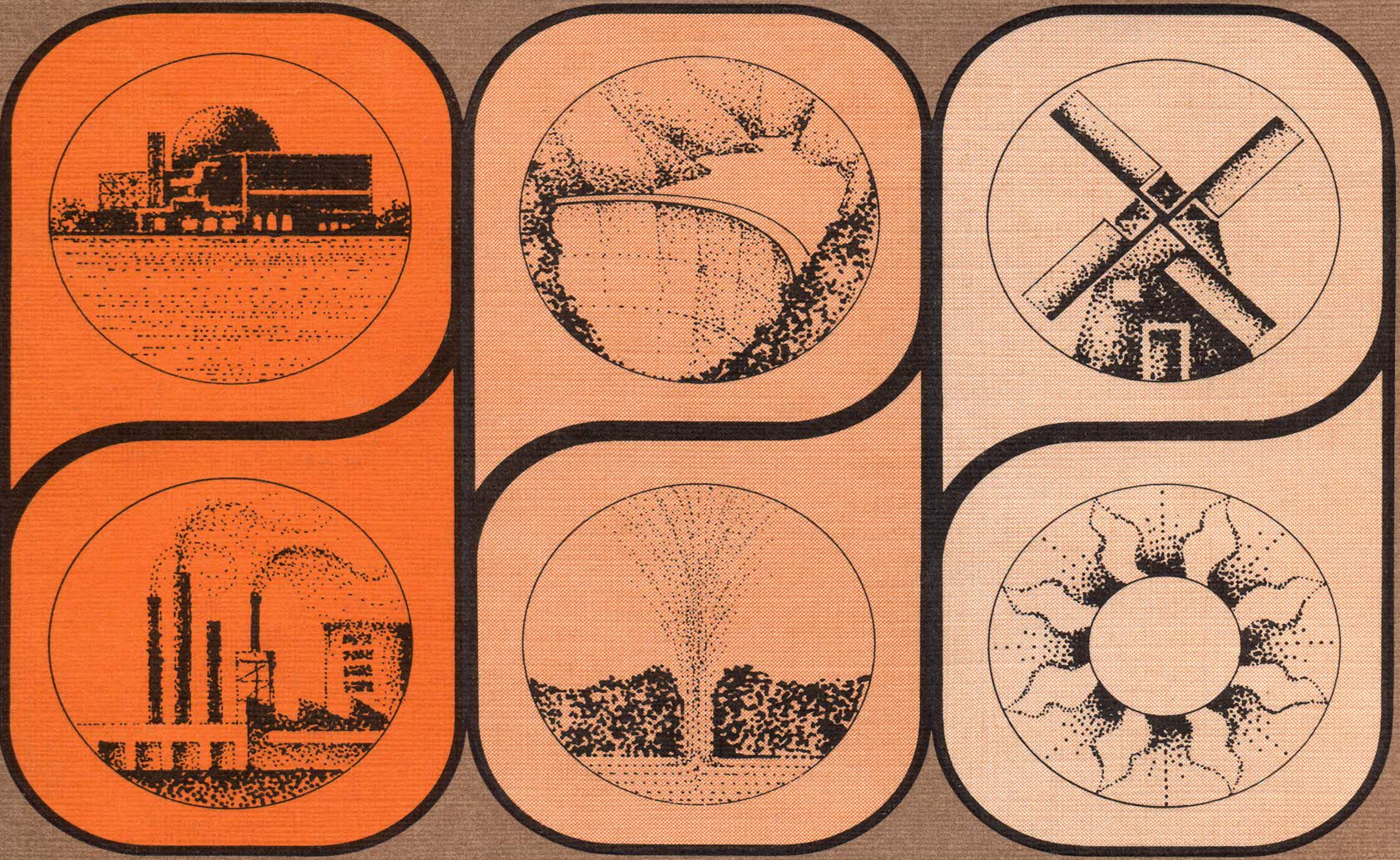
HEALTH IMPACTS OF DIFFERENT SOURCES OF ENERGY

PROCEEDINGS
OF A SYMPOSIUM
NASHVILLE
22–26 JUNE 1981
JOINTLY ORGANIZED
BY WHO, UNEP, IAEA



INTERNATIONAL ATOMIC ENERGY AGENCY, VIENNA, 1982

HEALTH IMPACTS
OF DIFFERENT SOURCES OF ENERGY

PROCEEDINGS SERIES

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PROCEEDINGS OF AN INTERNATIONAL SYMPOSIUM ON
HEALTH IMPACTS
OF DIFFERENT SOURCES OF ENERGY
JOINTLY ORGANIZED BY
THE WORLD HEALTH ORGANIZATION,
THE UNITED NATIONS ENVIRONMENT PROGRAMME
AND
THE INTERNATIONAL ATOMIC ENERGY AGENCY
AND HELD IN NASHVILLE, U.S.A., 22-26 JUNE 1981

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FOREWORD

All possible energy sources will need to be developed as the world population grows, as per-capita consumption of energy increases, and as the availability of certain major fuels is reduced. A world-wide quickening of technological development is therefore taking place in order to add some potential new and renewable energy sources to the present spectrum. The list of major sources -- fossil fuels, nuclear and hydroelectric -- is being enlarged to encompass the developing technology of, among others, solar photovoltaic, geothermal, wind, ocean current and biomass conversion products as potential future energy suppliers.

Most countries are now involved in formulating or revising their national energy policies. They are projecting scenarios for an optimal energy mix appropriate to available resources and in the context of other indigenous socio-economic and technological factors. All these methods of energy production and use pose some risk of detriment to human health and environment through toxic emissions. In the climate of recently growing public concern for the preservation of man's health in his environment, the concept of potential risk attributable to the energy options may, as in all other industrial activities, become an important factor in the development strategies of national and international energy policies. Studies have therefore begun in recent years under the auspices of national and international organizations, such as those sponsored by UNEP, WHO and IAEA, to evaluate quantitatively the health and environmental risks of energy production and use and to promote the perception of those risks in the context of community life.

There is a great deal of diversity in both the quality and the quantity of the pollutant emission potentials of the energy fuel cycles available, and consequently also in their potential health hazard. The level of understanding of the mechanisms involved, their stage of development for data acquisition, and the depth of evaluation, are grossly unequal for different energy fuel cycles: this leads to major uncertainties in the resultant risk estimates. The current knowledge and available supporting data from which risks are to be identified therefore need to be fully and clearly formulated and standardized if they are to be capable of influencing decisions of the authorities concerned and lead to a viable energy policy, regulatory control and guidelines for standard-setting.

There was obviously a need for a critical review and discussion of current methods of data acquisition, modelling, risk quantification and comparative analyses with regard to the health-impact potentials of chemical and physical pollutants emitted from the operation of the different energy fuel cycles. The present symposium, jointly organized by the WHO, UNEP and IAEA, provided such a forum.

Among other topics reported on in the present volume, the suitability of methods of health-risk assessment of energy technologies are reviewed in reports from the USA, Canada, the USSR, France, the United Kingdom and the Federal Republic of Germany, dealing with operational experience of various types of power plants under specific site conditions. Epidemiological data on energy-related fatalities, accidental disabilities, and other stochastic and non-stochastic health disorders, including late effects (carcinogenesis and genetic defects) suffered by occupational workers and the general public, have been validated for risk assessment and for the establishment of exposure-dose-response relationships in order to assist in setting up regulatory guidelines. Data from community health statistics and demography, such as parameters of mean birthweight, infant mortality rates, life expectancy, lost working days, change of trends in specific disease- and age-related mortality in countries with different levels of technological advancement, have provided bases for sensitive methodologies to assess the health impacts from pollutants emitted from energy sources.

The well developed health-risk assessment and regulation aspects of nuclear energy cycles have provided much guidance and have served as model systems for all other energy cycles. However, owing to the fragmentary nature of the input information from conventional energy sources, including, for example, aspects of dose-effect relationships, interactions and dosimetry, the current risk estimates in those cases often suffer from major uncertainties which must be remedied by further efforts.

The consensus of the symposium was as follows:

(1) The development and utilization of all energy sources pose some detriment to man's health and environment.

(2) The overall estimated direct and indirect health impacts of available energy sources are comparable between technologies, with small variations within a range of factors.

(3) It becomes evident from the current estimates that the total detriments to health attributable to energy production, whatever the energy source, form only a minute proportion of those from all causes in society.

(4) Potential health risks in terms of reduction of life expectancy attributable to some energy-conservation measures alone may be greater than the direct and indirect impacts from the generation of an equivalent amount of energy, irrespective of its source. Similarly, an inadequate energy supply might cause a greater loss of life than the production of that amount of energy.

(5) Further improvement in the understanding of dose-response of chemical pollutants along with the metabolic models and their interactions should facilitate

risk evaluation and the setting of regulatory standards for environmental health monitoring of conventional as well as new energy technologies.

This symposium has pioneered the review and comparison of available data on health impacts from a large cross-section of both established and developing energy-source technologies. The proceedings should serve as a valuable reference source for experts in life sciences and biomedical and environmental health research, epidemiologists, radiation biologists, genetic toxicologists, health and environmental regulatory authorities, energy planners and policy-makers.

The success of the symposium was due to a large extent to the expert advice and constant help of Dr. L.D. Hamilton of Brookhaven National Laboratory. The able assistance and hospitality extended by the host Government and the officials in charge of local arrangements, particularly Dr. C. Richmond of Oak Ridge National Laboratory, the United States Scientific Adviser to the meeting, are gratefully acknowledged. To all of them, and to the co-operative authors and participants, the organizers wish to express their gratitude.

EDITORIAL NOTE

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KEYNOTE ADDRESS

COMPARING THE HEALTH IMPACTS OF DIFFERENT ENERGY SOURCES

Keynote Address

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Abstract

COMPARING THE HEALTH IMPACTS OF DIFFERENT ENERGY SOURCES: KEYNOTE ADDRESS.

Assessing health impacts of different energy sources requires synthesis of research results from many different disciplines into a rational framework. Information is often scanty; qualitatively different risks, or energy systems with substantially different end uses, must be put on a common footing. Historically institutional constraints have inhibited agencies from making incisive intercomparisons necessary for formulating energy policy; this has exacerbated public controversy over appropriate energy sources. Risk assessment methods reviewed include examples drawn from work of the Biomedical and Environmental Assessment Division at Brookhaven National Laboratory and elsewhere. Uncertainty over the mechanism and size of air pollution health damage is addressed through a probabilistic health-damage function, using sulphate-particle exposure as an indicator. This facilitates intercomparison through analysis of each step in the whole fuel cycle between a typical coal and nuclear power plant. Occupational health impacts, a significant fraction of overall damage, are illustrated by accident trends in coal-mining. In broadening comparisons to include new technologies, one must include the impact of manufacturing the energy-producing device as part of an expanded fuel cycle, via input/output methods. Throughout the analysis, uncertainties must be made explicit in the results, including uncertainty of data and uncertainty in choice of appropriate models and methods. No single method of comparative risk assessment is fully satisfactory; each has its limitations. Several methods must be compared if decision-making is to be realistic.

The World Health Organization (WHO), the United Nations Environment Programme (UNEP) and the International Atomic Energy Agency (IAEA), which are jointly organizing this symposium, concern themselves with the health effects of different energy sources. WHO, specifically its Division of Environmental Health, has been studying pollutants, many of them originating from energy sources. WHO in collaboration with UNEP has issued Environmental Health Criteria reports, e.g. "Oxides of Nitrogen", "Photochemical Oxidants", "Sulfur Oxides and Suspended Particulate Matter", and has co-sponsored with the IAEA a study on the health implications of nuclear power production. WHO has a small-scale study under way on comparison of low levels of environmental pollutants from energy sources.

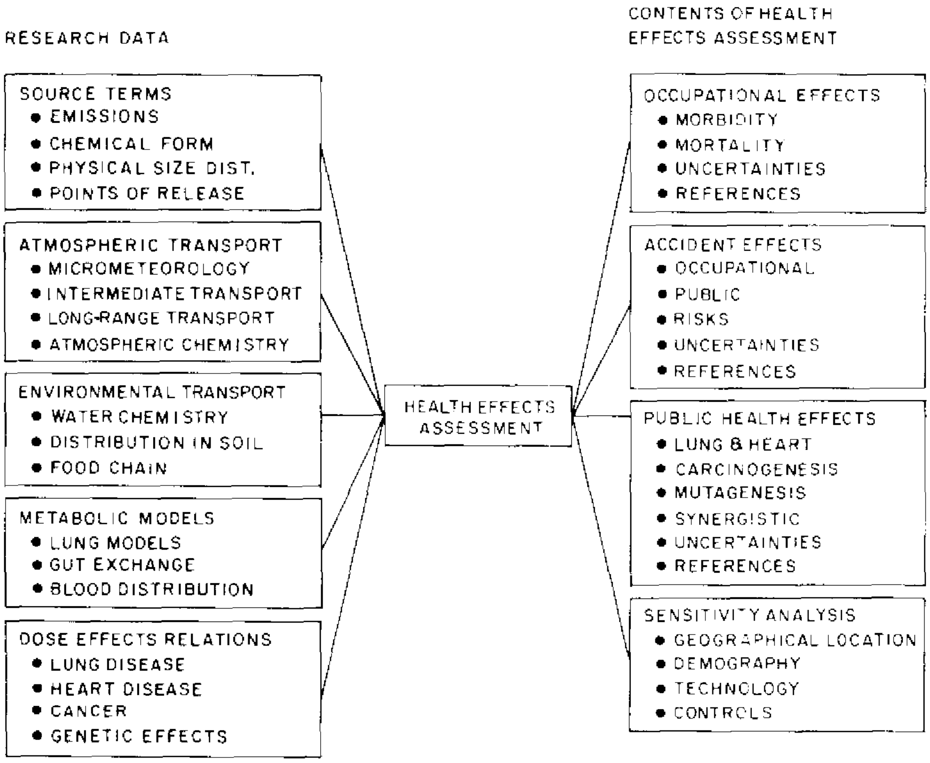


FIG. 1. Health effects assessment: inputs and outputs.

UNEP has began a continuing review of the health and environmental impacts of individual fuel cycles with the aid of international panels of experts. UNEP has published reports on fossil fuels (1979), nuclear energy (1979), and renewable sources of energy (1980). A comparative assessment of environmental impacts of different sources of energy is in preparation. UNEP also co-sponsored with the Beijer Institute, Stockholm, and the USSR Commission for UNEP an International Workshop on Environmental Implications and Strategies for Expanded Coal Utilization, from which an excellent report should be available soon.

The IAEA has a strong interest in the technical base from which to assess health consequences of the nuclear fuel cycle; it co-operates with other agencies on such assessments, e.g. the WHO Regional Office for Europe and the UNEP Nuclear Fuel Cycle Panel.

Assessment of the health impacts of different energy sources requires the co-operation of many disciplines. This symposium considers the diversity of approaches. Although the symposium is restricted to health impacts, coverage of this topic is far from all-inclusive. Figure 1 indicates the potential range of topics to be included in a health and safety analysis. Health assessment requires the physical and chemical definition and quantitation of emissions from each stage in an energy cycle, including definition of the chemical transformation in transport and dispersion of these emissions through air, water and food pathways. Dose-effect relationships derived from epidemiological studies are needed, after definition by use of metabolic models of the distribution of pollutants within man. This symposium covers some elements for this assessment. Accordingly we have sessions on: epidemiological indices for evaluating health impacts; interrelationships of various health, demographic, socioeconomic and environmental energy data necessary for risk estimation and assessment of health; environmental transport and transformation of discharges from energy influencing the estimation of dose to persons; experimental validation of the link between energy-associated effluents and biological damage; methods for quantitating and comparing health risks; risk analyses comparing different energy sources; new risk-assessment designs for comparing health effects of different energy sources. Because of widespread interest, manifested in the contributions focused on this area, a session is devoted to radioactivity in coal. Finally, a poster session covers various topics.

To measure the health costs of energy systems, the systems have to be put into a coherent framework. Figure 2 diagrams energy production, distribution, and use [1]. It furthers analysis of costs and hazards for each stage. The diagram includes: nuclear power; hydropower, geothermal and solar; coal, crude oil and natural gas. An additional topic, technologies under development, could be included. Stages in the fuel cycles are: (1) exploration and extraction; (2) refining and conversion; (3) transport and storage; (4) central-station conversion; (5) transmission, distribution and storage; (6) decentralized conversion; (7) utilizing device; (8) energy services by demand sector. Most of these stages entail their own health costs, some direct (e.g. risks of injury or death in mining), some indirect (e.g. release of pollutants into air and water; thence into food chains, etc.).

The range of information needed to calculate health effects of different energy systems -- emissions characterization, environmental transport, transformation and dispersion, demography, effects studies, risk assessments, comparisons -- means that such assessment depends on more than merely the assessors who make the final calculations. This principle I learned as a research physician. Often, recognition is given only to the physician who gives a drug to a patient; skimpy recognition is given those who developed and tested it in the laboratory, thus enabling clinical trial. Similarly, in assessing the health effects of different energy systems, we depend on the research of those who characterized the emissions, chemical transformation, dispersion,

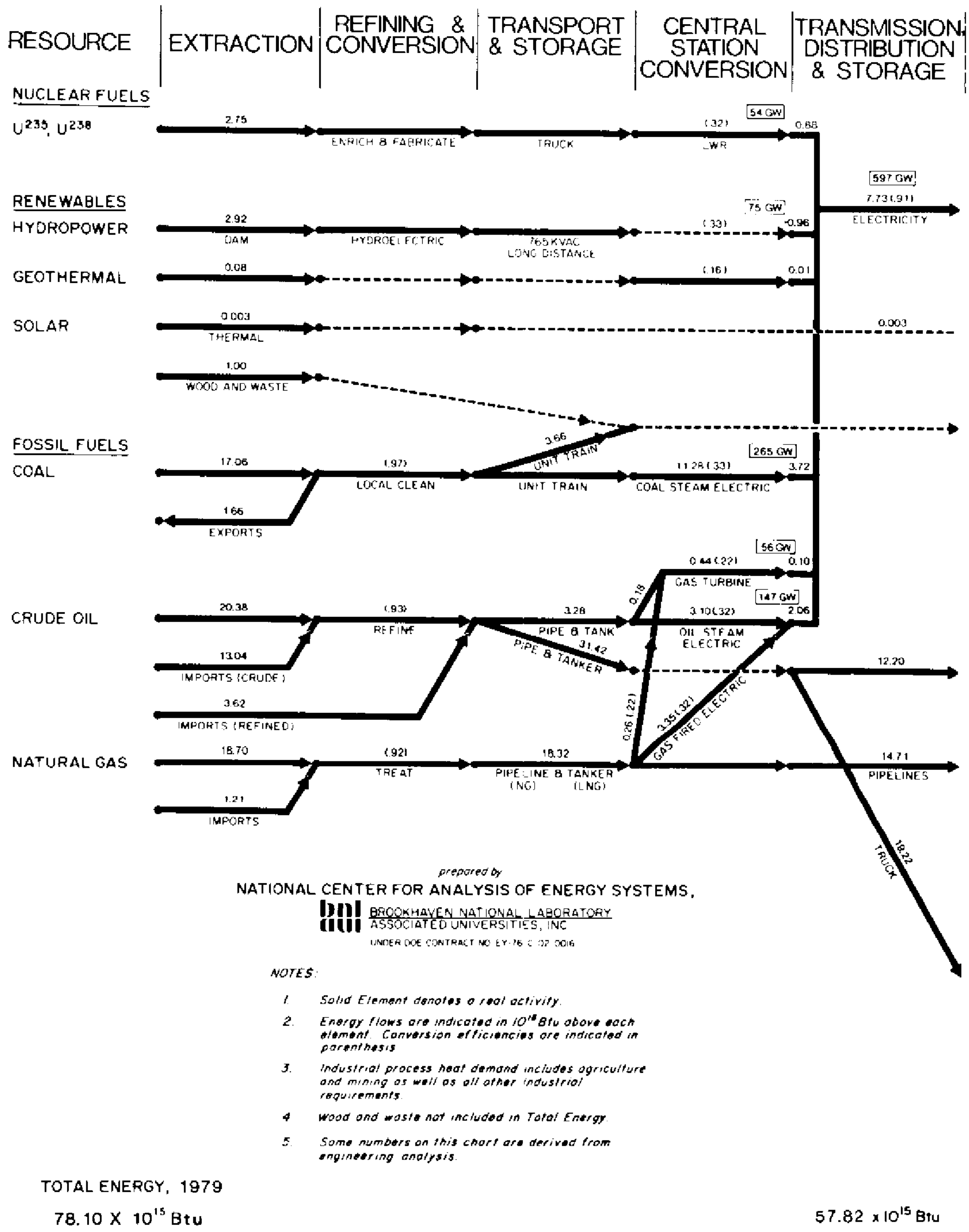
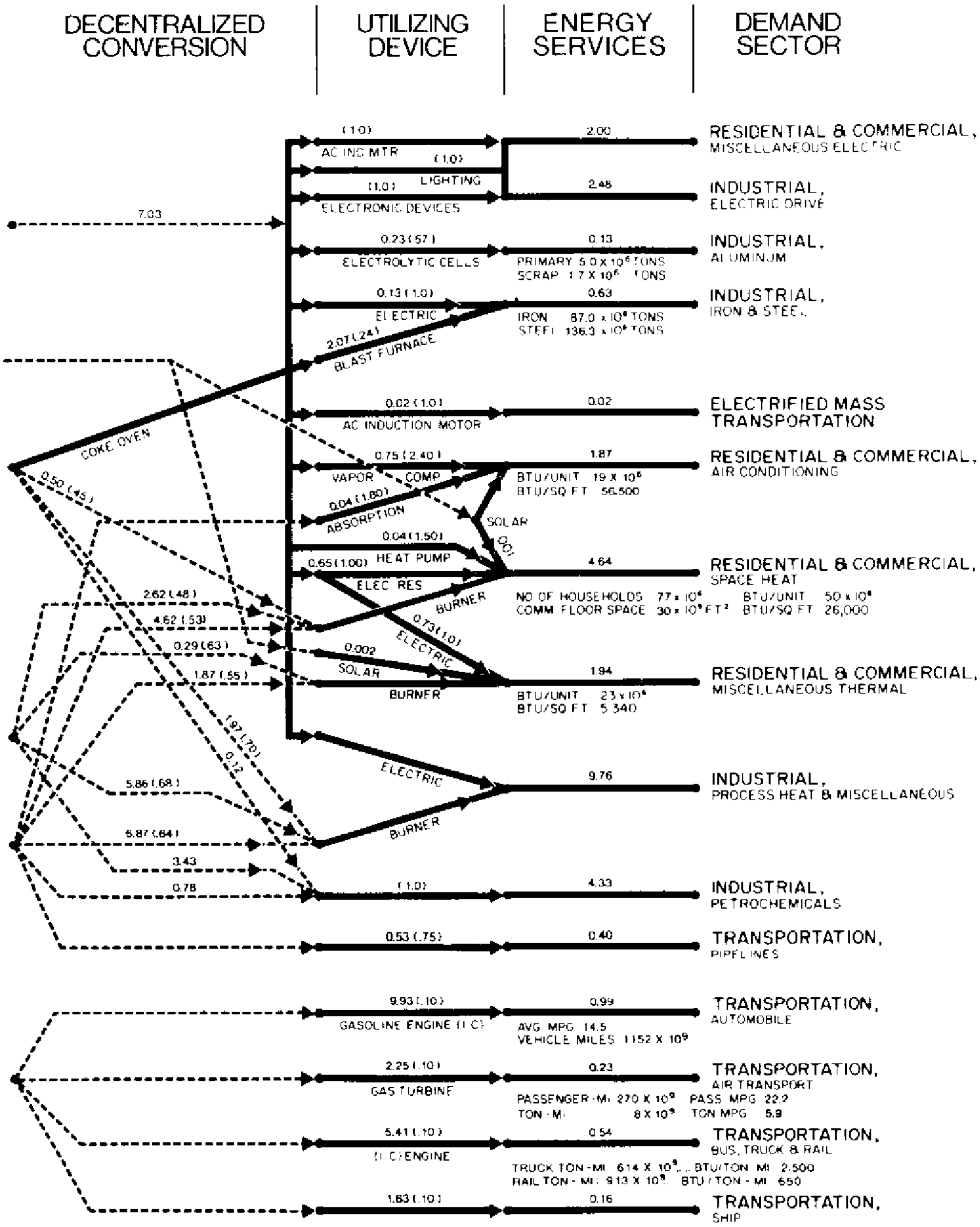


FIG. 2. Reference energy system for the USA in 1979.



30.12 $\times 10^{15}$ Btu

etc., just as much as those who finally apply the statistically developed damage function.("It is not the final blow of the axe which fells the tree.")

Ideally, assessment of energy systems distils in scholarly fashion vast technological and biological research. As part of science and scholarship, the analysis and assessment must be as objective as humanly possible. Impact assessment should not be perverted by special interests to proselytize for one form of energy over others.

Information on which to base risk assessment is often scanty. Even so, organizing data so as to clarify choices among energy options and gauge the uncertainties is a useful aid to decision-making. Since choices must be made from among technologies, standardized comparisons are a must. For technologies producing the same form of energy, e.g. electricity, a standardized unit of production or production rate can be used, such as a 1000-MW(e) power-plant-year. When comparisons extend across technologies based on different energy forms (e.g. coal-electric versus coal-gasification versus coal-liquefaction) the proper comparison is not always obvious. Electricity, gas and oil having the same energy are not equivalent: they serve different choices within the entire energy system for all demands (Fig.2). Our assessment must calculate risk for each component. Valid comparisons can be made only between entire fuel cycles or between entire alternative energy systems.

The great difficulty in risk assessment is how to compare qualitatively different risks. There may be trade-offs among air, water and land impacts; between short-term and long-term risks; or between extraordinary and routine risks. In comparing coal and nuclear-fuel cycles, for example, the high probability of air-pollution impacts from coal (although uncertain in magnitude) must be compared with the low probability of catastrophic events from nuclear energy. Quantitative analysis alone cannot tell how large an assured routine impact one is willing to accept to avoid an unlikely catastrophe. Risk assessment would help separate scientific issues from value issues in these very thorny technological problems, thus helping decision-makers clarify their decisions.

If such assessments are so useful, why is the field so recent? Why do we meet here as the first world-wide symposium? The developing information has been there, but pressures to organize the results of research coherently, i.e. to assess the health risks of different energy sources, are recent. Only just over nine years ago did the Division of Biomedical and Environmental Research of the US Atomic Energy Commission (AEC), anticipating their forthcoming transformation into the US Energy Research and Development Administration (ERDA), evince interest in comparing the health effects associated with alternative energy sources. Just eight years ago was there funding for the work begun at Brookhaven. One reason for this lag was that the AEC was concerned only with nuclear power. Comparative assessments would have required work on energy sources which could have been criticized as outside its jurisdiction.

The historical fact is that the Division of Biomedical and Environmental Research and its predecessors in the AEC, although they had supported most of the important research on radiation, had no programme which integrated their research with assessment of the effects of radiation, and especially the effects from the nuclear fuel cycle. Such necessary assessments – at least of the effects of radiation – were being carried out by national and international bodies of competent experts such as the National Council on Radiation Protection and Measurement (NCRP), the International Commission on Radiological Protection (ICRP), the National Academy of Sciences Committee on the Biological Effects of Ionizing Radiations (BEIR), and the United Nations Scientific Committee on the Effects of Atomic Radiation (UNSCEAR).

In effect, the Division of Biomedical and Environmental Research of the AEC had delegated responsibility for the assessment of the effects of radiation to these national and international panels and supported their activities directly or indirectly. But before World War II, consideration of the hazards of radiation and of radiation protection was itself the self-regulatory response of the medical radiological community to broad social pressures brought to bear by the courts, the news media and insurance companies. UNSCEAR reflected world-wide concern about radioactive fall-out from nuclear tests in the atmosphere. That the hazards of radiation were usually considered apart from the nuclear technology itself may have contributed to the little effect the precise calculations of risk – or the upper limits of risk – seem to have had on people concerned about radiation from the nuclear industry.

Now and in the past, many of those engaged in the development and deployment of energy technologies have been rather indifferent to health assessments, even opposing them because they regarded their results as potentially harmful to vaguely specified interests, and potentially increasing their difficulties in deploying energy sources.

In many countries, health and environmental uncertainties about electric power generation powerfully constrain development of electrical power. Increased global energy needs, coupled with public debate on energy technologies, now powerfully urge assessments of the implications for health of energy sources.

Rational risk assessment has barely begun, and in some ways it must compete with strongly held subjective views commonly held about risk. Yet subjective risk estimation should not be deprecated: living is a continual balancing of risks and benefits. But as the difficulty of choice increases, subjective or individual experience won't do. Economists and psychologists document that public responses to risk and benefit are not clearly rational. This stems partly from limited availability to the public of reliable information. There is also the disparity between mathematical definitions and intuitive notions of risk. Everyday risks appear familiar and personal, hence comprehensible; technological risks, particularly when they involve the general public, seem abstract and

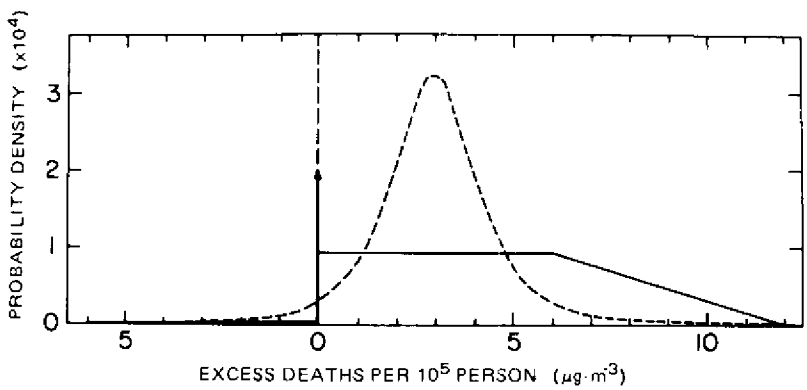


FIG.3. Biomedical and Environmental Assessment Division, Brookhaven, probabilistic damage function (solid line) representing the range of current estimates of sulphate health impact. The dashed curve is the classical estimate (Student's t-distribution with three degrees of freedom) obtained by treating the results of four other regression studies as independent estimates of the same slope [2].

forbiddingly impersonal. The widespread risks of the past – hunger and pestilence – had an aura of comprehensible inevitability. Technological risks are set apart by their controllability: since they involve conscious choices, they mirror existing conflicts of financial and social motives.

With these considerations as preamble, let us turn to some brief examples of how risk assessment has been applied to deal with the difficulties enumerated. Urban mortality has been shown by vast research to correlate with ambient air pollution. But linking damage to this or that component of pollution has emerged as extraordinarily difficult, requiring the elaboration of a host of analytical techniques, physical, chemical and pathological.

The approach we have taken for estimating air pollution impacts from fossil fuel combustion has been through construction of a probabilistic, subjective sulphate health damage function [2]. Sulphate appears to have the most constant association with measured health impact; it is thus used as a surrogate for overall air pollution. This damage function (Fig.3), reconciling the findings of several conflicting studies, expresses the probability that any given linear coefficient represents the real correlation between sulphate dose and excess deaths. It thus provides a way to deal quantitatively with the uncertainty in damage rates in the literature.

For perspective, Fig.4 gives the causes of death in the USA [3, 4]. This shows the proportions of Americans dying of various causes in 1979, including estimated fractions of each linked to smoking and alcohol. The estimates of

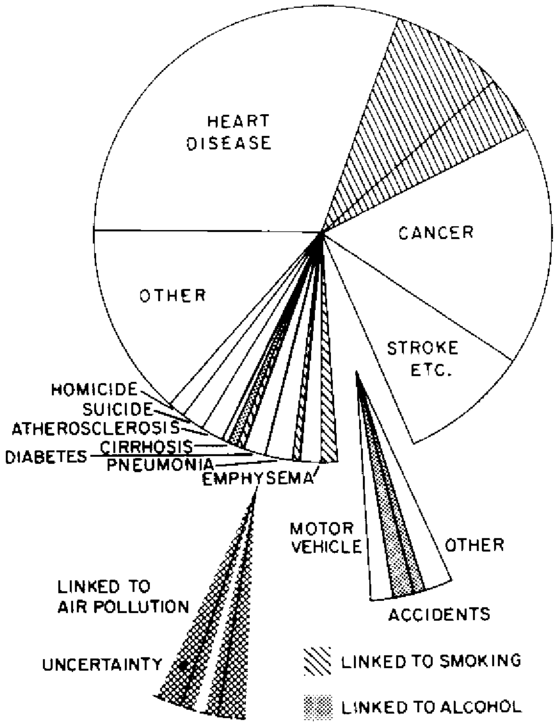


FIG.4. Proportion of Americans dying of various causes in 1979 (data from Ref. [3]), as presented by H. Fischer. Fractions linked to smoking and alcohol, from Ref. [4], are for 1967 but were proportionally assigned here to 1979 data.

deaths linked to air pollution from coal combustion by utility and industrial sources in 1975, using the above sulphate damage function (~50 000 deaths, ranging from 7500 to 120 000 [5]) were not assigned to category of death and so are shown separately.

The next example deals with what I have referred to as the fuel cycle approach. Figure 5 compares two power plants, each capable of producing 1000 MW of electricity with a 65% capacity factor. One is a typical nuclear power plant, the other a coal-fired plant. The areas in the figures are proportional to deaths per year in the USA for each component of the two fuel cycles [6]. Uncertainty is not indicated in the figure but can be found in the original references.

Figure 5 raises several questions that need further consideration. It covers direct fuel cycle effects only: mining, transport, processing and electricity generation. This immediately raises the question of the appropriate boundary

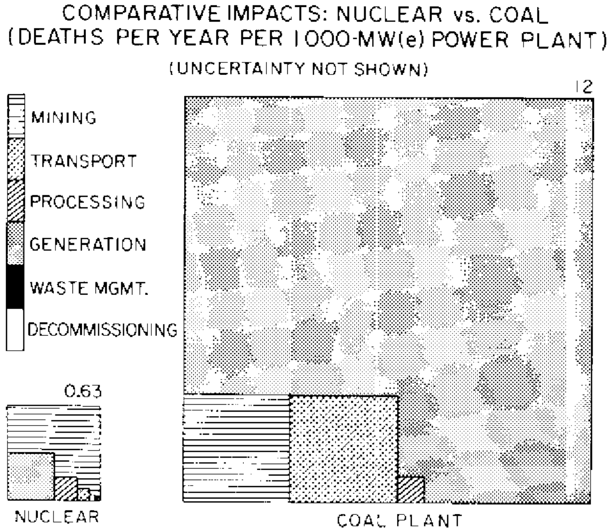


FIG.5. Comparison of aggregated risk from nuclear and coal fuel cycles, normalized to a 1000-MW(e) power plant with a 65% capacity factor. Shaded areas are proportional to deaths per year in the USA for each component of the two fuel cycles [6].

of study and how to assess indirect effects, a topic we come to later. Comparisons between energy sources must use consistent boundaries. Figure 5, as a summary, necessarily aggregates occupational and public deaths. This ignores any differences in loss of life expectancy associated with different causes of death. It also necessarily aggregates on an actuarial basis the effects of rare catastrophic accidents. All these require separate definition and, some might argue, separate treatment and consideration. As already indicated, the figure gives no indication of uncertainty.

The data summed in Fig.5 did not come easily, nor are they to be considered exact. Analysis of public health damage from use of fossil fuel does not have a long history of radiological health studies nor, until more recently, the benefit of comparably accurate measurement techniques.

International comparisons of health impacts can be directly revealing. Figure 6 shows fatal injuries per million miner-hours for underground coal mines [7]. US mines averaged 1.0 to 1.3 deaths/10^6 person-hour until after passage of the 1969 Coal Mine Health and Safety Act (CMHSA), which was followed by a sharp drop in the mortality rate. Note that productivity levels (tonnes produced per miner-hour) also dropped after passage of CMHSA, cancelling out most of the gain in terms of fatalities per tonne of coal produced.

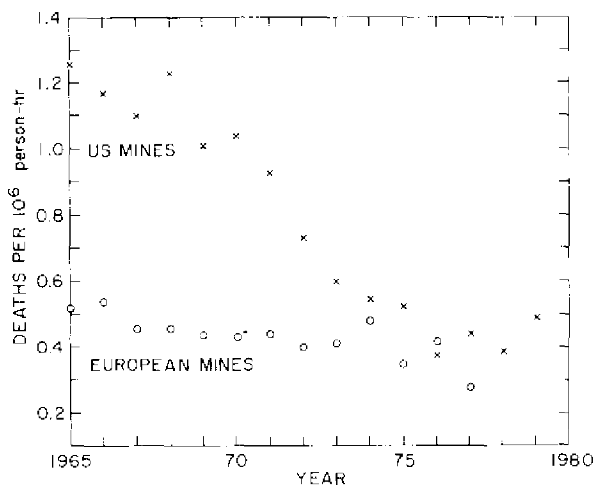


FIG.6. Trends in mortality rates for US and European coal-mining experience, 1965–1979 [7].

Clearly, health effects of different energy sources depend directly on the quantity and type of energy use. These vary greatly according to geography. Thus Fig.7 shows the share of fuel-wood in the total energy consumption of various countries and global regions. Fuel-wood is the fourth largest contributor to world energy supply after petroleum, coal and natural gas. Close to half the world population, especially those living in rural and urban areas of developing countries, depend for cooking and heating on fuel-wood, its derivative, charcoal, or, in the absence of either of these, on agricultural residues or dung. Fuel-wood constitutes more than half the world's consumption of wood, rising to 86% in developing countries and 91% in Africa [8].

Our studies of health effects of fuel-wood use in the USA, like our analysis of other energy sources, were based on the health effects of supplying a unit amount of energy (e.g. that needed to provide heat for 10^6 dwelling-years). Health effects were estimated in gathering, transport and combustion stages [9]. Oil and coal alternatives were scaled from earlier work (Fig.8). Figure 9 brings those rather abstract estimates down to the level of an individual. We calculated the total risk for a single dwelling unit as though faced by a single individual. The risk of death to this individual associated with the gathering, transport and burning of wood as the sole source of heat for his home over 40 years is expressed graphically as the volume of a cluster of cubes [9]. The total risk for the whole period is ~0.2%. This can be compared with the risk of death in an automobile accident during the same period of ~1%.

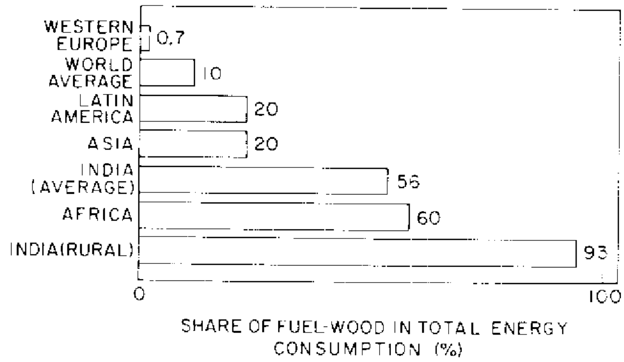


FIG. 7. Share of fuel-wood in the total energy consumption of various countries and regions [8]. India's share of fuel-wood includes agricultural residues and dung.

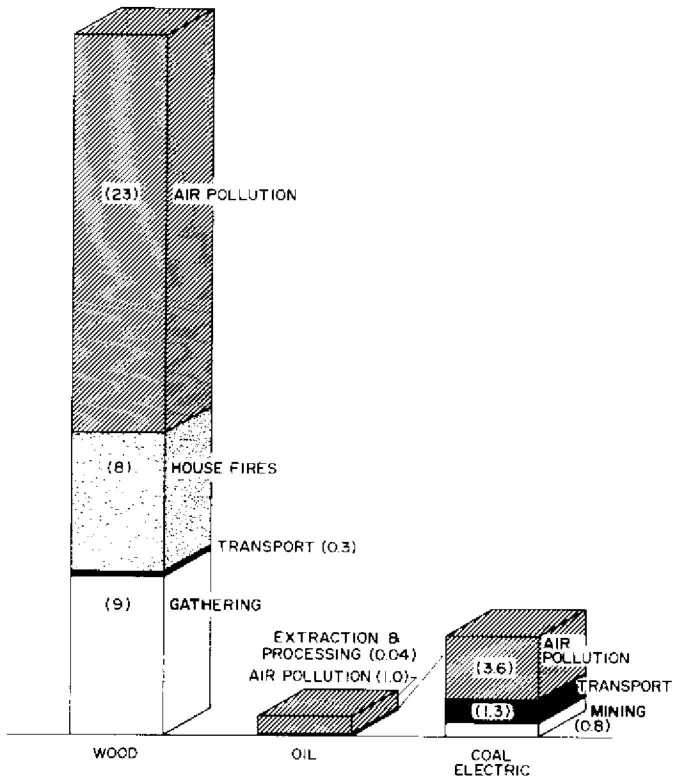


FIG. 8. Comparison of health effects of wood, oil and coal supplying space heat for 1 million dwelling years (6×10^7 GJ heat or 8.8×10^7 GJ wood input at 69% efficiency) [9].

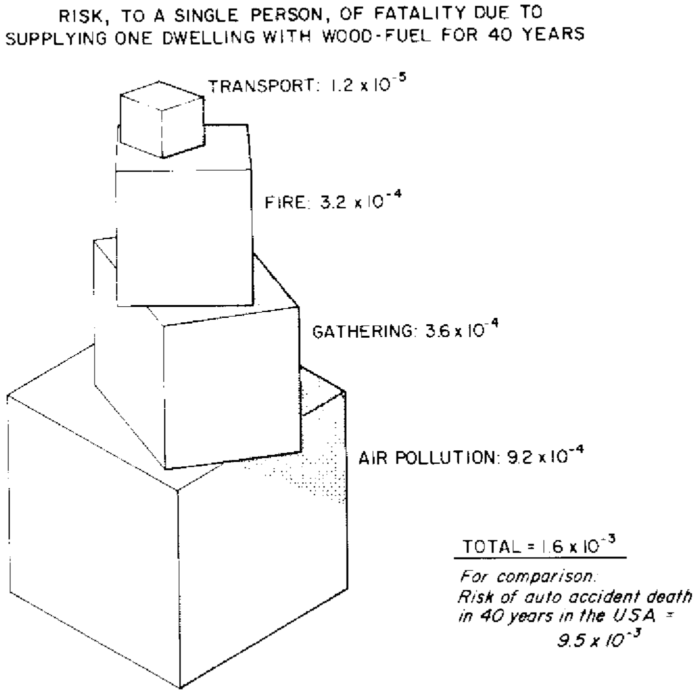


FIG. 9. Equivalent individual risk of death, by cause, from 40 years' operation of a residential wood stove [9].

In assessing the health effects of different energy sources we must take into account comparison of societal and individual occupational risks for different fuel cycles. Figure 10 shows how comparative risk to individuals from energy technologies per unit of time exposed can differ from the relative risk per unit of energy produced [10]. Wood pyrolysis is more dangerous to workers in the industry than residential photovoltaics, in this example, but it requires far less labour per Btu delivered and accordingly has much lower occupational impacts per unit of energy output. These are direct impacts, but system-wide impacts show similar behaviour.

In considering new and renewable sources of energy now receiving attention, such as solar energy, wind power and alcohol fuels, the idea of a 'fuel cycle' needs expansion to encompass industries such as solar photovoltaics, where virtually no risk attaches to the operation of the device but a substantial amount – often ignored – may be involved in its *manufacture*.

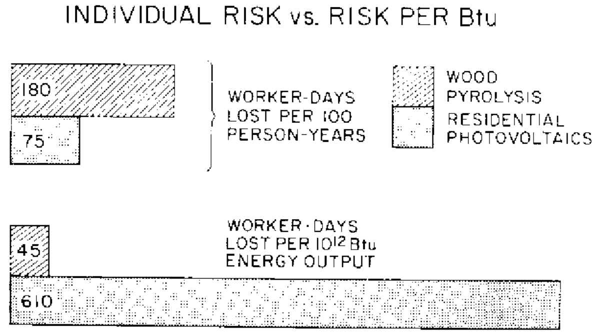


FIG.10. Occupational risk per unit of time exposed versus risk per unit of energy produced. Wood pyrolysis compared with residential photovoltaics [10].

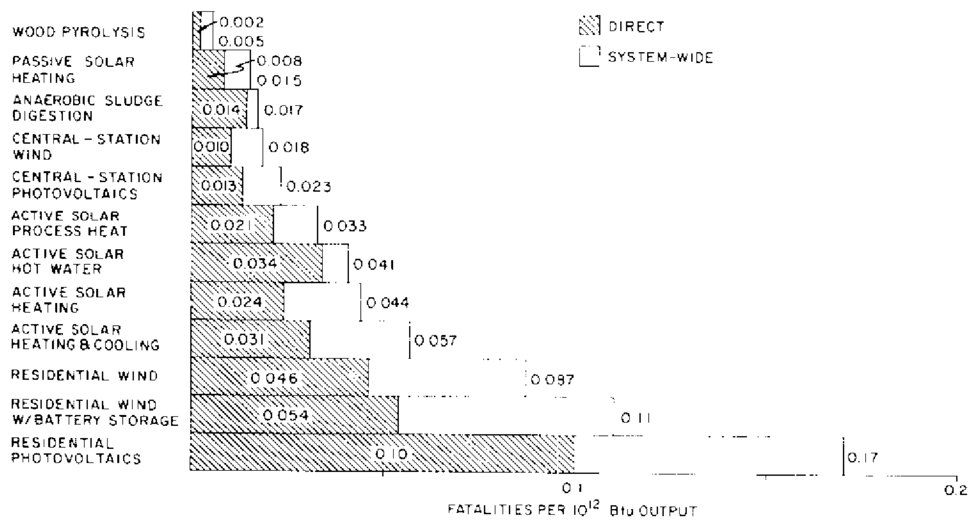


FIG.11. Direct versus system-wide occupational fatality risk from renewable energy technologies [6, 10, 11].

Attempts were made to deal with this difficulty by expanding the fuel cycle concept to include all stages in developing an energy system, including the concrete that goes into the plant that manufactures the glass for the solar collector. The issue of completeness has been addressed by noting that the backward analysis of manufacturing steps is equivalent to a set of simultaneous equations whose solution, if linear, is expressible as a matrix of values. Such an approach is familiar to economists as input-output analysis; and the appropriate

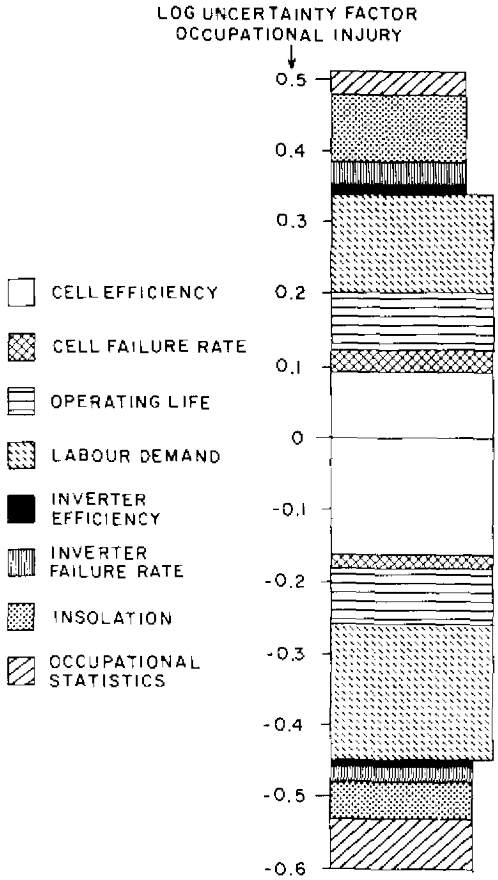


FIG.12. Uncertainty factors in occupational injury risk from projected photovoltaic technology. The central portion represents factors intrinsic to the technology itself [12].

numbers, showing how much each economic activity draws on the others, have already been derived, although for aggregate categories this may not exactly match the manufacturing steps one wishes to scrutinize for measuring health damage. The result of this sort of calculation is shown in Fig.11 for the direct and system-wide occupational health risks of renewable energy technologies. The figure compares the direct occupational impacts (fatalities) from a variety of renewable energy technologies with the total impacts when the entire range of supporting activities are included in the impact analysis [6, 10, 11].

A final example in our survey assessing the health effects of different energy systems is the meaningful inclusion of uncertainty in the estimates. Figure 12 displays some sources of uncertainty in estimating occupational health effects (in terms of worker-days lost) of photovoltaic energy systems [12]. Included are estimates of both extrinsic sources of uncertainty (insolation, occupational statistics, inverter system) and intrinsic sources (solar cell efficiency, lifetime, failure rate, labour requirements). This was based on high, low and expected estimates of the various factors in photovoltaic energy systems. The figure serves as an example and inevitably cannot include all uncertainty. The indicated uncertainty from these sources can modify median base case estimates of occupational injury estimates upward by a factor of ~3.3 or downward by ~4.0. If extrinsic sources of uncertainty are eliminated, uncertainty is still quite high, ~2.2 upward or ~2.8 downward. Of the intrinsic sources of uncertainty examined, significance increases as follows: failure rate, lifetime, efficiency and labour demand. Other sources of uncertainty (e.g. effects of exposure to toxic chemicals) are not covered in this figure.

No single method of comparative risk assessment in the energy industry is fully satisfactory by itself; each has its limitations. One needs to compare several methods if one is to improve the decision-making base for policy-makers and the public.

In conclusion, energy risk analysis has not yet progressed to where biological damage mechanisms are well enough understood, but, by statistical associations, degrees of impacts can be estimated. Progress has been made in how to compare multiple alternatives for energy generation and use. As these efforts continue, it becomes increasingly likely that the fruits of this discipline, entering into the public consciousness, will influence decision-makers.

I appreciate the honour the three agencies have paid me by the opportunity to open this historic meeting. As a physician I am convinced that one only lives once. I hope you will enjoy and profit from this meeting.

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Session I

EPIDEMIOLOGICAL PARAMETERS IN EVALUATION OF HEALTH IMPACTS

Chairman
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United States of America

COMPARATIVE RISKS OF HYDRAULIC, THERMAL AND NUCLEAR WORK IN A LARGE ELECTRICAL UTILITY

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Abstract

COMPARATIVE RISKS OF HYDRAULIC, THERMAL AND NUCLEAR WORK IN A LARGE ELECTRICAL UTILITY.

Data is presented on the fatalities and accidents that have occurred in the construction and operation of hydraulic, thermal and nuclear generating facilities brought into service and operated by Ontario Hydro, a large Canadian utility, in the period 1970 to 1979. Results are also given of a prospective cohort epidemiological study of thermal and nuclear station workers which was begun in 1974.

1. Introduction

Ontario Hydro is the electrical utility of the Province of Ontario, Canada. It is a publicly owned utility employing approximately 28 000 persons in the design, construction and operation of electrical generation, transmission and distribution facilities. As the corporate name implies, electricity generation by water power was the only mode of operation for many years, but this has changed and power is now produced from nuclear and fossil fuel as well as hydraulic sources in about equal quantities.

In recent years, there has been much interest in the risk associated with power generation by various means. This paper presents information on occupational fatalities and injuries resulting from operation of hydraulic, thermal and nuclear stations over the period 1970 to 1979 and similar information for the construction of facilities brought into service in that period.

2. Power Production and Installed Capacity

Currently, one-third of the power generated is hydraulic, one-third fossil and one-third nuclear. This is quite different from the contribution from these sources in 1970 when hydraulic generation was the major contributor with thermal supplying the remainder.

TABLE I. STATISTICAL DATA ON CAPACITY, POWER PRODUCED AND MANPOWER BY SOURCE

Power production mode	Net capacity installed 1970–1979 (MWe)	Power produced[c] 1970–1979 (GWe·a)	No. of employees 1979
Hydraulic	574	41.2	900
Thermal	8040[a]	31.0	2125
Nuclear	7084[b]	17.7	2393

[a] Excludes combustion turbines.
[b] Includes Pickering NGS 'B'.
[c] Includes power from all sources: not just those in column 2.

Table I gives the number of employees engaged in each mode of power production in 1979, the net power produced from each source over the 10-year period 1970 to 1979 and the net capacity installed between 1970 and 1979. In the case of nuclear generating stations, the number of staff excludes Head Office support staff and employees at our heavy-water plants.

The number of employees in hydraulic generation is an estimate as only the very large plants have operators and their own maintenance staff. The number of employees in the nuclear plants is currently greater than is necessary for the running of the plants by about 10%. The increased numbers are due to training needs during a period of rapid and continued expansion of the nuclear program.

Table II gives the period of construction and the number of man-years for the construction of the major thermal and nuclear facilities brought on line in the period 1970 to 1979.

3. Accident Data - Operations

Experience in hydraulic, thermal and nuclear generation is given in Table III. The data for hydraulic is not truly representative because many of the smaller stations are remotely controlled. As such, employees are engaged in both station maintenance work and distribution system work. The nuclear and

TABLE II. STATISTICAL DATA ON SOME MAJOR FACILITIES BROUGHT INTO SERVICE AFTER 1970

	Generating Facility	Net Capacity (MWe)	Period of Construction	Number of Man-years to Construct [a]
NUCLEAR	Pickering 'A'	2060	1965 - 1972	1.1×10^4
	Pickering 'B' [b]	2064	1974 - 1982	1.4×10^4
	Bruce 'A'	2960	1969 - 1977	1.7×10^4
THERMAL	Lambton	1980	1965 - 1970	0.5×10^4
	Nanticoke	3920	1968 - 1978	1.1×10^4
	Lennox	2140	1970 - 1977	0.6×10^4

[a] Ontario Hydro construction and contractors.
[b] Not currently in service – man-years estimated to completion of project.

TABLE III. ACCIDENT EXPERIENCE IN OPERATION OF GENERATING STATIONS

Accident Classification [a]		1970 - 1979		
		Hydraulic	Thermal	Nuclear
Fatalities	Number No./10^8 manhours No./GWe·a Severity Rate [b]	1 6.2 0.024 419	None has occurred	None has occurred
Permanent Total Disabilities	Number No./10^8 manhours No./GWe·a Severity Rate	None has occurred	None has occurred	None has occurred
Permanent Partial Disabilities	Number No./10^7 manhours No./GWe·a Severity Rate	1 0.6 0.024 7.0	1 0.3 0.032 5.9	3 0.9 0.170 6.2
Temporary Total Disabilities	Number No./10^6 manhours No./GWe·a Severity Rate	38 2.4 0.92 Not available	242 7.1 7.8 146	113 3.4 6.4 45

[a] Classified according to ANSI Z16.4 (method of comparison of occupational accidents).
[b] Severity Rate is defined as days of work lost per 10^6 man-hours worked.

TABLE IV. ACCIDENT EXPERIENCE IN CONSTRUCTION OF GENERATING STATIONS

Accident Classification		Nuclear	Thermal
Fatalities [a]	Number	6	5
	No./10^8 manhours	9.4	11.5
	No./GWe installed	0.85	0.62
	Severity Rate	565	687
Permanent Total Disabilities	Number	none has occurred	none has occurred
	No./10^8 manhours		
	No./GWe installed		
	Severity Rate		
Permanent Partial Disabilities	Number	2	9
	No./10^7 manhours	0.4	3.2
	No./GWe installed	0.28	1.1
	Severity Rate	15.8	60.6
Temporary Total Disabilities	Number	991	393
	No./10^6 manhours	18.0	14
	No./GWe installed	140	49
	Severity Rate	535	411

[a] Fatality data is for both Ontario Hydro and contractor staff. Other data is for Ontario Hydro staff only.

thermal station data is of good quality and is fully representative because these stations have a relatively constant number of employees.

4. Accident Data - Construction

We have obtained accident data for the construction forces that worked on our major facilities brought into service during the 1970 to 1979 period but it should be noted that the data is neither as accurate nor as complete as that recorded for operation of facilities. This is partly due to the work force being transient in nature and also because of the considerable number of contractors who work on construction projects.

Combined fatality statistics for contractors and Ontario Hydro staff are given in Table IV and cover the periods when the projects were under construction.

TABLE V. ONTARIO HYDRO STANDARDIZED MORTALITY RATIO

	Nuclear	Thermal	Other	Total
All causes	54	73	84	83
Neoplasms (cancers)	(62)[a]	119	91	92
Circulation (heart, stroke)	(36)	69	93	91
Accidents (occupational and non-occupational)	82	51	74	73
All other causes	11	52	55	54

NOTE: Observed deaths expressed as a percentage of predicted deaths (based on cumulative experience 1970–1979).

[a] Parentheses indicate results based on less than ten observed deaths.

Injury statistics are not available for contractors and are based on Ontario Hydro employees only. As such, the statistics have been significantly influenced by vigorous safety management on the part of Ontario Hydro project managers. If contractor injuries were taken into account, the results would be increased approximately by a factor of three.

5. Health Effects - Operations

In addition to information on accidents, information is available from an epidemiological study of employees and pensioners. Begun in 1974 as a prospective cohort study, it was made retrospective to 1970. The data analysis and biennial publication of results is by an independent consultant (Professor Terence W. Anderson, University of British Columbia). The comparison groups in the study are Thermal Generation Division employees, Nuclear Generation Division employees, other employees of Ontario Hydro, and the general population of the Province of Ontario. Hydraulic plant employees are encompassed in the "other employees" category. Construction staff, because of their transient nature, are generally not included in this study, apart from a few who are regular staff. A summary of the deaths in these categories is given in Table V.

TABLE VI. FATAL ACCIDENT RATES (No./10^8 man-hours) 1970–1979

Ontario Hydro Generating Stations	Hydraulic Plants 6.2	Thermal Plants 0	Nuclear Plants 0	
Canadian Utilities	Manitoba Hydro 4.8	Ontario Hydro 9.9	British Columbia Hydro 10.8	New Brunswick Electric Power Commission 11.5
American Utilities	Niagara Mohawk Power Corporation 1.9	Tennessee Valley Authority 4.7	Detroit Edison Company 6.0	Virginia Electric and Power Company 11.2
American Industries (overall = 7.7)	Trade 3.2	Manufacturing 4.4	Agriculture 29.2	Mining, Quarrying 38.9
Ontario Hydro Major Work Groups	Foresters 9.7	Mechanics 13.5	Electricians 16.4	Linemen & Groundmen 63.6

6. Discussion of Results

6.1. Accidental Fatalities and Injuries - Operations

The fatality risk in the three categories: hydraulic, thermal and nuclear generation, is low and compares favourably with other major work groups in Ontario Hydro, with Canadian[1] and American[2] Utilities and with industry in North America[3]. This is shown in Table VI.

Only hydraulic generation had a fatality in the ten-year period. Thermal generation had no fatalities among Ontario Hydro employees, although they had one fatality - a contractor's employee who was killed during maintenance work in 1976. This fatality is not included in Table III as contractor hours during operation are few and are not recorded.

6.2. Accidental Fatalities and Injuries - Construction

The fatal accident rate for construction of the major facilities which were brought into service in the period from 1970 on occurred mainly among contractor employees. The results are shown in Table IV. Data are not available for the limited hydraulic station construction after 1970 but this should not be dissimilar to experience with nuclear stations on a per-manhour basis.

The fatal accident rate is low in comparison with the fatal accident rate in U.S. construction, which is 31.5 per 10^8 manhours [3] and is about the same value as the fatal accident rate for construction in Ontario generally [4].

6.3. Delayed Effect Risk 1970 – 1979

There were 1656 deaths among Ontario Hydro employees and pensioners, with the great majority being pensioners.

The conclusions that we can draw from our epidemiological data results are limited by the length of the exposure time - 30 years for thermal and 19 years for nuclear - too brief for most of the nuclear employees to show ill effects because the majority of radiation exposure has occurred only since 1969. In fact, the number of deaths in the nuclear employee group is quite low, with correspondingly high statistical uncertainty.

Deaths due to cancer among nuclear-plant employees have been fewer than predicted on the basis of experience with the general Ontario population of the same age and sex ratio. We have only recorded 6 fatalities compared to a predicted number of 9.7 (Standardized Mortality Ratio (SMR) = 62%). Over the past ten years, the total number of deaths among nuclear employees (32) has been about half of what would be expected (58.9) among Ontario males of the same age distribution. This very favourable experience is due in large part to the "healthy worker effect" particularly noticeable in young employee groups. Also important is the fact that radiation exposures have been kept low to date, with only four employees in the occupational lifetime exposure category of more than 0.5 Sv. The total population dose to the end of 1979 is 265 man-Sv.

Thermal plant workers have also had a relatively good mortality experience over the past ten years although the "healthy worker effect" is not as striking because of their somewhat older age distribution. The SMR for cancers is slightly elevated in the thermal group and although this may be the result of small number variation (observed 23, predicted 19.3) there is also the possibility that it reflects exposure to substances such as asbestos in earlier years. This is presently under investigation.

7. Conclusion

Results to date in Ontario Hydro show that the risks to employees of generating electricity by hydraulic, thermal or nuclear means are similar and are low in comparison with general industrial experience in North America. Exposure to radiation has not resulted in an elevated Standard Mortality Ratio, but this may change when sufficient time has elapsed for full expression of delayed effects and when the number of fatalities are larger and less subject to statistical fluctuation.

Fatalities during construction are similar on a per-manhour basis for each power production mode.

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DISCUSSION

Y. NISHIWAKI: It is very important to collect data on occupational hazards in order to assess the overall risks associated with hydro, thermal and nuclear electricity generation. What is the ratio of fatalities to total investment in the capital construction of these systems of electricity production?

T.R. HAMILTON: Six fatalities occurred during the construction of nuclear plants involving an investment of approximately US $\$2.5 \times 10^9$, giving a ratio of about US $\$4 \times 10^8$ per fatality. In the case of thermal plants, there were five construction fatalities during expenditure of approximately US $\$1.6 \times 10^9$, or about US $\$3 \times 10^8$ per fatality.

I.M. TORRENS: Do you consider the statistics you have given for Ontario Hydro as in some way representative of utilities in North America, and have any comparisons been made with similar data from other utilities?

T.R. HAMILTON: Comparisons are made with occupational accident experience of utilities in North America on the basis of ANSI. However, other utilities in North America have not performed epidemiological studies and hence have no estimates for the delayed risk. Another difficulty in comparing Ontario Hydro with other North American utilities is that most of the latter are small, generate power usually only by one mode, do not carry out their own construction work and largely distribute the power they generate directly to customers. Ontario Hydro, however, operates on a wholesale basis, transmitting the electricity it generates to about 500 distribution utilities and large industrial customers.

C.E. ALMEIDA: Did you encounter any cases of occupational overexposure which could have given rise to a delayed effect? If so, how would you fit these into the present risk analysis?

T.R. HAMILTON: In the twenty years since our first nuclear station started operation, there have only been a few exposures in excess of the statutory annual dose limit of 50 mSv. The highest of these was a single dose of about 80 mSv. These exposures exceeding the limits have no more significance in terms of delayed effects than the doses received in normal operation. The latter are on average about 6 mSv per year and can be fitted into the present risk analysis either by using a fatal risk factor of about 10^{-2}/Sv or by reference to the results of our epidemiological study given in the paper.

SPATIO/TEMPORAL ANALYSES OF HUMAN BIRTH WEIGHT: AN INDICATOR OF SUBTLE ENVIRONMENTAL STRESS?*

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Abstract

SPATIO/TEMPORAL ANALYSES OF HUMAN BIRTH WEIGHT: AN INDICATOR OF SUBTLE ENVIRONMENTAL STRESS?

Analysis of variance (ANOVA) of birth weights in the Chicago area (1970–1972) and a 50% sample of all US births (1970) showed trends in mean birth weight that were similar to those reported in the literature and consistent temporally and spatially among the groups studied. Mother's age, parity, location, sex, and the interaction of mother's age and parity are the major determinants of mean birth weight. A second ANOVA with month of birth, location and sex also shows strong main effects and little interaction. Although the major determinants are of little predictive value for an individual birth weight, their highly consistent effects on the population mean suggested that the study of population mean birth weight might be useful in identifying subtle effects such as those of environmental pollution. To test this hypothesis, Niagara County, New York, was compared with four surrounding counties and with a group of nine other New York counties selected by cluster analysis to be similar on a set of demographic and socio-economic attributes. In both cases, mean birth weight in Niagara County was significantly less than other counties in the comparison. These results suggest that population mean birth weight may be a sensitive indicator of environmental stress.

INTRODUCTION

The present study examines the utility of birth weight records as a data base from which the health status of human populations may be monitored. These data are attractive for a number of reasons. First, birth weight trends are of interest from a public health standpoint because birth weight is perhaps the most important conditioning factor for infant survival [1-3] and an indicator of future growth and development [4]. Birth weight also offers a number of practical advantages as compared to more commonly employed barometers of population health status such

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as cause of death or life expectancy. These variables are the product of personal habits, occupation and environmental exposure to toxic agents interacting over a long period of time [5]. Thus, the extent to which trends in such variables reflect the present environment is problematical. Mortality measures are also affected by errors in the ascertainment of cause of death [6,7]. Comparatively, the birth weight is a simple determination.

There is also a substantial literature that suggests birth weight is responsive to a number of intrinsic and extrinsic environmental factors. These include mother's age [8-11], parity [8-13], prepregnant weight [12,14-16], child's race and sex [4, 10,15-18], maternal smoking and chronic alcoholism (both of which can reduce birth weight) [19-21], and the level of maternal health care or standard of living [9,22,23]. Of particular importance to the use of birth weight as an environmental monitor is the fact that it appears to be responsive to environmental stresses such as air pollution [24-27].

Finally, from a practical standpoint, birth weight data are routinely collected and are readily available in computer-readable form.

Two questions are of central importance to the possibility of using birth weight as a health status monitor. First, can one discern consistent effects of known influencing factors, such as maternal age, while controlling for only a small number of other influencing factors? This is of importance because standard birth certificates provide only a limited number of collateral variables in addition to weight. If "known" effects are obscured by noise from unknown sources, the effects of environmental stress may likewise be obscured. Second, if known effects are discernable, do known stressed populations show predictable patterns of depressed birth weight?

MATERIALS AND METHODS

The question of whether known influencing factors can be consistently delineated was addressed in the first part of this study, in which the effects of several major determinants of mean birth weight were assessed in two separate data sets which were available for an overlapping time period. One, provided by the Illinois Department of Public Health, was a three-year compilation (1970-1972) of all births in the six-county (Cook, DuPage, Kane, Lake, McHenry and Will) Chicago Standard Metropolitan Statistical Area (SMSA). The second set of data, obtained from the National Center for Health Statistics, was a sample of 50 percent of all U.S. births in 1970 (termed Detailed Natality Data). Both series

(Chicago SMSA and Total U.S.) included records of birth weight, place of birth, age of mother, parity, and child's sex and race and limited information on the mother's prenatal medical history. Both data sets are compiled routinely from information collected by the states. In the case of the Detailed Natality Data, the information from all states is collated by the National Center for Health Statistics, using only every other record received from each site, hence a 50 percent sample. Multiple births were excluded from the Chicago SMSA, but not the total U.S. data, because federal coding procedures in 1970 did not permit their identification. However, multiple births account for less than two percent of all births [17] and examination of the Chicago SMSA data for the effects of multiple births demonstrated the minor influence of these births on the overall population measure. Only births to white mothers were considered. In both sets of data, births to foreign residents (< one percent of the data) and records with invalid or unknown birth weight (< one percent of the data), age of mother, or birth order (parity) were eliminated from the analyses. Nevertheless, approximately 1.5 million records (791 508 males and 747 594 females) were included in the Total U.S. group and nearly 255 000 records (131 082 males and 123 694 females) in the Chicago SMSA group.

Our first analysis of these data considered two of the most readily available, frequently studied, and influential factors affecting birth weight: age of mother and parity [11,14,19,28]. All comparisons are for white births grouped by sex and location. Statistical analyses were performed using the procedure for the analysis of variance from the Biomedical Computer Programs [29] and the general linear model program from the Statistical Analysis System [30]. Grouping variables for the analysis of variance (ANOVA) were mother's age in years (15-19; 20-24; 25-29; 30-34; 35-39; 40-45), sex of child (male; female), parity (1; 2; 3; 4; 5; 6+), and location (Chicago SMSA; Total U.S.). An additional temporal comparison was conducted for month of birth (1 through 12).

Part two of our study made an evaluation of whether stressed populations display low mean birth weights. This portion compared Niagara County, New York, site of an extensive chemical industry [31] and numerous chemical dump sites, including Love Canal [32], with (a) four surrounding counties (Table IIIA) and (b) a group of nine other socioeconomically and demographically similar New York counties (Table IIIB). The second comparison was made to control for other possible influencing factors as described in the introduction. In it, selection of counties was made from all New York counties using a single linkage cluster analysis procedure from the BMDP-79 statistical package [33]. Characteristics upon which the clusters were derived included: proportion of the population receiving aid to dependent children, proportion of single-family houses, proportion of families earning less than $3000, population per square mile, median dollar value of single housing units, per-capita dollar expenditures, proportion of men 15-65 years old employed in professional or administrative positions, proportion of men 15-65 years old employed in agriculture (farmers and farm laborers), and the average month when prenatal care began for first and second births to mothers 21-30, in 1971.

In both comparisons, males and females were considered separately because the former are known to be heavier. Statistical analyses were performed using a variant of Friedman's rank sum test on randomized blocks [34]. Rank tests were employed because of their insensitivity to outliers and heterogeneity of within-group variance. For this comparison, within-sex mean county birth weights for each of four contiguous two-year periods from 1968 to 1975 were calculated. To minimize confounding influences, all were based on only first and second births for white mothers aged 21 to 30. Because two sexes and two sets of control counties were involved, a total of four analyses were performed. In each analysis, mean weights were rank-ordered within each two-year period and ranks were summed across periods. Our procedure tested a null hypothesis that the rank of Niagara County was assigned at random within time periods versus an alternative hypothesis that Niagara County had lower rankings than would be expected at random. The cumulative density function for this rank sum test was calculated using a simple BASIC program written by the second author.

The same rank sum test was also used to examine the possibility that sex ratio might be skewed in favor of females in Niagara County relative to the control. This examination was motivated by observations of other workers which suggest that males are subject to higher in utero mortality and are in general more sensitive to environmental insult than females [35,36].

RESULTS

The trends found in mean birth weight with mother's age and parity, categorized by sex and location are presented in Figure 1. Females are generally lighter than males and Chicago infants are lighter than those in the corresponding Total U.S. group. It appears that, despite the large effects of sex and location on mean birth weight, the pattern of variation with mother's age and parity is similar in all four groups (Figure 1, A-D). This conclusion is supported by the four-way factorial analysis of variance [37] of these data. Given the very large sample sizes involved, conventional tests of significance are uninformative, because even very small effects will tend to be significant [38]. However, the

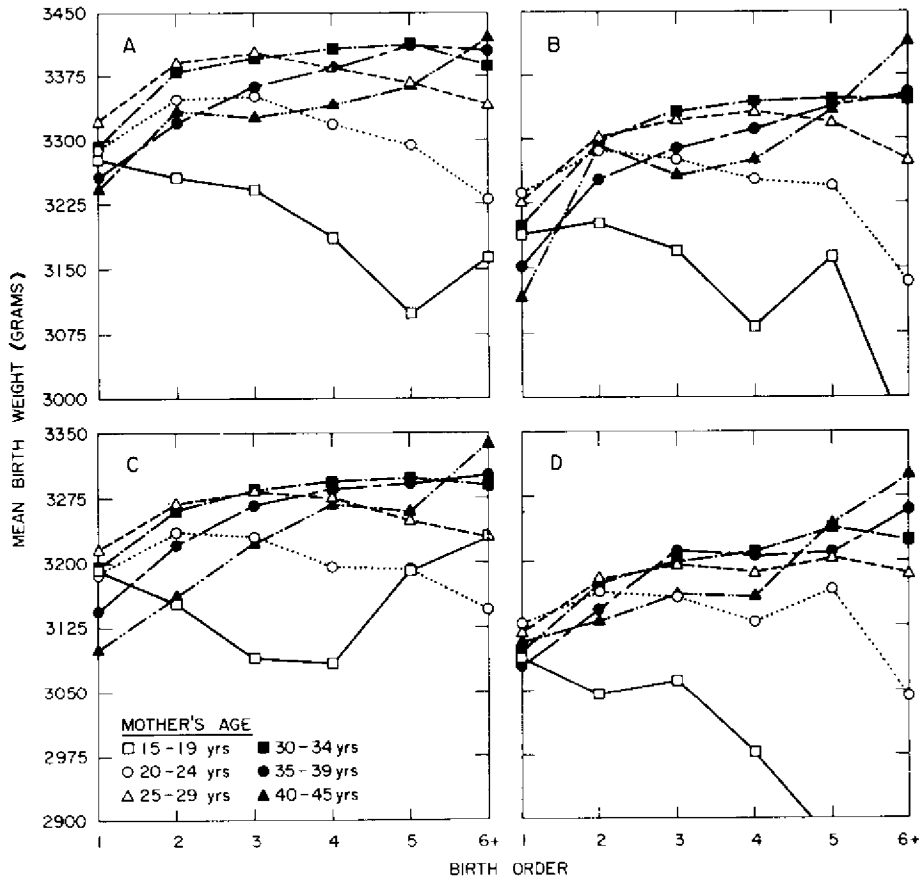

FIG.1. Mean birth weight for white infants by birth order and mother's age. (A) US males, 1970; (B) Chicago SMSA males, 1970–1972; (C) US females, 1970; (D) Chicago SMSA females, 1970–1972.

partitioning of the sums of squares and the resulting mean squares provide insight as to the relative importance of the independent variables and their interactions. The ANOVA verifies that sex, location, mother's age, parity, and the interactions of mother's age and parity show substantial effects. However, the interaction of sex and/or location with mother's age and/or parity accounts for less than 2% of the variation in mean birth weight. The finding of the nonsignificance of the interaction terms (Table I) is particularly noteworthy, given the sample size of nearly two million observations.

TABLE I. ANALYSIS OF VARIANCE (ANOVA) OF WHITE BIRTH WEIGHTS
(Grouping variables are: mother's age (15–19, 20–24, 25–29, 30–34, 35–39 and 40–45); parity (1, 2, 3, 4, 5 and 6+); location (Chicago SMSA, 1970–1972, or USA, 1970); and sex (male or female); $R^2 = 0.02$)

Source of Variation	Degrees of Freedom	Sum of Squares	% Total among Groups	F Ratio
1. Mother's Age	5	2076.1473	0.210	1566.90
2. Parity	5	714.0530	0.072	538.91
3. Location	1	1136.0427	0.115	4286.95
4. Sex	1	5123.0473	0.519	19332.25
5. Mother's Age x Parity	25	686.5024	0.070	103.62
6. Mother's Age x Location	5	31.2801	0.003	23.61
7. Mother's Age x Sex	5	27.1362	0.003	20.48
8. Parity x Location	5	13.9475	0.001	10.53
9. Parity x Sex	5	19.4389	0.002	14.67
10. Sex x Location	1	3.5465	0.000	13.38
11. Mother's Age x Parity x Location	25	13.8339	0.001	2.09
12. Mother's Age x Parity x Sex	25	12.8166	0.001	1.93
13. Mother's Age x Location x Sex	5	1.7498	0.000	1.32†
14. Parity x Location x Sex	5	1.8122	0.000	1.37†
15. Mother's Age x Parity x Location x Sex	25	6.9171	0.001	1.04†
16. Total among Groups	228	9868.2715		163.33
Error	1741350	461399.2272		

† Not significant ($p > 0.05$).

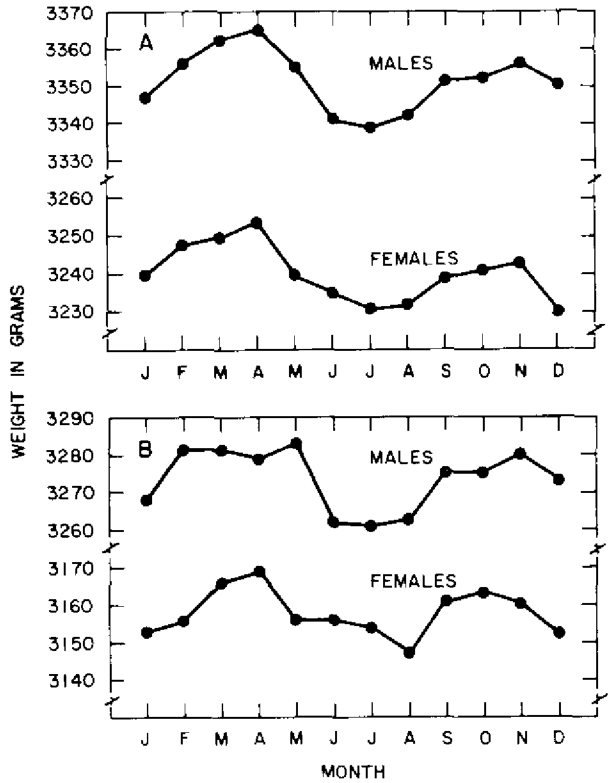

FIG.2. (A) Mean birth weight for US whites, 1970, by month of birth and sex; (B) mean birth weight for Chicago SMSA whites, 1970–1972, by month of birth and sex.

Figure 2 illustrates trends in mean birth weight by sex and location over the year. This figure shows that infants born in late spring or early summer are heaviest, whereas those born in mid to late summer are lightest. Table II, an analysis of this figure, has as its grouping variables sex, location and month of birth. The separation of factors into a four-way and a three-way ANOVA rather than one five-way ANOVA is dictated by the fact that the very large number of cells in the latter (6 x 6 x 2 x 2 x 12 = 1728) would have resulted in empty cells, rendering model estimation difficult [39]. Again, we find relatively large main effects but little evidence of any major inconsistencies of mean birth weight trends with either sex or location.

Results of the comparison of Niagara County with surrounding counties and to selected controls are presented in Tables III to V.

TABLE II. ANALYSIS OF VARIANCE (ANOVA) OF WHITE BIRTH WEIGHTS
(Grouping variables are month of birth (Jan. – Dec.); location (Chicago SMSA, 1970–1972, or USA, 1970); and sex (male or female); $R^2 = 0.015$.)

Source of Variation	Degrees of Freedom	Sum of Squares	% Total among Groups	F Ratio
1. Month	11	80.3952	0.012	27.43
2. Location	1	1395.1033	0.202	5236.87
3. Sex	1	5430.3577	0.785	20384.23
4. Month x Location	11	1.7527	0.000	0.60†
5. Month x Sex	11	3.3336	0.001	1.13†
6. Location x Sex	1	1.4358	0.000	5.39
7. Month x Location x Sex	11	1.5398	0.000	0.53†
8. Total among Groups	47	6913.9181		552.19
Error	1761515	469183.6076		

† $p > 0.05$.

Table III, Part A, presents the results of a rank sum test for Niagara versus neighboring counties. It is apparent that Niagara County mean birth weight is significantly lighter than expected for males and somewhat lighter, though not significantly so, for females. Table III, Part B, shows that Niagara County males are significantly lighter than the control population, but the Niagara County females are not, Nonetheless, for both comparisons the females are near significance ($P \approx 0.10$) and, taken overall, Niagara County shows significantly low birth weights.

Given that Table III suggests a differential response in males versus females, one might suppose that consideration of sex ratio may be informative. However, Table IV shows that this is not the case. A trend in sex ratio among counties is not even weakly suggested by our data.

TABLE III. RANK SUMS FOR COUNTY-LEVEL MEAN BIRTH WEIGHT, TWO-YEAR PERIODS, 1968–1975, NEW YORK, WHITES BY SEX

A. Comparison of Niagara County with surrounding counties

County Name	Males Rank by Year 68–69	70–71	72–73	74–75	Rank Sum	P*	Females Rank by Year 68–69	70–71	72–73	74–75	Rank Sum	P
Erie	2	2	2	1	7	.056	3	1	1	2	7	.056
Genesee	5	5	5	5	20	1.000	4	4	4	4	16	.944
Niagara	1	1	1	2	5	.008	2	3	2	1	8	.112
Orleans	3	4	3	3	13	.696	1	2	5	3	11	.432
Wyoming	4	3	4	4	15	.888	5	5	3	5	18	.992

B. Comparison of Niagara County with demographically and socio-economically similar counties

County Name	Males Rank by Year 68–69	70–71	72–73	74–75	Rank Sum	P*	Females Rank by Year 68–69	70–71	72–73	74–75	Rank Sum	P
Broome	9	7	9	5	30	.9285	9	10	8	8	35	.9930
Erie	3	3	3	2	11	.0330	3	2	1	4	10	.0210
Monroe	4	6	5	1	16	.1760	4	1	5	1	11	.0330
Niagara	1	2	1	4	8	.0070	2	4	6	2	14	.0997
Oneida	2	4	4	3	13	.0715	1	5	2	5	13	.0715
Onondaga	6	5	6	9	26	.7760	6	8	9	9	32	.9670
Rensselaer	10	10	10	6	36	.9965	5	3	7	3	18	.2780
Saratoga	8	1	2	8	19	.3372	8	9	3	7	27	.8240
Schenectady	5	9	7	7	28	.8655	7	6	4	6	23	.5995
Tompkins	7	8	8	10	33	.9790	10	7	10	10	37	.9985

* Probability of a rank sum as small or smaller than that observed, given random assignment of ranks.

DISCUSSION

The first part of our analysis differs from those of earlier workers in that it confirms the consistency of the effects of three major determinants across four data sets (two locations and sexes) by statistical analysis, whereas previously [8,11] they had been inferred from simple tabular representation of the data. The first part also differs from previous studies in that it uses ANOVA methodology to examine population trends as opposed to

TABLE IV. RANK SUMS FOR COUNTY-LEVEL WHITE SEX RATIOS AT BIRTH, TWO-YEAR PERIODS, 1968- 1975, NEW YORK

A. Comparison of Niagara County with surrounding counties

County Name	Rank 68-69	70-71	72-73	74-75	Rank Sum	p*
Erie	2	3	3	3	11	.432
Genesee	1	5	1	1	8	.112
Niagara	3	2	4	2	11	.432
Orleans	4	4	2	1	11	.432
Wyoming	4	1	5	4	14	.805

B. Comparison of Niagara County with demographically and socio-economically similar counties

County Name	Rank 68-69	70-71	72-73	74-75	Rank Sum	p*
Broome	6	4	3	5	18	.2780
Erie	3	6	3	6	18	.2780
Monroe	4	8	5	5	22	.5335
Niagara	5	5	4	2	16	.1760
Oneida	6	7	2	7	22	.5335
Onondaga	1	7	4	1	13	.0715
Rensselaer	7	3	6	3	19	.3372
Saratoga	2	1	1	2	6	.0015
Schenectady	5	2	3	4	14	.0997
Tompkins	8	5	2	7	22	.5335

* Probability of a rank sum as small or smaller than that observed, given random assignment of ranks.

regression methods aimed at prediction of individual birth weights [40,41]. For the independent variables considered here, fitting regression models is inappropriate because of the collinearity which is necessarily present in the dependent variables (e.g. parity tends to increase with maternal age) and because the effect of the independent variables on birth weight is nonlinear. These difficulties render regression coefficients unstable in that they are sensitive to the addition or deletion of variables or even data points in the model[42-44]. Further, if we calculate R^2,

TABLE V. INDUSTRY GROUPS PRESENT IN NIAGARA AND COMPARISON COUNTIES IN NEW YORK, 1972

Ind. Code	Industry Group in Niagara County	No. of Establishments	No. of Employees (1000's)	Other Counties in Study with This Industry (No. Establishments/No. Employees (1000's))
262 265	Papermills, Ex. Bldg. Paper; Paperboard Containers & Boxes	3	Not available	Erie (24/1.7), Onondaga (10/.7)
27	Printing & Publishing	28	1	Broome (47/N.A.), Erie (194/6.8), Monroe (164/5.8), Oneida (40/N.A.), Onondaga (91/2.2), Rensselaer (26/.6), Schenectady (15/.6)
28	Chemicals & Allied Prod.	6	0.6	Erie (68/4.2), Monroe (25/1.7), Onondaga (23/N.A.)
281	Industrial Inorganic Chem.	10	1.2	
286	Industrial Organic Chem.	9	N.A.	
331	Blast Furnace, Basic Steel Prod.	5	1.6	Erie (5/13.5)
34	Fabricated Metal Prod.	30	1.1	Broome (20/N.A.), Erie (159/13.2), Monroe (125/5.8), Oneida (22/1.4), Onondaga (67/313), Renesslaer (9/.6), Genesee (15/.8)
36	Electric, Electronic Equip.	14	3.2	Monroe (51/11.1), Onondaga (34/N.A.), Wyoming (5/2.1)

SOURCE: 1972 Census of Manufacturers [31].

TABLE VI. MEAN BIRTH WEIGHT, NUMBER OF BIRTHS, AND SEX RATIO BY COUNTY, TWO-YEAR PERIODS 1968–1975, NEW YORK WHITES

County Name		Males 68/69	Males 70/71	Males 72/73	Males 74/75	Females 68/69	Females 70/71	Females 72/73	Females 74/75	Sex Ratio 68/69	Sex Ratio 70/71	Sex Ratio 72/73	Sex Ratio 74/75
A. Comparison of Niagara County with surrounding counties													
Erie	$\bar{x}$	3.3168	3.3415	3.3926	3.3926	3.2014	3.2135	3.2429	3.2727	1.07	1.05	1.06	1.09
	n	3370	3205	4217	5633	3162	3046	3984	5186				
Genesee	$\bar{x}$	3.4190	3.4452	3.4604	3.4756	3.2572	3.2417	3.3099	3.3299	0.94	1.20	0.92	1.03
	n	187	196	242	344	198	163	261	333				
Niagara	$\bar{x}$	3.2929	3.3398	3.3728	3.4057	3.1898	3.2325	3.2693	3.2684	1.10	1.04	1.07	1.04
	n	756	685	955	1310	687	659	896	1261				
Orleans	$\bar{x}$	3.3679	3.3655	3.3940	3.4516	3.1814	3.2208	3.3873	3.2895	1.18	1.08	0.80	1.03
	n	118	112	133	208	100	104	166	202				
Wyoming	$\bar{x}$	3.4004	3.3424	3.3953	3.4675	3.2960	3.2524	3.2881	3.3479	1.18	0.98	1.16	1.18
	n	113	109	173	248	96	111	149	210				
B. Comparison of Niagara County with demographically and socio-economically similar counties													
Broome	$\bar{x}$	3.3833	3.3824	3.4120	3.4190	3.2396	3.2686	3.2793	3.3080	1.11	1.03	1.06	1.08
	n	757	733	902	1269	682	715	853	1175				
Erie	$\bar{x}$	3.1368	3.3415	3.3926	3.3926	3.2014	3.2135	3.2429	3.2727	1.07	1.05	1.06	1.09
	n	3370	3205	4217	5633	3162	3046	3984	5186				

Monroe	$\bar{x}$	3.3266	3.3672	3.4013	3.3906	3.2024	3.2012	3.2631	3.2548	1.09	1.08	1.10	1.08
	n	2661	2485	3129	4052	2446	2302	2835	3750				
Niagara	$\bar{x}$	3.2929	3.3398	3.3728	3.4057	3.1898	3.2325	3.2693	3.2684	1.10	1.04	1.07	1.04
	n	756	685	955	1310	687	659	896	1261				
Oneida	$\bar{x}$	3.3164	3.3594	3.3969	3.4035	3.1702	3.2339	3.2528	3.2889	1.11	1.07	1.05	1.11
	n	950	912	1182	1557	853	851	1123	1405				
Onondaga	$\bar{x}$	3.3632	3.3658	3.4021	3.4521	3.2299	3.2509	3.2964	3.3228	0.97	1.07	1.07	1.01
	n	1539	1537	1939	2486	1588	1431	1814	2455				
Rensselaer	$\bar{x}$	3.3878	3.3977	3.4175	3.4252	3.2083	3.2260	3.2789	3.2723	1.13	1.00	1.09	1.05
	n	521	487	627	774	461	489	576	736				
Saratoga	$\bar{x}$	3.3728	3.3263	3.3917	3.4326	3.2470	3.2586	3.2587	3.3028	1.01	0.94	1.04	1.04
	n	417	468	733	973	414	498	705	932				
Schenectady	$\bar{x}$	3.3479	3.3895	3.4072	3.4279	3.2301	3.2436	3.2608	3.2907	1.10	0.99	1.06	1.06
	n	552	532	706	878	500	537	667	830				
Tompkins	$\bar{x}$	3.3660	3.3832	3.4115	3.5099	3.2797	3.2486	3.3214	3.3246	1.14	1.04	1.05	1.11
	n	317	279	355	481	279	268	337	433				

$\bar{x}$ = mean birth weight.

n = number of observations.

which is a measure of the predictive power of a linear model for an individual observation [45] and the focus of regression, we obtain values of 0.02 for the analysis presented in Table I and 0.015 for the analysis presented in Table II, indicating that our model has little predictive value for the weight of an individual.

If, on the other hand, discrete effects on sample means are considered, which is the view taken in ANOVA, the standard error of the sample mean, $\sigma_{\bar{X}}$, is given by

$$\sigma_{\bar{X}} = \sigma \sqrt{N}$$

where N is the sample size from which the mean is determined and is the population standard deviation. Thus, in large samples such as those used here, the population mean and small effects on it can be estimated with negligible error.

The demonstration of a clear interaction between mother's age and parity, as well as the individual effect of each, in the determination of mean birth weight has not been previously reported. More importantly, we have shown that there is little interaction of the effect of mother's age or parity with either sex or location. Taken together, all the interactions of mother's age and/or parity with sex and/or location (Table I, lines 6-9, 11-15) account for less than two percent of the among-group sum of squares (Table I, line 16). This lack of interaction demonstrates that effects of mother's age and parity are quite consistent across the four populations (two sexes, two locations). A separate evaluation of seasonality confirms the insignificance of the interaction terms (Table II, lines 4-7) and the fact that they account for less than one percent of the among-group sum of squares (Table II, line 8). Again, despite the independent variables' low predictive value for an individual birth weight, their consistent effect at the population level is unmistakable, making population means predictable. Thus, the first part of the analysis affirmatively answers the initial question in the development of a health status evaluation methodology. Consistent effects on birth weight are seen with two major influencing factors while controlling for only a few additional factors.

The first part of our analysis also reflects the known difference in mean birth weight between sexes and, as stated earlier, the second part of our analysis suggests a differential response of males versus females in both comparisons. These points, in addition to the fact that the male fetus has a higher level of intrauterine mortality [35], indicate that the sexes may react differently to the environment. Studies that consider only combined male-female birth weight [11,12,46] may therefore be discarding useful information.

Our analyses of Niagara County versus two groups of control counties also suggest that county mean birth weights do indeed reflect subtle environmental stresses. Though urban-rural differences [25] might be a partial underlying factor in the comparison of Niagara to surrounding counties, they are eliminated in the selection of control counties for the second analysis, and the two gave very similar results. Using a log-likelihood approach to combining the results of the four pairwise comparisons [47], we obtain a chi square of 38.57 on 8 degrees of freedom ($P < 0.001$), which demonstrates that, overall, Niagara County mean birth weight is significantly lighter than the controls. While the interpretation of this difference as the result of a particular stress is problematical, Niagara County is the only New York county of those considered that has a substantial organic/inorganic chemical industry (Table V) and in addition contains a substantial number of chemical dump sites [32].

The necessity for caution when interpreting these results is reinforced by the fact that the differences in mean birth weight are generally less than 100 grams (Table VI). Further, the differential response of the sexes suggested in Table III is not evident in the analysis of sex ratios (i.e. we have no evidence of differential intrauterine mortality). Nonetheless, we have shown that trends in birth weight with known influencing factors are consistent and can be discerned with a minimum of adjustment for other variables. Further, a presumptively stressed population was shown to have significantly lighter birth weights than either surrounding counties or a systematically chosen group of controls. These results suggest that further studies of birth weight and other birth record data, such as gestational age or Apgar score [48], may yield indicators of population stress from environmental pollution.

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DISCUSSION

L.D. HAMILTON *(Chairman)*: Is the exposure period necessarily limited to the nine months in utero? Is it not possible that pre-conception exposure could affect body weight? If so, this would be an excellent index to use in areas liable to be affected by uranium mining.

M.E. GINEVAN: Pre-conception exposure could influence birth weight, possibly through an existing body burden. For some contaminants – for example radium, a calcium analogue which we are currently examining in a different context – one might expect to have a historical body burden which actually contributes to the exposure of the foetus. Such exposure would be a useful index in the case you mention as well as in a number of other situations where populations have been recently introduced. There is a great deal of potential in this approach because of the complete independence of the data from year to year. The births occurring now are not necessarily related to those which occurred two years ago, a fact which is not generally true of mortality statistics.

M. EL DESOUKY: In your study you corrected for the age and parity of the mother. Do you not think that other variables relating to the mother, such as nutritional status and body weight, should also be taken into consideration?

M.E. GINEVAN: The purpose of our research is to examine the shift in the population mean in the case of reasonable populations of, say, several thousand. We believe that it is possible to obtain significant information without having to correct for a lot of the ancillary variables for which good information is often not available -- one certainly does not have information on nutritional status, for instance. Information on the weight of the mother can sometimes be obtained but birth certificates are by no means uniform in providing such data. Since a great deal of data will be missing, it will therefore be very difficult to correct for these factors.

K. SUNDARAM: Did you also analyse for premature births in this population? If you feel that birth weight is a determining factor one would have thought that it would also be useful to consider the incidence of premature births or reduced gestational period.

M.E. GINEVAN: We have not yet analysed that part of the data, but I would agree with you that the most desirable analysis would be, at the very least, the bivariant one of gestational age and birth weight. Fortunately, US birth certificates contain this information for a very large number of births. In addition, since 1978 we have also had the Apgar score, which is a general measure of the infant's condition at the time of birth based on several criteria. The birth record may therefore ultimately provide two if not three useful pieces of information. I would agree that it would be of great interest to consider the gestational age because that information is available for every infant. Premature births, however, would be a less useful index since they concern a much smaller proportion of the population.

S.R. BOZZO: Was there a correction in your analysis for changes in medical attitudes to the weight gain of mothers during pregnancy?

M.E. GINEVAN: That was one of the reasons why our analyses were stratified across time. We were aware of a secular trend in the data and for that reason it was important to choose an analysis which could be plotted against time. We ranked our data within time periods and summed the ranks across time periods with the result that most of the secular trend in the data has been eliminated.

P. METALLI: The distribution of weight at birth is usually skewed, with a rather consistent and appreciable tail to the left consisting of babies who are considered underweight. Was this the case in your observations? If so, did you apply any correction factor or cut-off limit to the distributions before carrying out your analyses?

M.E. GINEVAN: Our distribution was skewed, but we did not apply any correction factor because in the analysis of variants we were mainly interested in obtaining the partitioning of variants to determine whether large interaction terms were present, and for that purpose it is not particularly important whether the data are skewed or not. For the purposes of hypothesis testing, however, it is important because an inflated mean square will result. In our analyses of comparisons of counties, this problem was averted as the rank test made no assumptions whatsoever about the data distribution.

MORTALITY AND POLLUTION: ERROR ANALYSIS FOR A CROSS-SECTIONAL STUDY

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Abstract

MORTALITY AND POLLUTION: ERROR ANALYSIS FOR A CROSS-SECTIONAL STUDY.

An error analysis is given for a cross-sectional study of the effects of air pollution on death rates for 57 towns in the United Kingdom. Smoke and SO_2 concentrations and figures for past levels of domestic coal consumption are used as indices of air pollution while socio-economic level, latitude, water calcium content and rainfall are used as control variables in a multiple regression analysis. There are two broad categories of error in the results. First, statistical fluctuations in numbers of deaths, uncertainties in population age distribution, lack of information on smoking habits, and uncertainties in pollution levels lead (among other factors) to a random scatter in the estimate of effects attributed to air pollution. This limits the precision and sensitivity of the analysis. Second, an inadequate choice of control variables can lead to a systematic bias in the air pollution term. In particular, the results vary widely according to the degree of detail of the representation of the socio-economic variables which have a strong effect on death rates. This source of bias leads to uncertainties over and above those allowed for in the usual calculation of confidence limits. Together these two categories of error severely limit the usefulness of cross-sectional regression analysis as a tool for assessing the health effects of low levels of air pollution.

1. INTRODUCTION

One component of analyses of the risks or cost-benefits of energy supply systems is an assumed relationship between air pollution and human health and, in particular, death rates. This may provide the dominant contribution to the assessed risk (1). The dose-response relationship may be taken either from time series analysis of a sequence of daily death totals or from cross-sectional analysis of death rates and pollution levels in different geographical areas (see for example Lave and Seskin (2)). For time series calculations, the sensitivity of the conclusions to the assumptions made in the analysis has been demonstrated by Buechley (3) and Schimmel (4).

This paper aims to give an error analysis for the cross-sectional method. It assesses the effect on the

TABLE I. STANDARDIZED COEFFICIENTS FOR REGRESSION OF DEATH RATE AGAINST ENVIRONMENTAL FACTORS (45–64 AGE GROUP)

Category	Pollution index	Social factor score	Pollution	Latitude	Water calcium	Rainfall	% variance explained
All causes, male	SO_2	0.44[a]	0.09	0.26[b]	−0.23[c]	0.17	78
	Smoke	0.43[a]	0.10	0.24[c]	−0.26[b]	0.16	78
	1952 coal index	0.31[b]	0.26[c]	0.21[c]	−0.23[c]	0.17[c]	79
All causes, female	SO_2	0.30[b]	0.18	0.33[b]	−0.05	0.37[a]	73
	Smoke	0.30[c]	0.10	0.33[b]	−0.14	0.35[a]	72
	1952 coal index	0.15	0.34[c]	0.27[c]	−0.10	0.37[a]	75
Bronchitis, male	SO_2	0.52[b]	0.01	0.32[b]	−0.09	0.07	64
	Smoke	0.50[a]	0.07	0.30[c]	−0.05	0.07	64
	1952 coal index	0.37[c]	0.27	0.24[c]	−0.01	0.08	66
Bronchitis, female	SO_2	0.34[c]	0.28[c]	0.18	−0.16	0.18	67
	Smoke	0.24[c]	0.46[a]	0.04	−0.25[c]	0.16	71
	1952 coal index	0.13	0.47[b]	0.14	−0.18	0.16	68

[a] Significance level $p < 0.001$.

[b] Significance level $p < 0.01$.

[c] Significance level $p < 0.05$.

precision and robustness of the conclusions of statistical variability in the mortality data, errors in the estimates of pollution level, and uncertainties in the treatment of confounding variables, especially socio-economic factors. Data for death rates and for British Standard (BS) smoke and SO_2 level around 1971, in the 57 U.K. county boroughs considered by Gardner, Crawford and Morris (5), are used to give illustrative calculations of these components of statistical scatter and/or bias in the calculated air-pollution effects.

2. CROSS-SECTIONAL CALCULATIONS

The main analyses were done using multiple linear regression, and it is convenient to deal with certain 'baseline' regressions whose sensitivity to various changes can then be explored. For this purpose the dependent variables (1969-1973 death rates for men and women in the 45-64 and 65-74 age groups, from all causes and from bronchitis alone) were considered in relation to the five environmental variables selected by Gardner et al. (5) in their analysis of death rates around 1951 and 1961. These were air pollution, latitude, rainfall, water calcium and a composite social factor score calculated by Gardner et al. from a principal component analysis of nine primary socio-economic variables. The main index of air pollution used was the 1970/71 winter average SO_2 concentration (6), though some calculations were done with BS smoke measurements and with Daly's 1952 domestic coal consumption index (7), which gives a measure of pollution patterns prior to the U.K. Clean Air Act of 1956. Further details are given elsewhere (5,8).

Table I shows the regression coefficients in some of the 'baseline' regressions for 1971. For example, for male death rates from all causes in the 45-64 age group:

$$\begin{aligned}\text{Mortality} = {} & 4.98 + 0.09 \times \text{winter } SO_2 \text{ concentration} \\ & + 0.44 \times \text{social factor index} \\ & + 0.26 \times \text{latitude} - 0.23 \times \text{water calcium} \\ & + 0.17 \times \text{rainfall}\end{aligned}$$

$$\text{(multiple correlation coefficient } R = 0.88, \quad R^2 = 0.78)$$

All coefficients are given in standardized units, that is, units such that the standard deviations of the dependent and each of the independent variables are unity. These permit a direct comparison of the apparent RMS magnitude of the pollution signal sought with the level of statistical 'noise' in the

TABLE II. COMPONENTS OF DEATH RATE (MALE, ALL CAUSES, 45–64) ASSOCIATED WITH VARIABLES IN REGRESSIONS

% change in death rate associated with:	Pollution variable in regression		
	SO_2	Smoke	Coal index
Mean pollution level	3.6	2.5	10.2
Standard error	3.4	2.7	4.7
95% confidence interval	±6.9	±5.5	±9.5
Range in social factor score	27.6	27.0	19.5
Range in latitude	14.1	13.0	11.4
Range in water calcium	11.0	12.4	11.0
Range in rainfall	9.2	8.7	9.1

regression (RMS error term $\varepsilon = \sqrt{1-R^2} = 0.47$ in the same units) and with the variations associated with the control variables. Neither here nor in the corresponding regression for BS smoke (standardized pollution coefficient = 0.10) does the pollution variable attain statistical significance, though a significant association is found with the Daly coal-burning index (pollution coefficient = 0.26, significant at 95% confidence level, $p < 0.05$).

Table II expresses the same results in more practical terms. The figures given represent the percentage changes in male, all causes, 45-64 death rates which are associated on the calculated linear relationship with mean levels of SO_2 or BS smoke or of the coal-burning index, and with the range of values of each of the control variables over the sample set of county boroughs. Again, the estimated components of mortality associated with SO_2 or BS smoke concentrations are seen to be small both compared with their statistical confidence intervals and with the variations in mortality associated with the control variables.

3. SENSITIVITY ANALYSIS

Some of the sources of error in the regression analysis may be illustrated by the examples 3.1 to 3.5 given below.

3.1. Small number fluctuations

An obvious source of uncertainty is random fluctuation in the dependent mortality variable. For Table II, death rates were calculated from 5 years' data, and, for example, the male death rates from all causes in the 45-64 age group are based in an average county borough (total population 150 000 to 200 000) on about 1500 individual deaths. In a total of this size, 'small number' or Poissonian fluctuations are expected with a standard deviation $\sqrt{1500} \approx 40$, or about 2.5%. These account here for about one sixth of the total unexplained variance in the regression, and contribute a component of about ±1.4% to the standard error of the air-pollution term of Table II.

3.2. Smoking

Similar local variations in death rates may be expected from local differences in smoking habits, which are not allowed for in the analysis. There are unfortunately no published data which show how the proportion of smokers varies from one county borough to another; for the purposes of a sensitivity analysis, calculations must be based on variations (typically ±10% around a mean of 50%) in the proportion of smokers in different occupational groups (9) or the range (about 10%) between the aggregate figures for conurbations on the one hand and rural districts on the other (10). Variations of ±5% in the proportions of smokers in different boroughs would, according to the death-rate data of Hammond and Horn (11) or Cederlof et al. (12), lead to variations of about ±2% in all-causes death rates or ±5% in bronchitis death rates. These again contribute to the uncertainties of Table II even if it may be assumed that there is no direct confounding of the air-pollution term by the effects of smoking.

3.3. Age distribution

'Unexplained' variations may also arise from the residual effect of variables which nominally have already been allowed for but in fact are not adequately described. For example, the effect of population age structure on death rates is apparently already controlled for by the calculation of death rates for specific age groups. Here, however, a breakdown of the 45-64 age group into separate 45-54 and 55-64 age groups shows that there are variations in the population age structure between different boroughs which would be expected to contribute variations of ±2% in all-causes death rates. Correction for these variations reduces the uncertainty in the pollution term of Table II by a factor 0.9.

3.4. Pollution estimation errors

Random or quasi-random errors of measurement in the independent 'causal' variables included in the analysis lead directly to a distortion of the corresponding regression coefficients. For example, suppose that the SO_2 concentrations used in the calculations of Section 2 (obtained by averaging over all available measuring sites in each county borough) differed by ±10 $\mu g/m^3$ (RMS) from the 'true' population weighted averages. (For comparison, the sample mean over the set of 57 boroughs is 139 $\mu g/m^3$.) A full analysis (to be published elsewhere) shows that errors in the pollution measure lead to two kinds of error in the estimated pollution-related component of mortality; a systematic underestimate, by a factor 0.9 for this regression, or by 0.3% for the figure quoted in Table II, and an additional component of random uncertainty which here has a standard deviation 1.0%. Both kinds of error become more serious as the regression variables exhibit a greater degree of multicollinearity.

3.5. Choice of regression variables

A further uncertainty in interpretation arises from the unavoidable arbitrariness in the choice of control variables. The problem is particularly serious when these control variables have a strong effect on mortality. For example, all-causes death rates vary by a factor of two over the U.K. social class scale (9), while there are regional differences of up to 20% in death rates which may result from differences in climate or in genetic factors, or possibly from selective migration to the more prosperous South-Eastern area. These effects are reflected in the importance of the social factor and latitude variables in Table II, producing differences of up to 27% and 14% respectively over the set of county boroughs considered.

For these regressions the sensitivity of the air pollution coefficient to the specification of the control variables has been illustrated by repeating the calculation (a) with the latitude variable excluded and (b) with composite social factor score replaced by a set of separate socio-economic variables (population density, migration balance,

TABLE III. POLLUTION-RELATED COMPONENTS OF DEATH RATE (MALE, ALL CAUSES, 45–64) IN REGRESSIONS WITH DIFFERENT CONTROL VARIABLES

% change in death rate in regression where control variables are:	Pollution variable		
	SO_2	Smoke	Coal index
Baseline (social factor index, latitude, water calcium, rainfall)	3.6	2.5	10.2
Baseline without latitude	7.2	6.0	14.8
Baseline: social factor index replaced by full set of primary socio-economic variables	1.1	−1.0	1.9

per cent population aged under 15, and indices of social class, overcrowding, income levels, educational levels and car ownership (7,13,14,15,16). The results are shown in Table III. In each case the air pollution term is substantially increased by the exclusion of the latitude variable, and substantially reduced (becoming negative in the regression for BS smoke) when the full set of socio-economic variables is included. In the case of SO_2 the air-pollution term varies over a range of 6% of total mortality according to the choice of control variables.

4. DISCUSSION

The aim here has been to investigate the precision and reliability of the cross-sectional regression analysis, considered as a scientific tool. From this point of view the most striking feature of Table II is the width of the confidence interval for the air-pollution term. To attain statistical significance ($p < 0.05$), air pollution would have to account for about 7% of total mortality in the 45-64 age group. This is very much larger than estimates which might be derived on a longitudinal basis from the change in death rates over the period 1961-1971 when pollution levels declined rapidly. Typical urban annual average concentrations of smoke and SO_2 for 1970-71 were about 60 $\mu g/m^3$ and 100 $\mu g/m^3$ respectively, compared with the annual averages of about 140 $\mu g/m^3$ for both pollutants measured in 1960/61, or with the figures of 150–200 $\mu g/m^3$ obtained for 1958/59 which might be considered more typical of pre-Clean Air Act levels (6). Between 1959-1963 and 1969-1973 death rates for men aged 45-64 fell by less than 1%, while there was a marginal increase in the corresponding rate for women (8). On an individual scale, an air-pollution-induced increase of 7% in age-specific death rates would correspond to an average loss of life expectancy of 8 or 9 months, as compared with the loss of 5 years (17) from the presumably very much more severe effects of smoking twenty cigarettes a day.

The examples of 3.1 to 3.4 show how the errors which limit the resolution of the analysis come from a wide range of factors. Some of these, like small number fluctuations in the mortality data or uncertainties in the pollution estimates, are for all practical purposes unavoidable. Even with perfect knowledge of population age distribution, smoking habits, socio-economic factors, etc., regressions with these mortality and pollution data could not detect (with $p < 0.05$) a pollution-related component of less than 3.5% of all-causes death rates.

The calculations of Section 3.5 demonstrate further the limits to the reliability of estimation of a small air-pollution term which arise from the uncertainty in modelling of the much more important socio-economic variables. The air-pollution contribution varies by 6% or more of the age-specific death rate depending on the control variables selected. This represents a range of possible systematic bias over and above the statistical confidence limits discussed above.

5. CONCLUSIONS

For the measurement of the effects of present-day concentrations of air pollution, regression analysis is a crude and unreliable instrument. Several factors - statistical fluctuations in numbers of deaths, errors of estimation in pollution levels, lack of information concerning age distribution or smoking habits in the populations studied - give uncertainties of a scale as large as that of the likely effects of air pollution. Interpretation is further hindered by confounding with the much more important gradients in mortality determined by socio-economic differences. Dose-response calculations for use in risk analysis or cost-benefit calculations should not be considered complete without a sensitivity analysis of the kind outlined here.

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DISCUSSION

I.M. TORRENS: Overall mortality rates – as distinct from mortality from a specific disease – might be expected to show little correlation with most items we try to relate to them. The conclusion your paper reaches is therefore not very surprising. But is it fair to extend that conclusion to more limited attempts at regression involving, say, bronchitis and air pollution?

J.H. PICKLES: One might indeed have hoped that regressions relating bronchitis to air pollution would have been a more precise tool than regressions involving overall mortality. In fact, in this study at least, the advantages of the more specific disease indicator are outweighed by the greater variability of bronchitis mortality with socio-economic status (by a factor of 5 in the United Kingdom) and smoking habits (by a factor of 3), and by the greater statistical scatter in the smaller numbers of deaths.

I.M. TORRENS: In your analysis, smoking is buried in the 'error'. Since it is quite an important cause of mortality, if it were possible to include it as a specific factor, might it not swamp the others?

J.H. PICKLES: Smoking is certainly an overwhelmingly important factor when one speaks of individual deaths. Its importance in these regressions which concern community average death rates depends on the extent to which smoking habits vary between towns. If there were exactly the same proportion of smokers in each town, the smoking-induced deaths would not confuse the regressions. In fact, differences in smoking habits between towns do exist; I have crudely and perhaps conservatively estimated that these introduce an r.m.s. scatter of 2% in death rates from all causes, or 5% on bronchitis death rates.

Another difficulty arises from the rapid increase in smoking among women over the last few decades. The 1910 birth cohort (60 years old in 1970) had on average smoked twice as many cigarettes as the 1900 birth cohort (60 years old in 1960). This trend of increased smoking started earlier in the larger towns and conurbations and it is likely that a bias has been introduced into the regressions as a result.

S.R. BOZZO: Is it possible to use a different factor instead of the SO_2 concentration in your calculation?

J.H. PICKLES: If the SO_2 concentration is left out of the regression, the new regression equation is

$$\begin{aligned} \text{death rate} = {} & 0.46 \times \text{social factor score} + 0.30 \times \text{latitude} \\ & -0.27 \times \text{water calcium} \\ & +0.15 \times \text{rainfall} \end{aligned}$$

in the same standardized units as before. The increases in the social factor and latitude coefficients reflect the correlation ($r = 0.52$ and $r = 0.62$ respectively) between these variables and SO_2. The proportion of variance accounted for by the regression falls by only 1%.

SESSION II

ROLE OF INTERACTING DEMOGRAPHIC COMPONENTS IN HEALTH EFFECTS AND RISKS

Chairman
A.K. BISWAS
United Kingdom

Invited Paper

THE RISK FROM ENVIRONMENTAL CHEMICALS

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Abstract

THE RISK FROM ENVIRONMENTAL CHEMICALS.

The elements of risk assessment, namely risk identification, risk estimation, risk evaluation and risk management, are described with respect to the control of environmental chemicals. The methodology of risk estimation is outlined and examples given of its application to regulatory decision-making for a number of chemicals in Canada. Finally, the extent and limitations of the process of risk evaluation are considered together with the need to recognize the importance of the public's perception of the level of risk.

The scope and nature of the chemical world created by our rapidly developing technological society over the last 30 years have had a dramatic impact on human health and the environment. The inventory of existing chemical substance in commerce numbers about 70 000 and the estimate of new chemicals entering the market each year in quantities greater than one tonne range from 200 to 1000. The hazards of these chemicals range from the acute to the long term: not only the highly publicized concerns with industrial accidents, home insulation, transport of dangerous chemicals, damage to wildlife, possible carcinogenic or mutagenic effects, and toxic waste disposal, but also the hazards of chronic neurotoxic or behavioural changes, or of subtle environmental damage.

The mechanisms available to society to control these hazards can be economic, educational, socio-cultural or regulatory. The economic approach could be to fine the polluter or to subsidize safety activities: educational programs can discourage hazardous activities or increase public awareness of the health consequence of particular chemicals.

Regulatory options are varied: the sale of chemicals may be banned; maximum allowable concentrations in food can be established; emission limits can be set; and, less rigorously, general guidelines can be proposed. In any event, before controls are imposed need and feasibility must be judged, preferably through the ordered sequence of risk identification, risk estimation, risk evaluation and, finally, risk management [1]. Initially, the hazard is recognized, then a scientific determination of risk is performed in as quantitative fashion as possible before the ultimately political evaluation is made on the acceptability of the risk and its consequences. This ordered approach helps separate the scientific estimation from the social evaluation and hence avoids some of the confusion that bedevils the risk assessment process.

Basis of Risk Estimation

There are three major sources of evidence that determine whether or not a chemical represents a danger to human health: epidemiology - in the occupational or general environment, animal experimentation, and *in vitro* testing. To some extent supportive information can be provided by correlations between chemical structure, in terms of functional groups, and biological activity; however, as closely related chemicals can possess different toxicity, this technique is of only limited value. Similarly, clinical studies, although appropriate for drugs, are not available for industrial chemicals.

Epidemiology can, in principle, provide the most definitive evidence of a chemical's risk to health. However, for environmental chemicals its main deficiency is its ineffectiveness in proving that long-term health effects result from a specific cause. Using either analysis by prospective cohort or case control studies, the long latency period for many effects obscures causal relationships. In reality, the requirement for a long period of use of the agent at an exposure level that produces discernible differences in population militates against the use of epidemiology for making regulatory decisions on environmental health issues. Many environmental chemicals - such as PCBs, heavy metals, organochlorine pesticides - are ubiquitously distributed so that it is almost impossible to find an unexposed population group. The increased pace of industrialization in many countries has led to the environmental dissemination of many exotic chemicals: if their biological action requires a long latent period it will be many years before epidemiology can chart their effect. The enormous number and range of chemical entities present in food make the search for dietary hazards particularly difficult. In Canada, for example, there are nearly 400 food additives and several thousand flavour substances. It would be well-nigh impossible to determine the detailed retrospective information required for an epidemiological assessment of their individual contribution to the health of the general public. Finally, the possibility of additive or synergistic health effects between the multitudinous chemicals present in the environment cannot be discounted.

In summary, epidemiology is more to support public health decisions on environmental chemicals rather than to initiate them.

Experiments with animals raise two fundamental questions: how valid is the comparison of man and animal? how legitimate is the extrapolation of high to low doses?

Although the basic biological processes of molecular, cellular and organ function are similar from one mammalian species to another, there are marked differences between the standard rodent model and humans. For example, the rat strain is homogeneous; it is maintained in a carefully controlled uniform environment of food, light, sound, etc., so that the insult is restricted to a single causative factor [2]: in addition, rats have no gallbladder and are independent of a dietary source of ascorbic acid. In some senses the more sophisticated the test the more limited its application. Nevertheless, nearly all the 26 chemicals (or industrial processes) that have been positively associated with human cancer through the program of the International Agency for Research on Cancer [3] are known to be carcinogens to animals. The converse argument - i.e. chemicals carcinogenic to laboratory animals are carcinogenic to man - is now backed by considerable experimental evidence, some of which is indirect, but must always be qualified by consideration of the nature of the test and species, specific incidence site, route of administration, and the whole spectrum of metabolism and excretion. Conversion factors are used in estimating risk levels for humans from data obtained in other species, e.g. body surface, body weight, nutritional conditions, tissue distribution, and retention. Extrapolations using body surface area rather than body weight provide a more conservative approach and one that is more appropriate for the comparison of metabolic processes.

Two distinct approaches have traditionally been followed to estimate from animal experiments the tolerable level of exposure for humans. One approach is to derive "acceptable daily intakes" based on levels of chemical that fail to produce an observed effect in the animal. The other approach, which is now held to be the most appropriate for carcinogenesis, is to extrapolate using mathematical models to low levels where no response can be observed. For food additives and pesticides, for example, an acceptable daily intake has often been established through the application of a 100-fold safety factor. This uncertainty factor admits the possibility that man may be up to 10-fold more sensitive than the animal species tested and allows for a 10-fold variation in sensitivity within the human population [4]. The magnitude of the safety factor may be modified depending on the chemical and kinetic properties of the test compound, the effects induced as well as the quality of the available toxicological data: safety factors as high as 1000 have been proposed [5]. The use of the "no observed effect level" (NOEL) concept tacitly assumes that a threshold dose exists and is, of course, dependent on sample size: response rates of 0/10 and 0/1000 will have different interpretations. With 50 animals on test there is a better than even chance of observing no effects with a compound for which the population risk is as high as 1 per cent [6]. The slope of the dose response curve is also of relevance: a moderate safety factor may not provide adequate protection if the response curve is shallow [7].

The mathematical extrapolation of dose-response curves outside the experimental range is generally employed for the interpretation of genetic effects, particularly carcinogenicity. At least six models are available: probit, logit, Weibull, one-hit, multi-hit and multi-stage [8]. Extrapolation to low doses is fraught with imponderable factors and the inappropriateness of equating physical radiation damage with the cell reactions of chemical carcinogenesis, or of assuming that biological effects are proportional to dose regardless of the size of dose or the rate of exposure, is being vigorously debated today. In these types of experiments, even if there is no evidence of carcinogenicity, it is not possible to prove conclusively that such a risk does not exist [9].

Munro and Krewski [10] have critically reviewed the limitations of statistical and stochastic models and noted that the shape of the dose-response curves has a considerable effect on the estimates of risk made by the different models. Because of these uncertainties, U.S. regulatory authorities have advocated the use of conservative one-hit extrapolation procedures [5]. Although this procedure may be appropriate for potent electrophilic carcinogens it may not apply for less potent chemicals that can induce tumors through perturbation of normal physiological processes. In addition, the target site of the carcinogen can differ in different species: aflatoxin B1 is carcinogenic in rats but not in adult mice. Compounding of these factors can, on occasion, lead to estimates that are terrifying in their vagueness. For example, the National Academy of Sciences in their report on saccharin estimated that the expected number of cases of bladder cancer in the United States due to exposure to 120 mg saccharin per day may range from as low as 0.22 over the next 70 years to as high as 1 144 000 [11]. These estimates of risk span a range of eight orders of magnitude!

The time, expense and physical difficulties of animal tests have stimulated the search for short-term test methods to detect carcinogens. The most promising approach to date relies on the correlation between mutagenesis and carcinogenesis. A number of mutagenicity assays have been developed, of which the Salmonella microsome assay developed by Bruce Ames and associates [12] is the best known, most widely used, and most thoroughly validated. None of these tests gives incontrovertible proof of carcinogenicity but the Salmonella assay detected about 90% of the carcinogens examined, as also did the in-vitro mammalian cell transformation assay: in combination these tests detected virtually all carcinogens tested (99.2%), although the incidence of false positives was relatively high (8.8%) [13, 14].

As the various factors influencing the relationship between mutagenic potency and carcinogenic activity of a chemical are better understood, it is likely that the assay procedures will be improved to increase the correlation. However, it is doubtful whether the short-term mutagenicity assay will ever replace the need for human epidemiology and animal cancer tests.

This, then, is the armamentarium with which the hazard of a chemical can be determined. In addition to national evaluations, international committees of WHO and FAO have provided detailed analyses for a number of chemicals, especially pesticides, and food additives. There have not, however, been the concerted and detailed international assessments that have been directed to the biological effects of ionizing radiation by that renowned trinity of UNSCEAR, ICRP and BEIR. Partly, this is because the hazards of radiation have been tragically visited on human populations but also, I would suspect, because the logic of a rigorous analysis has proved particularly attractive to the physicists and mathematicians associated with nuclear energy. Be that as it may, chemists have belatedly learned their lesson, and risk estimations for chemicals are now the order of the day. A new International Programme of Chemical

Safety was launched by WHO in 1980 and is now co-sponsored by ILO and UNEP. This programme should provide the international validity and cachet for the evaluation of chemicals that is so urgently needed.

How to choose which chemical to be tested from the multitude available and potentially toxic is a problem that has exercised many national agencies. In Canada, a List of Priority Chemicals under the Environmental Contaminants Act is gazetted annually by the Departments of Environment and National Health and Welfare. Three sets of criteria are applied to the selection of chemicals for the list: toxic effects to human health or the environment, persistence, quantity in use. Within these criteria a committee by consensus develops a list. In the U.S.A., the Toxic Substances Control Act requires that an Interagency Testing Committee advise on up to 50 priority chemicals to be tested: as of November 1980, 42 chemicals were on the list. Selection of the chemicals is based on eight priority factors: (i) quantity manufactured; (ii) quantity that will enter environment; (iii) occupational exposure; (iv) non-occupational human exposure; (v) similarity in structure to chemicals known to present unreasonable risk to health or the environment; (vi) existence of data concerning health and environmental effects; (vii) extent to which testing will develop useful data on risk of injury to health or the environment and (viii) foreseeable availability of testing facilities and personnel. Consideration may also be given to chemicals known or suspected to cause cancer, gene mutation or birth defects.

In an attempt to harmonize test procedures for chemicals and hence reduce the adverse effects on international trade, the OECD Chemicals Group is analyzing a number of key processes used for evaluation. Preliminary agreement has been reached on the minimum set of pre-marketing data required for the initial assessment of a chemical. The International Joint Commission Committee responsible for the "Assessment of Human Health Effects of Great Lakes Water Quality" has been developing an approach similar to that of the OECD for ranking the approximately 400 chemical substances identified (as of 1978), as contaminants of the Great Lakes by the Great Lakes Water Quality Board. The development of a suitable hazard rating scheme is envisaged as comprising the two components; (a) ratings based on toxicity data, and (b) ratings based on the potential degree of exposure of population groups to the contaminant.

Regulatory Control of Chemicals in Canada

In Canada, the control of environmental chemicals by regulation is now using to a varying, but increasing, extent the techniques of risk estimation as will be evident from the following examples.

A federal-provincial working group has completed in 1978 a comprehensive review and re-assessment of the earlier Canadian Drinking Water Standards and Objectives. Trihalomethanes are known to occur in drinking water as a result of treatment by chlorine. The most commonly found trihalomethane in drinking water is chloroform, and a survey 1976-77 [15] of some 70 Canadian municipalities showed that its concentration can reach 121 μg/ltr. The question is raised: Is this an acceptable level? Epidemiological studies, in the lower Mississippi and Ohio river valleys, have not shown unequivocal evidence of a direct cause-effect relationship between chloroform contamination in drinking water and a variety of recorded cancers, although an association with bladder cancer has been suspected [16]. However, the U.S. National Cancer Institute rodent study found a dose-related incidence of malignant tumours in the kidneys of male rats and the liver of mice of both sexes [17]. The daily dose to produce these effects was high - in the range 100-500 mg/kg body weight. In addition, studies with cough suppressants and mouthwashes have shown that hepatotoxicity occurs in humans at oral doses of chloroform between 1 and 25 mg/kg/day (70 kg person): occupational exposure to chloroform in the pharmaceutical industry is known to produce liver damage to the workers [18].

Tardiff [17] has carried out a detailed analysis through four different models - margin of safety, and the statistical probit/log (Mantel/Bryan), linear or one-hit, and two-step - to determine the maximum risk incurred by drinking tap water containing chloroform. A tenfold margin of safety applied to liver damage in man gives a maximum daily dose of 0.03 mg/kg. Extrapolation from the rodent studies gave the maximum risks at a maximum daily dose of 0.01 mg/kg range up to 0.4 cancers/million/year. For a 70-kg man consuming 2 litres of water daily, this intake represents a chloroform concentration of 0.35 mg/litre: this is the recommended maximum acceptable concentration of total trihalomethanes in drinking water. The objective level, i.e. the ultimate quality goal, is less than or equal to 0.0005 mg/litre.

In this example, extrapolation from animal studies to man by means of statistical system is defensible because we know the human target organ for chloroform and there is some evidence of similar metabolic pathways. On this basis, we can say that the risk of cancer to a member of the population from drinking water attributable to chloroform lies between no risk to kidneys and liver and a maximum of 1 in 2.5 million/year.

Asbestos, which is now clearly indicted as a human carcinogen, is extensively used in dry-wall patching products. Analytical studies of the fibre concentrations of 10 dry-wall taping compounds during sanding, dry-mixing and floor-sweeping operations have shown average peaks as high as 47 fibre/cm^3[19]. The U.S. Consumer Product Safety Commission has developed a risk assessment model based on a number of assumptions including: linear dose-response relationship between asbestos and lung cancer time to tumour being dependent on dose, effect of dose is cumulative. These lead to a log dose-log response curve: the experimental data on cancer deaths attributable to dose being derived from the extensive epidemiological studies of Selikoff. Calculations of high intermittent exposure for four uses of asbestos patching products per year gives an exposure level equivalent of 0.4 fibres/cm^3/day for one year; it is estimated that this will cause an additional 10 lifetime respiratory cancer deaths/million. If effect of dose is cumulative and is assumed to have the same effect as if that dose had been accumulated in the year of exposure, then continued use of asbestos for five years will raise that estimate to 990 deaths/million.

It will be crystal clear why the Department of National Health and Welfare recommended that asbestos be banned in dry-wall patching compounds under the Hazardous Products Act. Such action has now been taken.

Clinical studies with arsenic medicinals and epidemiological investigations of exposure to arsenic in the occupational and general environment indicate that inorganic arsenic is a human carcinogen. The U.S. Environmental Protection Agency has developed a preliminary estimate of human cancer risk due to exposure to airborne arsenic, based on a linear extrapolation model and the results of epidemiological studies of occupational exposure [20, 21, 22]. The increase in the lung cancer rate per increase of 1 $\mu g/m^3$ of arsenic in the atmosphere was estimated to be 5.3%. Based on concentrations of arsenic measured in ambient air in the U.S., it was calculated that for the 2.16 million people residing in the vicinity of arsenic emissions exposure to airborne arsenic resulted in a lifetime risk of respiratory cancer of greater than 1 in 100 000. A linear dose-response model was also adopted to estimate the risk associated with the ingestion of arsenic in drinking water. The model is based on data from Taiwan which relates the increased incidence of skin cancer to levels of arsenic in drinking water [23].

On the basis of the well-documented evidence that arsenic is a human carcinogen, a recommendation was made by the Department of National Health and Welfare to the Department of the Environment to minimize the exposure of Canadians to arsenic through the regulation of controllable emissions. The development of regulations which will limit arsenic emissions from gold-roasting operations is being considered via the Clean Air Act.

A risk estimation more closely related to the health effects of energy production is that for the oxides of nitrogen. The major sources of localized high-pollutant levels of NO_x (nitric oxide and nitrogen dioxide) are the gasoline-burning internal-combustion engine and stationary industrial-combustion facilities which include power generating plants. Ambient concentrations in urban environments depend on the nature and number of sources and vary considerably with time, climatic conditions and distance from the source. In Canadian cities, mean annual concentrations of NO_x have been reported to range from about 18 $\mu g/m^3$ to 80 $\mu g/m^3$; peak levels (1 hour maximum concentrations) may be much higher [24].

Nitric oxide and nitrogen dioxide are both extremely reactive and attack the surface of the respiratory tract [25]. Data are lacking to describe with accuracy injurious effects attributable to low levels of NO but the adverse effects of NO_2 on biological systems are well-documented. Effects have been classified as direct short-term, chronic long-term and indirect [26], and this fact has been reflected to some degree in the development of air quality objectives, or standards. Nitrogen dioxide exerts its primary toxic effect on the lungs and is associated with increased susceptibility to respiratory infection, changes in pulmonary function, morphological changes, edema and emphysema.

A WHO review of the environmental health criteria for nitrogen oxides concludes that the epidemiological data are not adequate for long-term exposure guidelines [27]. Short-term exposures can give adverse effects - based on animal and human studies - at concentrations of nitrogen dioxide as low as 940 $\mu g/m^3$. The risk estimation was derived by the WHO group in terms of an arbitrary safety factor of 3-5. A nitrogen dioxide concentration of 190-320 $\mu g/m^3$ for a maximum of one hour was proposed as an exposure limit consistent with the protection of public health.

Ambient Air Quality Objectives for nitrogen dioxide under the Clean Air Act have been derived from epidemiological studies [24] and the "acceptable" level is at present up to 400 $\mu g/m^3$ for a one-hour exposure. These objectives are currently under revision in the light of more recent data.

There is a growing belief that the quality of indoor air may play a significant role in influencing health and that too much attention has been placed on the quality of the outdoor environment. We spend up to 70% of our time in non-occupational environments and the impetus for energy conservation has led to a progressive sealing-up of our homes. We know of the potential problems with radon daughters in some Canadian homes and our recent experience with urea-formaldehyde foam insulation has caused us to be concerned with the contamination of indoor air. The Department of National Health and Welfare is working with representatives from provincial governments to develop guidelines for indoor air quality.

Increasing problems with urea-formaldehyde foam insulation led to a ban on the sale of the product in December 1980. This decision was based on the report of an Expert Advisory Committee which noted the inherently unstable nature of the polymer leading to the release of formaldehyde. Formaldehyde is an irritant to the eyes, skin, and breathing passages and can produce allergic responses. Under controlled conditions formaldehyde has been shown to induce nasal passage carcinoma in rats and mice. Mathematical models predict that exposure between 4 ppb[1] and 1 ppm would produce one cancer in a million rats [28]. The 1 ppm estimate is based on mathematical models that appear to fit the perceived mechanism of action of formaldehyde.

In 1970, the Department of National Health and Welfare established a guideline for mercury in fish at 0.5 ppm and subsequently the FAO/WHO Expert Committee [29] recommended a tolerable weekly intake of 200 μg methylmercury per 60 kg adult based on animal and human studies. Swordfish was effectively removed from the Canadian market by this decision. In June 1979, the Departments of Fisheries and Oceans and Health and Welfare lifted the guideline for swordfish on the basis that the consumption of this delicacy is low and on the recommendation that it is limited to one meal a week. Essentially, the risk estimation for swordfish was made on the basis of level of consumption.

[1] Parts per billion, where 1 billion = 10^9.

The renowned Canadian decision to remove the artificial sweetener saccharin from foods, drugs and cosmetics was based on animal studies in which rats were fed a diet containing 5% sodium saccharin for two generations [30]. Male rats of both the parent and second generation showed a significant incidence of bladder tumours: the dose exceeded human exposure by at least 800 times - based on the bottle of 'diet' drink a day. Although other corroborating data were available, this was the key experiment which led to the regulatory decision at a time when some 200 000 lbs of saccharin a year were being used in Canadian foods. Some epidemiological support for this decision came from a case-control study in Canada [31] that was considered as evidence of increased risk of bladder cancer in males consuming artificial sweeteners, particularly saccharin. The difficulties in interpreting these results are considerable. Although saccharin was discovered 100 years ago, it is within the last few years that its use has dramatically increased. In addition, a long latency period for the induction of cancer - as suggested by the animal experiments - militates against the use of epidemiology for the public health decisions.

Limitations of Risk Evaluation

The above examples illustrate the manner in which a federal agency establishes a risk estimation of an environmental hazard as a basis for regulatory control. The next state of risk assessment - risk evaluation - represents societal judgement and is ultimately political, beyond the realm of scientific analysis. Nevertheless before that judgement is made there are a number of serviceable and worthwhile techniques that can be used to provide a holistic approach whereby the total context of the problem is considered. A particularly valuable technique for risk evaluation is to compare the risk either with other risks or the presumed benefit. A schematic approach by Whyte and Burton [1], enables risk equations be derived in the form: elevated risk, comparative risk, balanced risk, and risk-benefit. Elevated risk means comparison with the natural background - the level of 'noise' in the system. The contaminant may have always been with us - like mercury in fish, or fluoride in drinking water - and its beneficial or adverse effects tolerated. Comparing risks between different occupational groups, modes of transport, recreational activities, natural disasters and so forth in such terms as deaths/population unit/year or individual risk of cancer per year has been widely applied by Pochin [32] and Wilson [33]. When applied to disparate activities, devastating comparisons can be made. Pochin [34] gave a one-in-a-million risk of death to 400 miles travelled by air, 3/4 of a cigarette smoked, or 20 minutes of being a 60-year-old man.

In the same fashion, the average risk from the radiation exposure involved in nuclear power production can be estimated. On the basis of a world population deriving 1 kilowatt per person from nuclear sources around the world - the average individual exposure of 4 millirem would lead to genetic as well as malignant risks about equivalent to smoking one cigarette every two years.

Calculations such as these beg, of course, more questions than they answer. There is no doubt that we all accept a much higher level of risk - at least a thousand-fold according to Starr [35] - for activities that we deem voluntary as opposed to those regarded as involuntary, or imposed from outside. In reality, the distinction is not so nice as it seems. The power of advertising, peer pressure and other social forces often radically diminish the area of operation for free will. If we all genuinely accepted the odds there would hardly be the flourishing casinos or lotteries in evidence.

Balancing risks from alternatives lies at the heart of the social evaluation process. It has been quantified for energy production. Qualitatively the knowledge that replacement pesticides were available for DDT, saccharin for cyclamates, replacement fluids for PCBs, other insulating materials for urea-formaldehyde, influenced the public health decisions to ban these chemicals.

Even when risks are well-known and calculable there remains the problem of weighing them against the benefits giving rise to the risk. Pesticide risks to ecological systems can be weighed against increased food production. Chlorination of drinking water indisputably protects against some most unpleasant water-borne diseases. Some of these benefits may be commonly accepted as outweighing the risks but we are now facing the basic problem of societal judgement. A seemingly objective technique like cost-benefit analysis is of little value for environmental risk assessment for it requires the variables to be measured in commensurate, usually monetary, terms. Different people place different values on things such as human life, aesthetics and national security. Many factors cannot be satisfactorily expressed in dollar terms because they involve important values on which there is no agreement. To some extent the same criticism applies to the allied technique of risk-benefit analysis where implicit monetary values are attached to risks. Thus, if the exposure standard to acryonitrile is reduced to 0.2 ppm at an annual cost of $126.2 million in the expectation of preventing one additional cancer death a year in textile workers, the cost of that life is implied [36]. Nevertheless, these techniques and the associated ones of cost-effectiveness and decision analysis are increasingly required for the assessment of chemicals. In Canada, a detailed socio-economic impact statement is required for any decision related to health, fraud or safety that is expected to have a major economic impact on industry [37]: currently set at over $12 million.

There is no question but that in the last decade we have made great progress in building a rational scheme on which to assess the risk from environmental chemicals. But rationality is not enough. If it was, the public perception of hazards would equate with that of the professionals and the news media would have to find new sources of drama. The public is particularly exercised about risks over which they have no control and those which affect identifiable victims, particularly children. Inexorably, it is emotions at the deepest level that have been the lever of innumerable legislative and administrative public health decisions.

Rational argument and logical thought are supreme in our ability to provide a structured, ordered system of risk assessment for chemicals. But they must not be taken too far. It was Jung who said:

> "Rational argument can be conducted with some prospect of success only so long as the emotionality of a given situation does not exceed a certain critical degree. After that reason is replaced by slogans."

Some questions of fact, such as the extrapolation of animal response to minute doses, can be stated in the language of science but not answered by science - termed 'trans-science' by Weinberg [38]. Sometimes what passes for irrationality may be a well-founded suspicion of the operation of society. It is imperative that we recognize the limitations as well as the powers of the scientific method if we are to ensure that the political process which serves us all arrives at the just decisions required to control environmental hazards.

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DISCUSSION

B. SØRENSEN: How did you determine that the environmental conditions for rat experiments were 'minimum stress'?

E. SOMERS: The fact that the rats are kept at controlled temperature and humidity, with controlled light source, noise levels and diets provides a less stressful environment than that to which the human population is exposed.

B. SØRENSEN: As a general comment, I think it would be a good idea to replace the term 'acceptable risk' by 'accepted risk'. This would make it less embarrassing when levels have to be altered.

Invited Paper

RURAL AND URBAN ENERGY SCENARIO OF THE DEVELOPING COUNTRIES AND RELATED HEALTH ASSESSMENT

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Abstract

RURAL AND URBAN ENERGY SCENARIO OF THE DEVELOPING COUNTRIES AND RELATED HEALTH ASSESSMENT.

The pattern of energy use in India is considered in order to assess the impact on health of rural and urban energy sources in the developing countries. The health impact of the 'non-commercial' sources of energy used in India is discussed, with particular reference to the use of firewood and farm wastes for domestic cooking. The commercial energy sources considered include coal, oil and electricity. The generation of electricity from coal, hydro sources and nuclear fuels is discussed with regard to their health impact. The production and use of biogas instead of dried animal dung for domestic cooking in the rural areas of India is proposed in order to reduce the health detriment. On the basis of the past trend in the use of commercial and non-commercial energy in India, projections are made for the future, taking into consideration health detriment and environmental damage associated with different sources. Finally, bases for changing the energy-use pattern in the developing countries are discussed, with particular emphasis on renewable sources and nuclear energy.

1. INTRODUCTION

The energy consumption patterns of the world have shown continuous change since the middle of last century. Just over a hundred years ago, nearly 70% of the energy used in the USA was derived from firewood. This was followed by the growth and decline in the use of coal, and currently nearly 80% of energy is derived from petroleum and natural gas [1]. The extensive use of fossil fuels has led to considerable degradation of the environment. Against this background, the contemporary energy scene in the developing countries constitutes a fascinating and challenging study for energy planning with regard to health, environment and economic growth, particularly for the rural districts. This is practically a virgin area where future energy planning can be based on conservation and extensive use of renewable resources in an efficient manner, which would lead to better quality of health and environment than in highly industrialized cities.

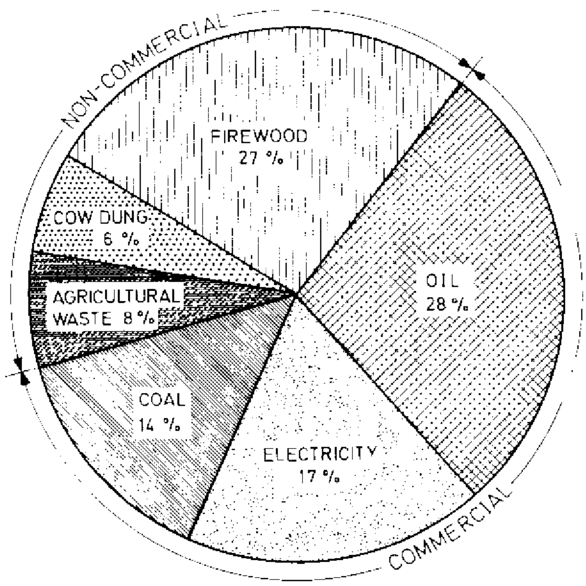

FIG.1. Consumption of commercial and non-commercial energy in India in terms of coal replacement value (1979).

Even though each developing country has a different resource potential and economic base, it may be interesting to consider in detail the energy scenario of India in relation to the impact on health and environment of different methods of producing energy. This is because a very wide spectrum of energy resources is used in India, ranging from nuclear power to the use of dung. Although commercial power production in India is based on technologies essentially similar to those used in the developed countries, energy consumption in most rural areas is based on 'non-commercial' methods, including the use of firewood and farm wastes for domestic cooking and extensive use of direct solar heat and animate energy.

There are nearly 576 000 villages in India, of which only 45% or so receive electricity from commercial power stations [2], mainly those located near urban areas or the power stations. In view of the heavy cost of transmission lines, many remote villages are not served by commercial power stations. Electricity is used mainly for lighting, irrigation pumps, small industries and food-grain processing; it represents barely 1% of the total energy used. Animate energy is also used extensively in the villages for farming and transport. It is estimated that nearly 400 million rural inhabitants derive their energy requirement (mainly household cooking) from non-commercial sources. The adverse health impact of these non-commercial fuels is indeed large. Figure 1 represents diagrammatically the current pattern of energy use in India [3] in terms of coal replacement value, which represents the amount of coal needed to substitute for the energy source in use, as defined later in this paper. All energy-related figures quoted in the text are for 1979.

2. HEALTH IMPACT OF NON-COMMERCIAL ENERGY SOURCES

Non-commercial energy sources provide nearly 50% of the energy for almost half the world's population, including the developing countries of Asia, Africa and Latin America. In India, nearly 41% of primary energy is derived from non-commercial sources. Sufficient data are not available to give a quantitative picture of the health detriment resulting from them. However, the available information on the fuels used, the nature of pollutants released and morbidity and mortality data show that the problem needs urgent consideration. The fuel commonly used is firewood, burnt in open hearths with only 10–15% efficiency. Dried cattle dung and agricultural waste products are also used extensively, the former commonly used for initiating the burning of firewood. With the growth in population, the use of these resources is also increasing. In fact, the major air pollution problem in India is smoke and toxic gases from domestic use of such fuels. The gaseous pollutants released during the combustion of these fuels include CO, SO_2, NO_x, NH_3, HCl and hydrocarbons. The particulate matter released as smoke consists of condensed volatile hydrocarbons, soot and fly-ash particles. Table I gives the estimated emission of different pollutants in $kg \cdot t^{-1}$ fuel for different fuels used [4]. CO is produced owing to incomplete combustion; SO_2 is produced from the oxidation of the sulphur content of the fuels; and oxides of nitrogen are produced in the process of combustion. A variety of volatile toxic hydrocarbons are produced through free radical reactions, particularly the carcinogenic polycyclic aromatic hydrocarbon, benzo(a)pyrene.

The indoor air concentration of different pollutants depends on the degree of ventilation inside the dwellings. A limited survey of rural households conducted in the State of Gujarat, India, has shown a large range of pollutant concentration. The survey included 200 houses using non-commercial fuels in a variable mix and with different degrees of ventilation. Table II gives the range of concentration of SO_2, NO_x, B(a)P and suspended particles in these houses [5]. It is noteworthy that the maximum values far exceed the air-quality standards. The average daily cooking time spent indoors ranged from 6 to 8 hours, and the exposure was mainly incurred by housewives and children.

The commonly reported health effects of exposure to smoke and gases during domestic cooking are asthma, chronic bronchitis, emphysema, chronic *cor pulmonale*, and cataract of the eyes. *Cor pulmonale* is caused by chronic obstructive pulmonary disorders, and accounts for rising morbidity trends in a number of

TABLE I. ESTIMATED EMISSIONS OF MAJOR POLLUTANTS FROM NON-COMMERCIAL SOURCES ($kg \cdot t^{-1}$ fuel)

	Firewood	Dry cattle dung	Agricultural waste products
Particulates	31.4	13.1	30.0
Organics (including hydrocarbons)	23.5	9.8	22.0
Sulphur dioxide	19.6	8.2	18.8
Nitrogen oxides	3.9	1.6	3.8
Carbon monoxide	1.6	0.7	1.6
Hydrogen sulphide	1.2	0.5	1.1
Hydrogen chloride	1.2	0.5	1.1
Ammonia	1.2	0.5	1.1
Totals	83.6	34.9	79.5

countries in the South-East Asia region. An extensive study was carried out between 1962 and 1974 on the etiopathology of chronic *cor pulmonale* in India with support from the National Institute of Health, Bethesda, MD, USA, and other agencies [6]. In this study, a very high incidence of *cor pulmonale* was reported from two hospitals in Delhi. The patients were mainly women from rural areas using non-commercial fuels for cooking. The incidence was found to be particularly high in northern India, probably because of the colder climate during winter and frequent smog. Nearly 17% of 2360 cardiac cases admitted to two general hospitals in Delhi were diagnosed as *cor pulmonale;* nearly 17–31% of cardiac deaths were due to *cor pulmonale.* This survey also showed that women developed *cor pulmonale* early in life and died young. Patients were exposed to these primitive fireplaces from childhood and suffered from chronic respiratory problems all the time. The occurrence of the disease in women 10 to 15 years earlier than in men established the basic cause as air pollution from fuels used for domestic cooking. Cigarette smoking was found to be a contributory cause in men. Women patients also showed cardiac enlargement, excessive derangement of pulmonary function and more severe congestive heart failure [7].

TABLE II. RANGE OF CONCENTRATION OF SO_2, NO_x, B(a)P AND SUSPENDED PARTICLES IN 200 HOUSES SURVEYED IN GUJARAT STATE, INDIA

	Maximum value	Minimum value	Ambient air-quality standards
Sulphur dioxide ($\mu g \cdot m^{-3}$)	1100	50	365 (US)
Oxides of nitrogen ($\mu g \cdot m^{-3}$)	1300	12	100 (US)
Benzo(a)pyrene ($\mu g/1000\ m^3$)	9317	1270	10 (FRG)
Suspended particles ($\mu g \cdot m^{-3}$)	1500	500	150 (US)

Note: The ambient air-quality standards are for daily mean values.

Similar reports of domestic-energy-related health detriment have been received from other developing countries of South-East Asia, including Thailand, Burma, Nepal and Bangladesh [6]. A survey conducted in the Fiji Islands also showed high incidence of chest pain and eye irritation attributed to smoke from the burning of firewood [8]. Estimates based on the surveys conducted in northern India and Nepal show that nearly 1–1.3% of the rural population surveyed suffered from *cor pulmonale* [6]. The incidence of chronic bronchitis and emphysema was much higher. In terms of health detriment per unit of energy per person, the detriment from this source is orders of magnitude higher than from the commercial production of electricity. The increasing population and sharing of dwellings by a larger number of people must certainly have increased the magnitude of the problem. It is therefore important to consider an alternative which would bring relief from this source of suffering to the rural population.

The production and use of biogas is one such alternative. It has been estimated that full-scale adoption of biogas could meet at least 50% of the rural energy requirement for the domestic sector.

In addition to health detriment, extensive use of firewood has led to soil erosion, deforestation and resulting floods and climatic changes. The annual damage caused by floods has shown a rising trend for the last 30 years [2]. The use of non-commercial fuels has also led to scarcity of firewood in rural areas.

Firewood plantation has been proposed for some regions of the country. This, combined with large-scale adoption of biogas technology, could help in conservation of forests.

2.1. The biogas energy alternative

Biogas can be generated in simple gas plants by anaerobic fermentation of the slurry made by mixing water with wet animal dung and some farm wastes. The gas is mainly methane (50–60%) with impurities of CO_2, N_2 and H_2, and trace levels of CO, O_2 and H_2S. The fuel value of the biogas is comparable to that of coal, with nearly twice the efficiency in utilizing the heat output. It is a clean fuel with no polluting smoke. This energy source has therefore been suggested to partially replace the direct burning of dung cakes and farm waste. From the point of view of health, this would also ensure hygienic surroundings free from mosquitoes, flies, and insects that breed on wet manure heaps.

The biogas alternative is particularly appropriate since India has the highest cattle population in the world. About 235×10^6 bovine cattle in India [9] yield nearly 1200×10^6 t of wet dung annually. This quantity of dung has biogas potential equivalent to 20×10^6 t of oil per year, which is equivalent to 80% of the country's consumption of oil and nearly twice the indigenous production of oil in 1979.

The installation of biogas plants first started in India in 1962/1963. This work was entrusted to the Khadi and Village Industries Commission constituted by the Government of India and also subsequently to the Department of Science and Technology of the Government. The programme included help in design and installation of plants and the grant of Government loans and subsidies to persons wishing to install these plants. Today nearly 80 000 biogas plants are in operation in the country [10]. During the current five-year plan, a target of nearly half a million new plants has been set.

Typically, the biogas plant for a small family of four to five persons can be built by using dung from three cattle – two cows and a calf – yielding 20–25 kg of wet dung per day from which nearly 1.5 m^3 of biogas can be generated daily for domestic cooking and lighting. The manure can be used in the family kitchen garden for growing vegetables, and the surplus could be sold to small users. However, large-scale adoption of family-size plants is not economic. The medium-size plants can cater for five to ten families, and the large village-size plants are built to handle almost a tonne of dung per day from 150 to 200 farm animals. Enough gas pressure is generated in large plants to use gas pipelines up to 1.5 km long. Larger plants are in fact a good source of organic fertilizer for village farming needs. Such plants can be built to cater for the fuel and manure needs of individual villages. They would also help to improve sanitation in the villages. In addition, their construction and operation are both labour-intensive and appropriate for rural areas of India.

In assessing the positive health impact of the biogas alternative, we must consider the decrease in morbidity and mortality due to cleaner cooking conditions free from smoke, as well as the lesser problem of mosquitoes, flies and insects, which take a sizeable toll in rural areas through the spread of parasitic diseases. Baseline data need to be generated for the current incidence and pattern of diseases in the rural communities through programmes of epidemiological studies.

3. THE USE OF COMMERCIAL ENERGY IN INDIA

The primary sources of commercial energy in India are coal, petroleum products, natural gas, hydroelectricity and, to a limited extent, nuclear power. The health assessment for these will now be discussed. Electric power stations are mainly based on coal and hydroelectricity, contributing 56% and 37% respectively to the installed capacity [11]. Petroleum products are mainly used for transport, agriculture and the household. The transport sector accounts for the major consumption of petroleum products. During the two decades before the oil crisis, the use of petroleum products, both for transport and industry, increased, owing to their cheapness and convenience. Coal locomotives were replaced by diesel locomotives, and coal-fired furnaces in textile mills and other industries were replaced by oil-fired furnaces. Petroleum products have been little used for generating electricity.

In terms of coal replacement value, nearly 47% of the total commercial energy is contributed by petroleum products, 24% by coal and 29% by electricity [3]. The present dependence on oil is indeed large and one may estimate a shift in the primary energy use from oil to coal, and to nuclear energy on a long-term basis. Health assessment for the use of coal, and methods of reducing the health detriment from this mode of primary energy, must be considered on a priority basis.

The contribution of nuclear power to commercial energy, which is only about 3% at present, would continue to grow slowly and would become significant only by the end of the century. Since the health detriment from the use of nuclear power is very small compared to coal [12,13], its contribution to commercial electricity in India may become comparable with other sources after a few decades.

3.1. Assessment of health detriment from the use of coal

India has large deposits of coal with total reserves for depths up to 600 m estimated at nearly 81×10^9 t [14], of which only about 30.9×10^9 t are recoverable. The current annual production is in the range of 100 to 105×10^6 t. Of this, about 34×10^6 t of coal are used for nearly 80 thermal power stations of capacity ranging from 10 to 700 MW(e) each, with total capacity of nearly 17 GW(e). In addition to generating electricity, coal is used in the steel industry, railways, cement production, textile and other industries, and for domestic cooking in urban as well as rural areas. About 7 to 8 $\times 10^6$ t of coal are used for domestic cooking. The use of coal for electricity production would continue to increase, at least for the next few decades. With this background, we may now look at the assessment of health detriment associated with the use of coal in India.

It is well known that coal has the greatest potential for environmental damage. Coal smoke and fly-ash from power stations and industry in the urban areas give rise to all the major categories of pollutants, including trace metals, carcinogenic hydrocarbons and a large number of gaseous pollutants. The sulphur content of most types of coal is in the range of 0.3–5%, leading to significant releases of SO_2 in the stack effluents. Nearly 40 other elements are present in coal, including cadmium, mercury and the radioactive elements radium, uranium and thorium. Accumulation of cadmium and mercury in the biosphere through long-term use of coal for electricity generation can cause serious health concern. The likely health detriment associated with the radioactive content of coal has received considerable attention in recent years. Mining and transport of coal also give rise to health detriment.

Assessment of health detriment in India must be carried out for general environmental pollution from the use of coal for power production and industry, as well as for indoor pollution from heavy domestic use of coal. The air pollution problem has reached an alarming state in many Indian cities, particularly with regard to particulate pollution. Most of the thermal power stations are of older design and are not provided with electrostatic precipitators for removing fly-ash. Only the newer stations have control equipment for reducing particulate pollution.

No epidemiological studies on the health detriment from urban domestic use of coal have yet been reported, and there is an urgent need for such studies. The statistics from major hospitals in large cities show very high incidence of respiratory disorders in the exposed population groups [15].

3.2. Assessment of health detriment from the use of oil

Efficient combustion of oil gives very little polluting hydrocarbons and CO. However, SO_2 and NO_x are produced in varying concentrations, the former depending on the sulphur content of the oil and the latter on the temperature of combustion. Pollution problems arise mainly from the incomplete combustion of oil, as in the case of automobiles and industrial-oil furnaces. The pollutants are generally classified as gaseous hydrocarbons, including ethylene, propylene, etc.; particulate hydrocarbons, including the carcinogenic polycyclic aromatic compounds; and inorganic gases, including CO, NO_x and SO_2. Many toxic gases are also formed by photochemical oxidation reactions after the release of primary products into the atmosphere, particularly the formation of acrolein, peroxyacetyl nitrate (PAN), and formaldehyde. Ozone is also formed in these reactions in fairly large concentrations.

The annual consumption of oil in India for commercial energy is around 24 million tonnes [3]. According to recent estimates, India has oil reserves of nearly 1500 million tonnes. The indigenous production is likely to increase considerably in the coming years. Nearly 70% of petroleum products are used in the transportation sector alone, and only 8% are used for power generation. Only 5% of oil is used as petrol, mainly for cars plying in the urban areas. In fact, the urban air pollution is mainly caused by petrol-driven cars and diesel-driven trucks. The pollutants are released near ground level and there is very little dilution. The tuning of vehicles is generally not satisfactory, and this leads to abundant smoke in the high traffic density areas with high concentration of carcinogenic polycyclic aromatic hydrocarbons. The problem is more acute in the densely populated cities, Bombay and Calcutta.

Preliminary health surveys carried out in Bombay and Ahmedabad have shown some correlation between the concentration of pollutants and mortality and morbidity [15]. Detailed epidemiological studies are projected at Bombay under an IAEA Research Contract. Under this contract, detailed measurements of the polycyclic aromatic hydrocarbon B(a)P have been carried out at different locations in and around the city of Bombay, representing (a) a locality with high traffic density, (b) a locality with industries using coal, and (c) a suburban locality. In the area with predominant traffic density, nearly 80% of the B(a)P was found to be on respirable-sized particles, with concentrations exceeding the air-quality standards during the winter months [16].

3.3. Health aspects of the use of natural gas

Natural gas, which contains 80–90% methane and heavier hydrocarbons, is known to be one of the least polluting fuels. Particulates and SO_2 are minimal in natural gas used for thermal power stations. However, the NO_x release is comparable to that of coal and oil, depending on the temperature of combustion. Natural gas also contains a small quantity of radon, which is too low to be significant enough to create adverse radiological health effects [17]. Risk estimates for energy alternatives have shown that natural gas has the lowest overall risk associated with its use [13].

The known resources of natural gas in India are very limited. After the discovery of natural gas in the Bombay High region, two thermal power stations in Bombay operating on coal were switched over to gas, with very significant decrease in pollution. Although the present wells have a limited gas supply, Bombay High has a large potential for natural gas. It is also proposed to supply this gas for household cooking in the Maharashtra State.

3.4. Risk assessment for hydroelectric power generation

The main risk associated with the generation of hydroelectric power is accidental death during dam construction. It has also been suggested that earthquakes may be caused by dams in certain zones. Parasitic diseases have been reported in some communities affected by the change of irrigational practice after the impounding of river water. In comparative terms, however, health detriment caused by the production of hydroelectric power is very small.

There are nearly 100 hydroelectric stations in India [18], including large stations, medium-size stations, and microhydro stations with capacity of only 1 to 2 MW(e). The largest station in India is at Bhakra Nangal, with total installed capacity of nearly 1200 MW(e). The installed hydroelectric capacity for the entire country is nearly 11 GW(e). The hydropower potential of India is nearly 80 GW(e) installed, based on Ref.[3]. There is scope and urgent need for increased generation of electricity from this clean and renewable source.

No systematic data are available for risk assessment of this source, nor have there been reports of major health problems. An earthquake with its epicentre near the Koyna hydroelectric dam, near Bombay, was reported to have been caused by the dam, but there has been no detailed study of this.

3.5. Risk assessment for the development of nuclear power

Several studies have been made on the relative risks from generating electricity by different methods, and it is generally concluded that nuclear power generation has the lowest component of associated risks [12,13]. This is mainly due to extensive health and safety measures adopted in all stages of the nuclear fuel cycle. The safety record of the operation of nuclear power stations has been generally good.

In the developing countries, the progress of advanced nuclear technologies is dependent on the progress of a conventional engineering base, both of which have shown a steady growth in India. Although nuclear power at present constitutes only a small fraction of the installed capacity, a large nuclear power programme has been projected in the national plan. The presently known reserves of uranium in India are not large, and extensive prospecting is continuing. Fuel availability may therefore not be an impediment in the development of nuclear power. With the development of breeder-reactor technology, including fast breeders and thermal breeders, even the known reserves of uranium and thorium would constitute an inexhaustible source of energy, and the advanced nuclear fuels will become competitive in the international market. Health and safety aspects of the production and use of nuclear power have developed considerably in India, and progress has also been made in engineering safety in the design and operation of nuclear power stations.

Nuclear power programmes have been launched by several developing countries which have significant commitment to this source of energy, in spite of severe constraints such as engineering infrastructure, skilled manpower and funding [19].

4. HEALTH AND FUTURE ENERGY STRATEGIES

The above health assessment for different sources of energy provides a very good base for projecting future energy strategies for the developing countries. Non-commercial sources used by rural communities present the greatest health hazard. The use of coal for commercial energy has also led to significant health detriment in the world population; new technologies for its use would ensure greater safety in future. However, in view of the high capital costs of introducing these new technologies, their adoption in the developing countries would be slow. In the future energy choices, it may be prudent to place greater emphasis on the renewable sources for rural populations and on advanced coal stations and nuclear power for the urban areas. Development of nuclear power has been slow, and significant commercial use of such stations would still take a few more decades. On a long-term basis, breeder-reactor programmes must also be developed. These choices will be examined in greater detail.

The renewable sources, including solar energy, wind, tides and biogas, have limited potential for commercially-usable energy. Hydroelectric power potential is large in India and it can be economic in remote hilly regions by cutting down the cost of transmission lines. Biogas in rural areas may be a good alternative to burning firewood and farm wastes, but it cannot meet the entire energy needs for diverse applications. The energy from biogas may, in fact, be considered as a byproduct of a method of treating biological wastes and manufacturing high-quality organic fertilizers. Their economic viability should not be judged merely on the basis of energy and manure output.

4.1. Health and cost-effectiveness of energy choices and projections

It is important that any strategy proposed for the developing countries must be cost-effective, in addition to improving health standards. The non-commercial fuel is obtained essentially free of cost, and no cost/benefit analysis can be carried out for the alternatives in this sector. Emphasis should be placed on renewable and labour-intensive sources. The development of biogas plants is indeed a step in the right direction, and effort and investment in this source should probably be stepped up considerably.

For cost- and health-effective future projections, due consideration must be given to past trends. Figure 2 gives the trend in the consumption of commercial energy in India during the period 1953–79 [3], including coal, oil and electricity.

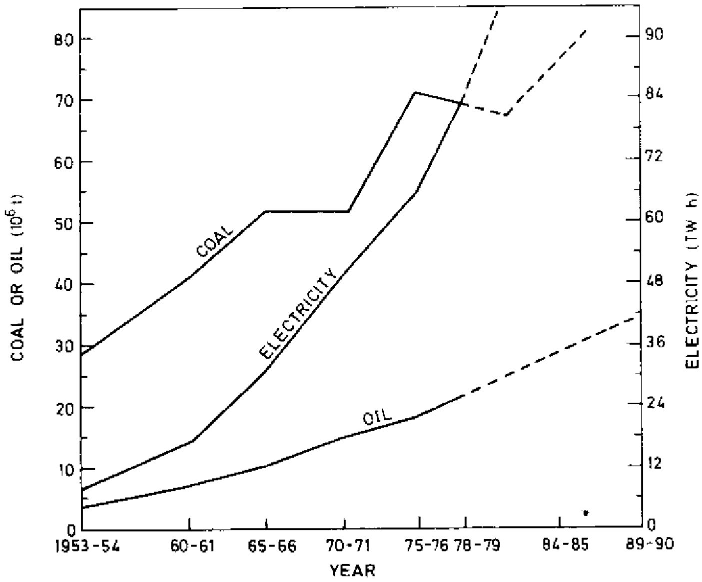

FIG.2. Trends in the consumption of commercial energy in India.

The unit of measurement used in this graph is the coal replacement measure, which is generally used for energy-related issues in India. This unit takes into account the efficiencies involved in typical cases of substitution. The conversion factors are given in Ref.[3]. The consumption of coal excludes coal used for electricity generation; oil consumption excludes oil used for non-energy purposes, for electricity generation and in refineries; electricity consumption includes supplies from non-utilities and excludes transmission and distribution losses. Figure 3 gives the trend in installed electric capacity during 1960–79, with contributions from thermal, hydro and nuclear generation [3]. The dashed lines show extrapolation of the past trend. It is seen that there has been a nearly five-fold increase in installed capacity in two decades, with the major contribution from coal-based stations. Against this background we can consider some suitable models for energy planning in the developing countries.

4.2. Models for a cost- and health-effective energy mix for developing countries

Energy systems with stringent pollution control may not be generally cost-effective for developing countries. It is therefore necessary to develop models for an energy mix, so that optimization can be made for any country or region

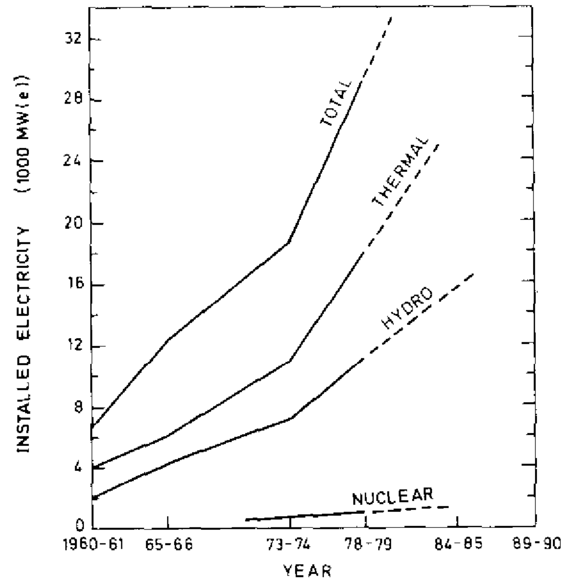

FIG.3. Trends in the growth of installed electricity in India.

with a given set of conditions. In this process of optimization, the health detriment from a given system can be taken into account. Since quantitative data on health detriment for different alternatives are not available, a simple approach would be to divide the energy systems into categories on the basis of their performance with regard to health. The following two categories may be assigned to the energy sources:

Low detriment	High detriment
Natural gas	Direct use of coal
Hydroelectric power	Coal thermal plants
Biogas from farm wastes	Firewood and farm wastes
Nuclear power	Direct use of oil

Depending on the availability of raw materials and technology, the optimized energy mix can be obtained by a simulation model, taking into account all possible inputs for each of the sources, including weighting for health detriment.

Figure 4 gives past trends and future projections for energy supply through the year 2020 from different sources, including projections for electricity supply and different fuels used for industry, transport and household. The trends of

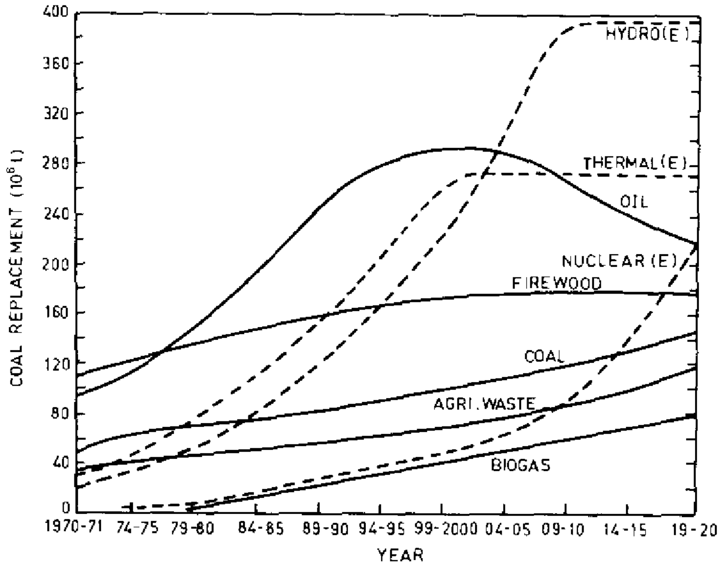

FIG.4. Forecasts of use of different sources of energy in India through the year 2020, based on resource potential and environmental health considerations. Electrical energy (E) forecasts are shown by dashed lines. The conversion factor from TW·h to coal replacement value for electrical energy is taken as unity (Ref.[3]).

energy supply for the decade 1970–80 have been used as the baseline for these projections. Due consideration has been given to a variety of complex factors. The environmental impact and supply limitations in the case of coal and oil have been taken into account in these projections. The general considerations for individual energy sources are explained below.

In the case of oil, used mainly for the transport sector, the compounded growth rate of 5% per annum is considered up to the year 1990, reduced to 2% for the period 1990–1995, further reduced to 1% per annum from 1995–2000, and saturated during the period 2000–2005. From 2005 onwards, a reduction of 2% per annum is considered, up to the year 2020. This may be considered only a conceptual model which takes the resource limitation and constraints of import into account. It also takes into account the need for optimizing the consumption of oil for urban transport in order to reduce the air pollution problem.

In contrast, an increasing growth rate has been projected for nuclear power. The past slow trend of growth increases on the basis of 10 000 MW(e) to the year 2000 projected in the national plan, followed by 6% compounded growth to the year 2010 and 10% growth rate to the year 2010–2020. This increase in nuclear power generation would considerably cut down the use of coal, oil, firewood and agricultural wastes, thus making these resources available for non-energy uses as well as improving the quality of the environment. After the establishment of a strong industrial infrastructure for making the components of nuclear power stations, the anticipated growth should be achievable, and may even be exceeded.

A fascinating part of the projections given in Fig.4 is the use of energy from biogas, starting with barely 0.4×10^6 t of coal replacement for the year 1979/80, a hundred-fold increase is projected through the year 2000, followed by the same trend. Such a growth would be possible through improved engineering of biogas plants and large-scale use of village-level plants. Although the contribution of biogas to the overall energy consumption is very small, it has several advantages, as already detailed.

In view of the large hydropower potential in India, 7% growth has been projected throughout, saturating at around 2010, when most of the hydro potential would have been utilized. Earlier utilization of hydro potential has advantages for health and environment because any delay in its utilization would mean burning additional coal and oil. The major constraints in the utilization of hydropower potential have been higher capital costs and very long construction time.

The growth of electricity from thermal power stations, mainly based on coal, is also taken as 7% per annum up to the year 2000, when it saturates. Although the coal reserves of India are large, the high-grade coal required for power generation is limited and it would not be possible to support a continuous growth. In fact, a downward trend in the use of coal beyond the year 2000 is also likely.

The rate of growth in the consumption of firewood has been 2.5% per annum during the period 1970–80. It is projected to decrease to 1% during the 1980s and to continue up to the year 2000 when it levels up. It is anticipated that with the process of development, firewood plantations will be initiated on a large scale to conserve the forests, and indoor use of firewood would be replaced by more efficient fuel biogas to reduce the air pollution problem. In the rural areas, where sufficient waste firewood is available, its domestic use may be continued in well-ventilated dwellings.

The direct use of coal in the steel industry, railways and other sectors is projected to increase only at the rate of 2% per year compounded, compared to the average rate of growth of 6% during the last two decades. This decrease in the rate of growth of coal consumption may be compensated by greater availability of electric power and increased use of electricity for transport and industry.

The use of agricultural waste products, excluding animal dung, is projected to continue at the past rate. It is anticipated that utilization would be directed more towards brick-making industries, potteries, etc.

The pattern of growth of energy depicted in Fig.4 is rather conservative. Savings in energy may also be effected by using newer methods of conservation for both electricity and direct fuels. This is particularly desirable for environment

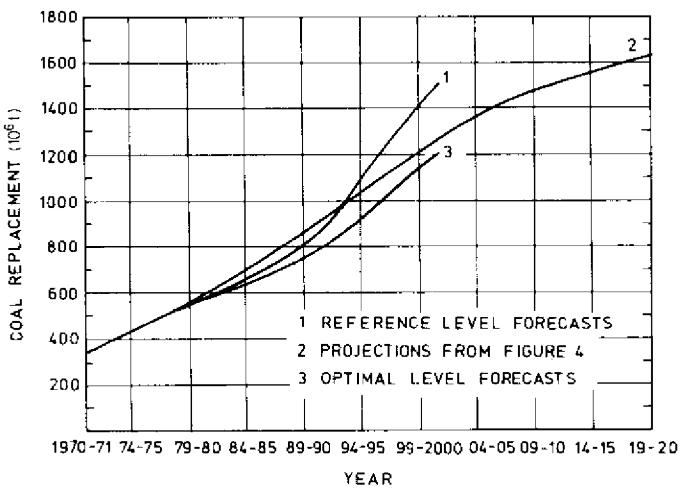

FIG.5. Projections for total energy utilization, based on Fig.4, and reference and optimal level forecasts based on Ref.[3].

and health. It is estimated that, by using conservation methods, 5–10% saving in the consumption of energy can be achieved, which is equivalent to the average projected annual increase in the production of energy.

Figure 5 gives the growth of total energy production from different sources, based on Fig.4. It also includes the energy demand forecasts given in Ref.[3], for different models. The projections based on Fig.4 give values in between those obtained from the models used by the Working Group on Energy Policy of the Planning Commission, Government of India [3].

There are several other interesting features of the forecasts projected in Fig.4. They are based on faster growth of electricity compared with other sources of energy, since this is a convenient form of energy for the consumer and it also permits better control of environmental pollution and less dependence on non-commercial fuels. The projected growth of hydro and nuclear power is faster than in the past. Although both require a strong technological base and heavy capital investment, they do not involve large-scale fuel transport. The projected growth of thermal power stations based on fossil fuels (mainly coal) is also fast up to the year 2000, for two main reasons. Firstly, this is at present the only source based on plentiful availability of fuel in the country, and, secondly, the industrial infrastructure has already been developed for these stations. With the development of a stronger technological base, faster nuclear power growth beyond 2000 as projected in Fig.4 can then be achieved.

Faster growth of electricity is particularly good for overall reduction of the health detriment caused by the use of energy in all forms. The changed scenario in the year 2020 as depicted in Fig.4 shows less dependence on coal, oil, firewood and farm wastes, and complete absence of the use of dung as direct fuel. Plentiful biogas would be available from the treatment of dung and farm wastes, and this could be used as a source of energy, in addition to improving sanitation in the rural areas.

5. CONCLUDING REMARKS

In the long-term use of fossil fuels, we should also consider the problems of increase in atmospheric CO_2 and the resulting increase in temperature. The direct release of waste heat in the water bodies and atmosphere is an additional constraint in the use of fossil and nuclear fuels. These problems have to be viewed in terms of long-range global use of energy. Although the renewable energy sources do not present these problems, they have only limited commercial energy potential. Therefore, the use of fossil and nuclear fuel must be optimized with due consideration to health and environment. In this regard, energy conservation must be equally borne by the developing countries and by the industrially advanced nations. Innovations in energy conservation may be considered as important as the development of new technologies for energy generation.

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DISCUSSION

E. EL-HINNAWI: The health implications of non-commercial sources of energy are not as simple as you suggest. There are, for example, health hazards associated with firewood collection — snake-bites, scratches, wounds and so on and the operation is mainly carried out by women and children. On the other hand, burning agricultural wastes and firewood is a common practice in many villages to repel mosquitoes and flies. This reduces the incidence of malaria and also the amount of insecticides imported, with the result that the environmental consequences of these chemicals are lessened.

There is, therefore, a need for careful assessment of the health implications of non-commercial sources of energy; cost/benefit analyses of present practices should be made, and new methods of reducing hazards should be studied.

K.G. VOHRA: These general comments are valid, particularly when we are considering a wide spectrum of rural areas in the developing countries.

A.K. BISWAS (*Chairman*): Dr. El-Hinnawi has raised an important point concerning both the accuracy of the methodologies applied to estimate the use of non-commercial sources of energy at national levels and the health risks associated with such uses. Most such estimates should be seen at present as no more than the best informed guesses available, which must be continually updated as further data are collected. A number of the health risks accruing from the use of non-commercial energy sources are currently under consideration, but certainly not all. For example, in Africa women and children undertake 90% of journeys to collect firewood and water Dr. El-Hinnawi has in fact just referred to this kind of thing. They are thus more exposed to health risks than men. Though significant health risks are associated with these journeys, they are seldom considered since quantitative data are not available. A recent study estimated that the incidence of Gambian trypanosomiasis could be reduced by 80% if journeys to collect firewood and water were eliminated, and the main beneficiaries would be women and children. While I personally consider the estimate to be somewhat on the high side, the health risks involved are undoubtedly considerable. Attempts should thus be made to incorporate these risks in future analyses.

S.R. BOZZO: Studies of the incidence of rheumatic fever, which can give rise to *cor pulmonale*, have been conducted in rural India to explain the high incidence of the latter. Was there a correlation with rheumatic fever in the studies you discussed?

K.G. VOHRA: No. The studies I discussed were not correlated with rheumatic fever.

B. SØRENSEN: Could you indicate the total accumulated occupational man-rem exposures for the civil nuclear programme in India?

K.G. VOHRA: They are well below the levels set by the International Commission on Radiological Protection.

I.M. TORRENS: In many villages in India and Nepal, indoor air pollution which is due to the burning of non-commercial fuels and which gives rise to lung and eye diseases could be largely controlled by raising the primitive level of technology used — for example by building chimneys in houses instead of having only a hole in the roof. In these circumstances, would it not be wiser to concentrate manpower and financial resources on helping to raise the educational and technological level rather than on epidemiological analysis? The latter is normally used to detect a correlation, while in this case the problem and the solution are evident.

K.G. VOHRA: In rural areas where building material is available, houses are provided with chimneys. The problem mainly exists in the remote rural areas where material is scarce and houses are constructed with mud walls and thatched roofs, and normal chimneys cannot be built. Arrangements for ventilation are improvized and are not efficient.

With massive rural education programmes, improvements are taking place in many rural areas. In several of these areas, even non-commercial resources are scarce and cannot be used extensively for making construction materials such as bricks or wood panels to build houses with efficient chimneys. Clearly, the epidemiological studies for the rural areas I mentioned are not suggested as an alternative to the educational and technological developments which are taking place.

ASSOCIATION OF COAL, ATOMIC ENERGY, SOCIO-ECONOMIC AND OTHER ENVIRONMENTAL VARIABLES WITH THE RISK OF DYING

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Abstract

ASSOCIATION OF COAL, ATOMIC ENERGY, SOCIO-ECONOMIC AND OTHER ENVIRONMENTAL VARIABLES WITH THE RISK OF DYING.

Death rates have been calculated by cause, age, sex and race for each county, state economic area, and other groups of counties of the USA for the period 1959 to 1976, data being obtained from official sources. Attention has been given to appropriate methods of using these rates for epidemiological study. Within the USA there are marked and consistent differences in death rates for middle-aged white males: low in the west central plains area and high in the south-east coastal plain. if the USA as a whole had rates as low as in the low-rate areas, there would be 160 000 fewer deaths per year under the age of 75. Coal- and metal-mining is strongly associated with high rates for middle-aged females as well as males. The residents of an area around a nuclear plant, after 22 years of exposure to low levels of radiation from this plant, do not show clear evidence of either decreased or increased risk, for various forms of cancer and other causes, as compared with rates for those living further away in the same states, or as compared with US rates. In the USA, persons living at higher elevations tend to have lower death rates. A number of factors may be hypothesized as being responsible for these lower rates, including the higher levels of background radiation at higher elevations. Many other factors also present evidence of being associated with differences in risk, including cigarette-smoking and various socio-economic or cultural variables.

1. INTRODUCTION

There are substantial variations from one area to another within the USA in the risk of dying – presented in reports by the US National Center for Health Statistics (NCHS) and others [1–6]. Our previous reports also show marked geographical differences in the risk of dying, also using age-sex-race-specific and age-adjusted death rates [7–9].

For all diseases combined, death rates for white males age 35–74 show marked differences, with the highest-rate areas in south-eastern USA in some instances experiencing a risk 35–100% greater than that in the lowest-rate areas (Fig. 1). For several different time periods, our calculations have repeatedly

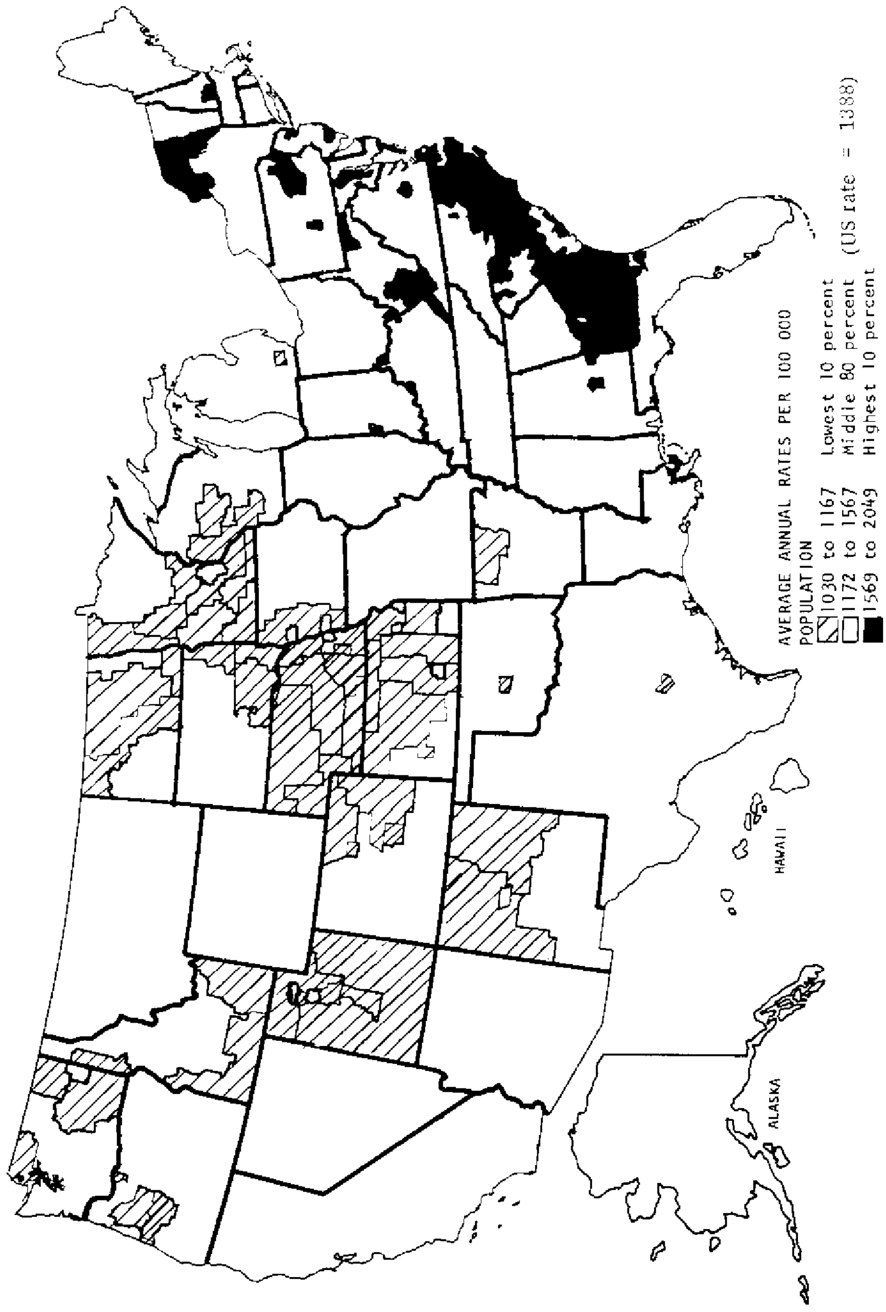


FIG.1. Death rates for all diseases combined, white males, aged 35 to 74 (age-adjusted) by state economic areas, 1959 to 1976.

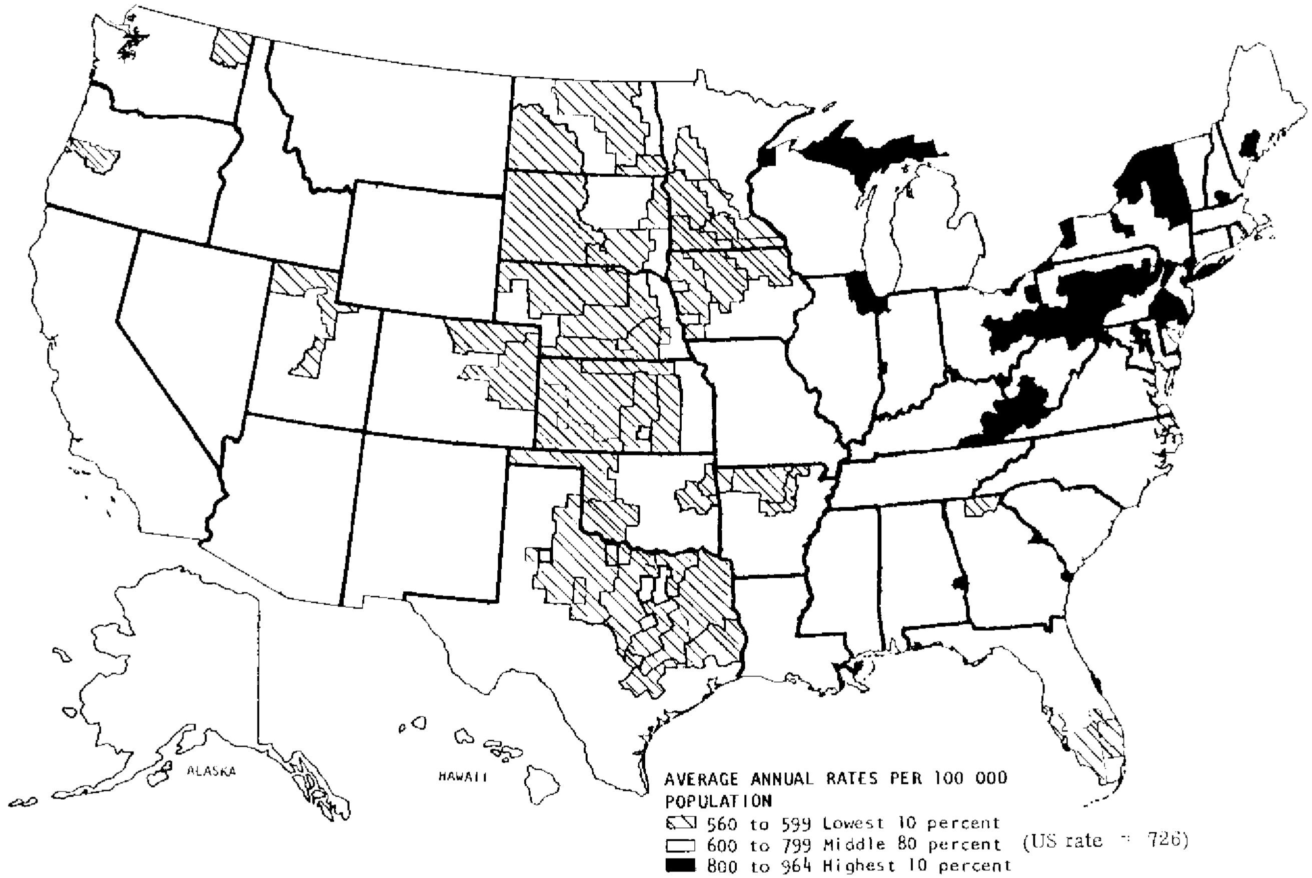


FIG.2. Death rates for all diseases combined, white females, aged 35 to 74 (age-adjusted) by state economic areas, 1959 to 1976.

shown that, if the factors responsible for the lowest death rates could be identified and if the USA could thereby achieve the low mortality experience of these lowest-death-rate areas, there would be 160 000 fewer deaths per year under the age of 75 [8–10].

Our concern is with the general question: what are the environmental factors causing these differences in death rates? More specifically, is there an association with the risk of dying of factors such as coal-mining, metal-mining, radiation from atomic energy or nuclear plants, elevation above sea level, socio-economic and other factors?

2. MATERIAL AND METHODS

2.1. Mortality

Data on each death were obtained from death detail tapes from NCHS for each year from 1962 to 1976 and tabulations made by us. For 1959 to 1961 and 1949 to 1951, counts were obtained from CNHS and other US Public Health Service offices. We obtained tapes containing population data from the US Bureau of the Census.

Emphasis has been placed on white death rates by sex for ages 35 to 74 age-adjusted by 10-year age groups by the direct method, to control for variations in age distribution from one area to another. Age 35 to 74 has been selected to maximize geographic differences along with minimizing standard error. For white males, the rates for the areas for the 18-year period 1959 to 1976 used for Fig.1 have, on the average, a standard error of less than 1% of the rate. A similar map for white females (Fig.2) is based on rates with standard errors averaging about 1% of the rates.

We have also calculated death rates for more than 50 causes of death, by age, sex and race, for each state, county and state economic area (SEA) of the USA, consisting of (a) 207 metropolitan areas within states and (b) 303 groups of similar non-metropolitan counties, as defined by the Bureau of the Census [11].

Substantial consideration has been given to possible systematic error or bias. Procedures have been developed and applied to minimize such possible error, particularly for handling cause of death, age, and usual residence problems [3, 9, 12, 13].

2.2. Other variables

Socio-economic and other data have been obtained from the Bureau of the Census, NCHS and other sources.

The hypothesis that mining or history of mining is associated with high death rates was suggested by the high rates in the east central Pennsylvania coal-mining areas for 1949 to 1951 and other early studies (Fig.3). Counties in the sample were chosen on the basis of their rates, 100 lowest rate, 100 nearest-the-median rate, and 100 highest rate. The geologist participating in the study classified each county as to mining status, without knowledge of its death-rate status [9, 14].

Analyses of the closeness of the associations of many independent variables with various death rates have relied heavily on the product moment or Pearsonian coefficient of correlation, r, on multiple correlation, R, and chi-square.

3. RESULTS

3.1. Mining

Of the 100 counties with the highest death rates for all causes for either sex, whites age 35 to 74, for 1959 to 1969, 35 were classified as moderate or heavy mining or history of mining (coal or metal) while of the 100 lowest-rate counties (not shown) only three were so classified (Fig.4) [14]. This is obviously a statistically significant contrast by any conventional standard. Of these 35 high-rate counties with substantial mining, 29 were selected on the basis of the female rates and 15 on the basis of the male rates. Since mining is primarily an occupation of males, and female rates are even more clearly associated with mining than are male rates, the association, whatever it may be, is more than just occupation.

3.1.1. Coal mining

Of the 35 highest-rate counties classified as having substantial mining, 14 are coal-mining counties, of which 13 were identified on the basis of their high rates for females for 1959 to 1969. In counties with small to moderate populations there will obviously be considerable chance fluctuation or standard error. Yet in the period 1968 to 1972, all 14 counties had female rates placing them among the 10% highest in the USA, and in the more recent period, 1973 to 1976, 13 of the 14 counties were in the '10% highest' group – versus an expected 1.4 counties if death rates were randomly distributed by county.

Five of the counties were in the highest 3% of the rates for each sex for 1973 to 1976 whereas, by chance, the expected number would be only 0.4 of a county. A further observation: of the 14 highest-rate counties classified as coal, in the period 1959 to 1969 only five were selected on the basis of the

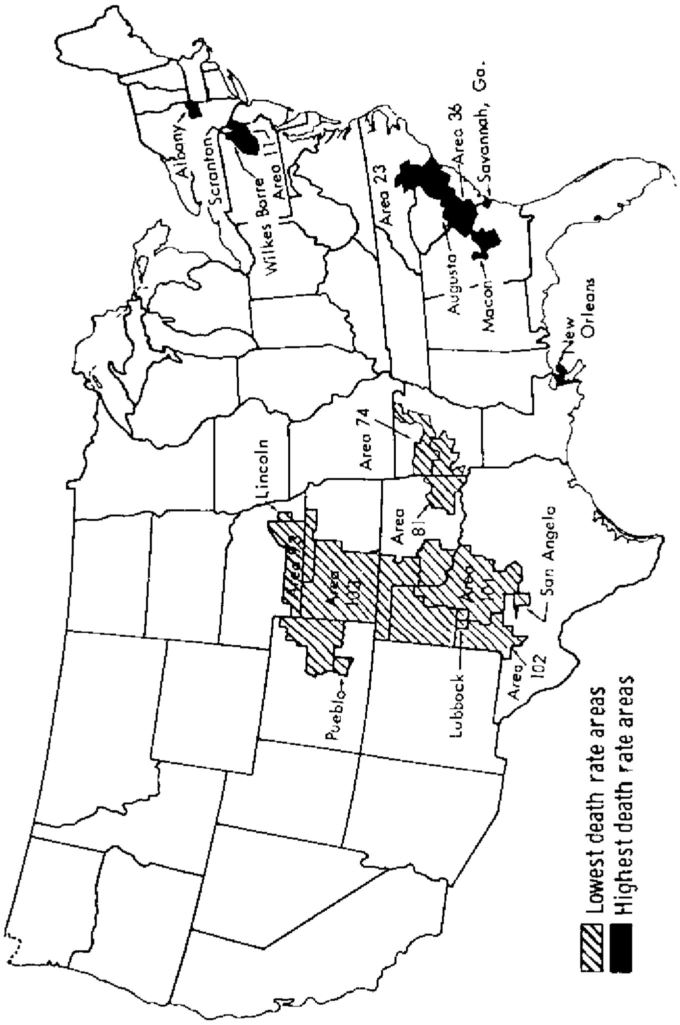


FIG.3. Ten lowest and ten highest death rate areas for cardiovascular diseases, whites, aged 45 to 74, 1949 to 1951.

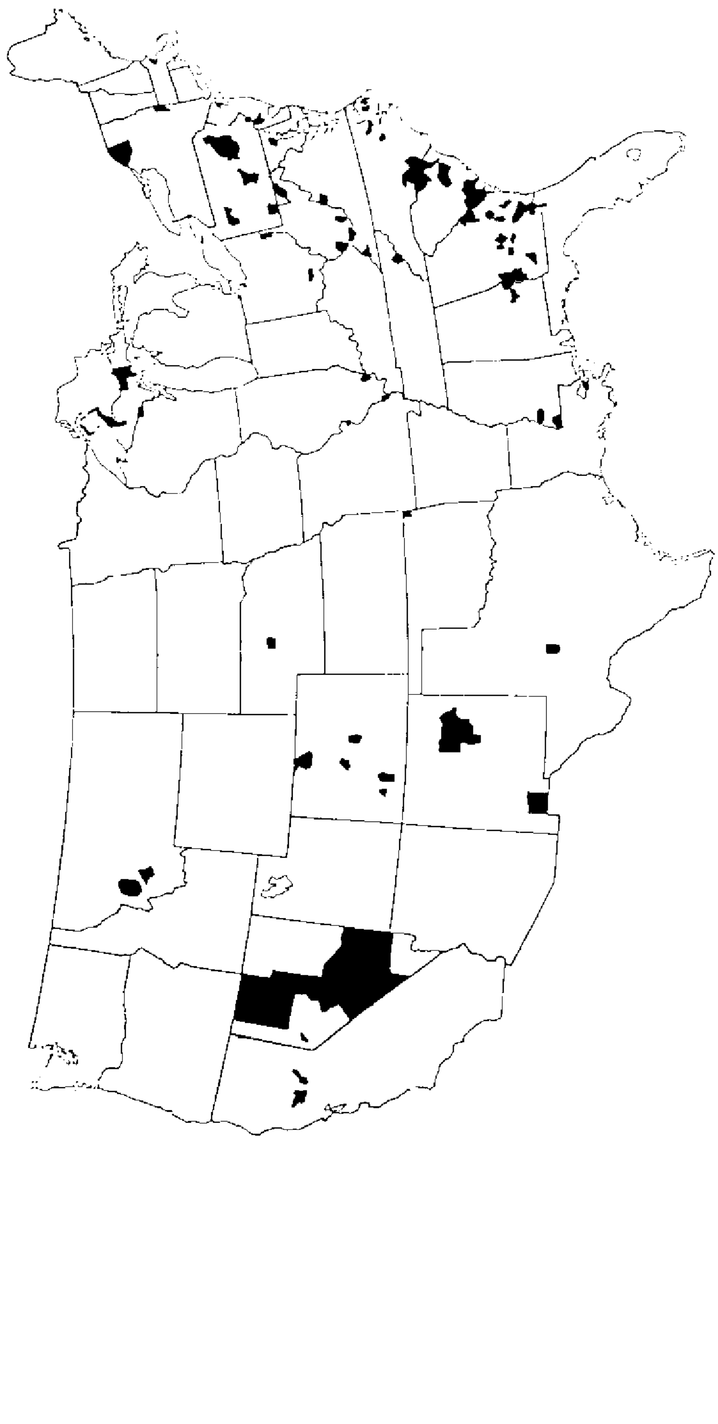


FIG.4. The 100 counties with the highest all-causes death rates for either sex, whites, aged 35 to 74, 1959 to 1969.

rates for males, but in the period 1973 to 1976 there were still five counties with male rates in the highest 3%.

Natural causes, or all diseases combined, show a stronger association with mining than do any of the major specific causes. However, the cardiovascular diseases show a stronger association than do malignant neoplasms.

Further study is needed to determine the relevant causal factors producing this association, such as chemical content of the drinking water, possible air pollution, inhalation and ingestion of specified particulates, among others.

3.2. Atomic energy

The residents of counties surrounding the Savannah River Plant (SRP) have been exposed to whatever radiation may emanate from a nuclear plant since the beginning of 1955 – prior to the operation of any nuclear power-generating plant in the USA (Fig.5). It is our understanding that the radiation from this plant is similar to that from a nuclear electricity-generating plant and provides an increase of only a very small percentage of the background radiation in this area. Our approach is to measure the actual risk of dying for the usual residents of this area, compared with the corresponding death rates for those living in Georgia and South Carolina,50 to 99 miles and further from SRP, and with US rates. Thus, our method of measurement is independent of measures monitoring radiation.

3.2.1. Breast cancer

Female breast tissue has been described by Upton as probably the human tissue most sensitive to radiation (Fig.6) [15]. Yet for metropolitan Augusta, consisting of Richmond County, Georgia, and Aiken County, South Carolina, the breast cancer rates are clearly lower than the US rates. The contrast is even greater for non-metropolitan counties within 50 miles of SRP. The Augusta area and the nearby non-metropolitan counties each have slightly lower rates than do the corresponding areas 50 to 99 and 100 to 149 miles from SRP, but this difference is too slight to be statistically significant.

3.2.2. Respiratory cancer

Death rates for cancer of the respiratory system for white males have been rising more rapidly and are higher in the south-east than in the USA as a whole. That is, the rates for Macon and Columbus (each more than 100 miles from SRP) and for Augusta have been rising rapidly. The high rates for individual non-metropolitan counties also presents a pattern of independence of distance from SRP [6].

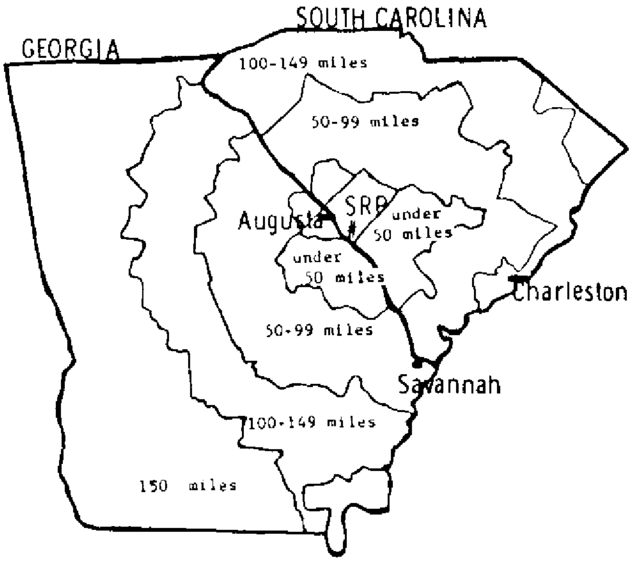
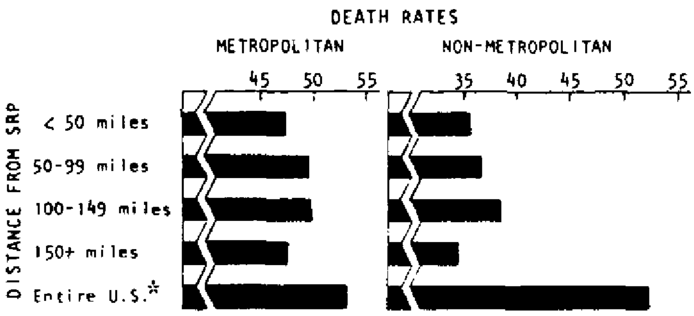


FIG. 5. Environs of the Savannah River Plant (SRP).



* Metropolitan and non-metropolitan combined.

FIG. 6. Breast cancer death rates for white females aged 35 to 74 by distance from Savannah River Plant (SRP).

TABLE I. STANDARD MORTALITY RATIOS, NON-RESPIRATORY CANCER, WHITES, AGE 35–74
(US rate for each group = 100.0)

Area and sex	1959–61	1968–72	1973–76
Metropolitan Augusta			
Male	101.5	95.2	101.6
Female	85.8	100.7	101.0
Non-metropolitan, within 50 miles of Savannah River Plant			
Male	83.6	92.9	96.3
Female	81.3	81.5	82.0

For blacks and for white females in non-metropolitan counties near SRP, the respiratory cancer rates are rather consistently *less* than the US rates.

The pattern of respiratory cancer rates in relation to SRP is thoroughly confusing to us. We have controlled, in appropriate demographic-epidemiological ways, for age, sex and race. We do not at present have data on cigarette-smoking patterns by age, sex, race and county. In view of the strength of cigarette-smoking as a risk factor for respiratory cancer, shown in many hundreds of studies, it appears important to us to control for this factor also before endeavouring to measure the impact of other factors, such as radiation.

3.2.3. Non-respiratory cancer

Death rates for all forms and sites of malignant neoplasms except respiratory (or non-respiratory cancers as a group) have rates for different time periods similar to or less than the US rates, as shown by the standard mortality ratios in Table I, with the US rates in each instance given a value of 100. Metropolitan areas generally have rates 4% or 5% higher than US rates [9]. Thus, the metropolitan Augusta rates in each instance are slightly lower than would be expected for a metropolitan area.

Rates for leukaemia, thyroid and bone cancer, as well as for other sites, have been analysed separately for three time periods [16]. Because of the small numbers of deaths and the resulting high standard errors, it has thus far been impossible to establish a clear relationship of either decreased or increased risk for specific sites of cancer for persons living near SRP.

TABLE II. STANDARD MORTALITY RATIOS FOR CONGENITAL ANOMALIES
(US rate for each group = 100.0)

Area, race and sex	1959 61	1968 72	1973 76
Metropolitan Augusta			
White male	60.0	82.1	153.6
White female	106.9	114.7	98.3
Black male	46.2	146.4	40.0
Black female	102.7	111.3	96.8
Non-metropolitan, within 50 miles of Savannah River Plant			
White male	116.4	100.0	87.0
White female	101.1	101.5	91.7
Black male	51.6	95.2	114.7
Black female	50.0	81.7	146.0

3.2.4. Congenital anomalies

Rates of congenital anomalies or birth defects are summarized by the standard mortality ratios in Table II, as was done for non-respiratory cancers. The number of such ratios under 100 is exactly equal to the number of ratios over 100. Even the highest ratio for white males for the Augusta area, 1973 to 1976, more than 50% in excess of what was expected, is not quite statistically significant ($p > 0.05$). And for the same area, same time period, black males have a ratio of only 40% or 60% below what was expected. Analyses of these rates for areas near SRP with corresponding areas in Georgia and South Carolina further from SRP do not present a clear picture of either decreased or increased risk.

Analyses of rates for the category 'other infant mortality' present an equally confusing picture.

3.3. Elevation

Elevation above sea level (see Table III) has consistently shown a substantial negative correlation with the risk of dying — the higher the elevation, the lower the risk.

TABLE III. CORRELATIONS OF SOCIAL INDICATORS AND DEATH RATES, WHITE MALES, AGE 35 TO 74, 1968 TO 1972

	Natural causes 000–796	Cardiovascular-renal 390–458 592 594	Stroke 430–438	Cancer 140 209
Education, median (n = 617 co.)	−0.28	−0.28	−0.25	
Income, median (n = 617 co.)	−0.28	−0.26	−0.25	
Dependent children* (n = 617 co.)	+0.27	+0.16		+0.14
Population density (n = 508 SEA s)	+0.20	+0.12		+0.28
Elevation (n = 508 SEA s)	−0.39	−0.43	−0.32	−0.55

* AFDC (aid to families with dependent children per 1000 population).

In the map previously shown for 1949 to 1951 death rates (Fig.3), the ten lowest-rate areas have elevations higher than the highest-rate areas with one exception: the coal-mining areas of east central Pennsylvania, with high death rates, are at an elevation approximately equal to the lowest elevations of the low-rate areas. The lowest-rate areas for the most part may be described as 'high-plains' areas, generally at elevations of 300–2000 m. The lowest-rate areas in Colorado and some of those in Utah are not in the mountains, even though we think of those states as being in the Rocky Mountains.

We have not yet clearly identified the factors associated with elevation that are responsible for this pattern of lower risk. The Canadian provinces of Saskatchewan, Alberta and Manitoba have the lowest rates in Canada – thus continuing the pattern seen in the USA. Various geochemical, hydrological and atmospheric science hypotheses show a degree of promise, but for the variables for which we have so far been able to obtain data, the correlations with the risk of dying are lower than just elevation with death rates. European death rates do not show such a clear pattern of association of low rates with high elevations. For example, the Netherlands generally have extremely low rates for middle-aged males, for cardiovascular diseases and for all diseases combined [8].

At higher elevations, including the areas with low white male death rates, the background radiation is several times as great as the background radiation

TABLE IV. CORRELATIONS OF RATES, HEALTH RESOURCES AND DEATHS, WHITE MALES, AGE 35 TO 74, 508 SEAs, 1968 TO 1972

	Natural	Cardiovascular-renal	Stroke	Cancer
Physicians (all)			−0.20	+0.31
General practitioners	−0.36	−0.29	−0.24	−0.30
Days hospitalized	+0.17			+0.22
Nurses			−0.22	+0.24
Dentists	−0.24	−0.32	−0.40	

in the south-east coastal plains, the large area with very high rates [17]. This association has been recognized to exist for cancer, but it may be stronger for the cardiovascular diseases [18, 19]. It is obviously desirable for us to obtain satisfactory estimates of background radiation by state economic areas in order to test more thoroughly the hypothesis that hormetic doses of radiation are associated with lower risk of death due to the cardiovascular diseases and other chronic diseases.

3.4. Socio-economic variables

Many socio-economic or cultural variables, along with the chemical content of the drinking water, atmospheric science and geochemical variables, present us with an abundance of moderate to low-level correlations which are statistically significant ($p < 0.0001$), primarily because our analyses are based upon such a large number of areas, in most instances 500 or more (Table III).

Median income presents a low, negative correlation with white male death rates – the higher the income, the lower the death rate. But questions may be raised, such as: What is associated with higher income? If the higher income is used to purchase cigarettes, to eat an excessive number of calories and saturated fats, and to refrain from exercise, is it reasonable to expect even the low negative correlations shown?

With increased population density, there is a slight tendency toward higher death rates, for natural causes and for cancer, being statistically significant ($p < 0.0001$). Yet, to what extent is population density a socio-economic variable and to what extent is it an index of air pollution and other physical science variables?

Patient-care resources present some puzzling problems in analysis (Table IV). Cancer death rates show a positive association with physicians (including specialists) per thousand population, but a negative association with general

TABLE V. CORRELATION, 37 INDEPENDENT VARIABLES WITH WHITE FEMALES, AGE 35 TO 74

Cardiovascular-renal diseases	Multiple R
Per cent not moving last five years	0.62
Per cent professional, managerial or farming	0.70
Median family income	0.77
Per cent 'walked to work'	0.81
Per cent manufacturing, mining	0.84
Median education	0.86

practitioners per 1000 population. Other correlations suggest that this puzzling association is primarily due to the tendency for metropolitan areas to have both higher cancer death rates and higher rates of physicians (including specialists) per 1000 population [9].

4. ANALYSIS OF VARIABLES

More work is needed in the application of multiple regression and other techniques, but this implies:

(a) Developing indices that actually measure the variable we wish to measure, for various areas;
(b) Collecting data from many sources for the same geographic areas;
(c) Recognizing the interrelationships or confounding effect of many variables;
(d) Developing models that are compatible with existing data, rather than depending solely upon the general model built into a computer program.

For the 92 large metropolitan areas, we have been able to collect data on more than 100 variables, from which we selected the 37 with the highest correlations: 10 drinking-water variables; 10 weather or atmospheric science; 2 geographic; and 15 socio-economic or cultural variables, which served as the independent variables, and the cardiovascular-renal diseases death rates for white females as the dependent variable (Table V). The first six variables picked by the program as contributing most to the correlation were all socio-economic. None was either drinking-water or weather variable. Yet these six variables had a correlation of 0.86, and R^2 is 0.74, indicating that 74% of the differences in white female rates from area to area are associated with these six variables. Parallel correlations for white males produced similar but slightly less dramatic results.

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DISCUSSION

B. SØRENSEN: Would it not have been better to discuss coal-mining regions and uranium-mining regions (and milling sites) rather than compare coal-mining with uranium reactor operation?

H.I. SAUER: In our sample of 300 counties, the geologist classified only four as being the scene of heavy or moderate uranium mining, two of which were among the 100 lowest-death-rate counties and two among the 100 near-median-death-rate counties; none was among the 100 highest-death-rate counties. He also reported a small amount of uranium mining, which he considered unlikely to have an effect upon the health of the residents, for eleven additional counties; of these eleven, four were lowest-rate counties, two near-median-rate counties, and five highest-rate counties. However, all five of these highest-rate counties also had heavy or moderate mining of several other metals or of coal.

Thus, at present, neither uranium-mining nor reactors appear to be associated with high-death-rate counties.

DATA RESOURCES FOR ASSESSING REGIONAL IMPACTS OF ENERGY FACILITIES ON HEALTH AND THE ENVIRONMENT*

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Abstract

DATA RESOURCES FOR ASSESSING REGIONAL IMPACTS OF ENERGY FACILITIES ON HEALTH AND THE ENVIRONMENT.

Atmospheric emissions from fossil-fuel power plants and other sources continue to cause concern about impacts of these pollutants on human health and the environment. Assessing these impacts requires a regional-scale approach that integrates spatial and temporal patterns of emissions, environmental factors and human populations. Two examples of regional studies are presented, including a comparison of patterns of coal-fired power plants and selected diseases and identification of areas sensitive to acid rain which may transfer acid and toxic metals to aquatic systems and man. Energy, socio-economic, health and environmental data are often collected and summarized for counties in the USA. Counties are well-defined geopolitical units which can be used to integrate data, to aggregate data into larger regional units, and to display data as thematic maps. However, researchers are too frequently faced with the tedious task of assembling and reformatting files from several data-collection agencies prior to conducting regional studies. Systems such as UPGRADE, DIDS, SEEDIS and Geoecology have standardized many files into integrated data bases which utilize counties as the primary spatial unit. These systems are compared and data resources discussed.

1. INTRODUCTION

Energy production requires governments to make choices as to the development of alternative resources and technologies. Scientific and public concerns about large-scale, non-site-specific effects of energy production on human health and environmental quality have resulted in regional assessments of potential energy-related impacts. Such integrated assessment and planning requires the availability of data on energy production and emissions, environmental resources,

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human population, health, and other factors in compatible formats and spatial scales for large geographic areas. This paper discusses the role of regional studies and available data resources for addressing energy-related impacts on human health and the environment.

2. REGIONAL STUDIES

Over the past decade, there have been an increasing number of studies in which several variables are integrated over large geographic areas. Contributing to the increased number of regional efforts has been the increased awareness by the scientific community and funding agencies that some problems, such as acid rain, need to be addressed at a regional scale.

Computers have also been a key factor in facilitating regional studies. Data management systems allow large files to be stored and easily merged for analysis. Statistical packages, such as the Statistical Analysis system [1], also are readily available to perform necessary analyses. Computer-generated maps are easy to produce without extensive cartographic training. Currently, computers can produce high-quality, color-thematic maps quickly and at low cost. Collections of thematic maps published at identical scales, such as the atlas of cancer mortality [2], have helped stimulate integrated studies to explain spatial patterns in thematic maps.

A regional approach is useful for planning, impact assessment, and hypothesis generation. Regional planning studies may involve identifying candidate sites based on many factors. Many sites can be quickly screened so that the site-specific selection process is significantly reduced. Impact assessment at the regional scale can incorporate the long-range transport and cumulative aspects of energy-related emissions. Such studies can be used to identify sensitive areas or populations at risk, and, as dose-response relationships are established, estimates of potential impacts can be made. Comparing spatial patterns by overlaying maps often suggests relationships between various factors. The results of such a study can be used to design more detailed studies to test these hypotheses. Maps generated for regional analysis are also extremely valuable in editing spatial data.

Hypothesis-testing and epidemiologic investigations cannot be adequately addressed by currently available county-level data. In general, regional data bases have not been designed for epidemiological studies. They do not contain individual exposure data or time series data adequate for diseases with long latency periods [3].

3. INTEGRATED SYSTEMS

Integrated systems have been developed which combine data from several thematic areas for regional analysis [4]. Systems include UPGRADE (User Prompted Graphic Data Evaluation), Council on Environmental Quality (CEQ); SEEDIS (Socio-Economic, Environmental, Demographic Information System), Lawrence Berkeley Laboratory (LBL); DIDS (Decision Information Data System), U.S. Department of Commerce; and Geoecology, ORNL. Other less formalized systems exist at Argonne National Laboratory (ANL) and Brookhaven National Laboratory (BNL). All of these systems have county-level, integrated data bases; national coverage; energy, environmental and health data; computer graphics, including mapping; and statistical analysis capabilities. This section compares the general capabilities of the systems and describes the Geoecology system in more detail. Sources of thematic data and county cells, which are key features of the integrated systems, are also discussed.

3.1. System Comparisons

Each of the systems has its own strengths and special features. Geoecology emphasizes developing county-level environmental files while utilizing readily available software. UPGRADE, DIDS and SEEDIS have all developed extensive software. UPGRADE has user-oriented interactive retrieval and display capability; DIDS has quick-response interactive color-mapping capabilities; and SEEDIS has extensive spatial-analysis capabilities. SEEDIS can aggregate or disaggregate data at one spatial scale to relate to data at other scales. Although BNL and ANL do not have as extensive formal systems, they have data collections that are routinely integrated, analyzed and displayed.

Although these systems share many common data sets and capabilities, each system has evolved with slightly different orientations. The national laboratories (ANL, BNL, LBL and ORNL) have developed resources as required to perform research funded by the Department of Energy, and generally have limited capabilities to provide support to other groups. UPGRADE primarily provides support for producing the CEQ annual report. It has been used to study relationships between air and water factors and human health. Other federal agencies are using UPGRADE, and the system is accessible commercially from Sigma Data Computing Corp. DIDS is oriented to decision makers in the federal government and currently contains demographic and socioeconomic data. The Geoecology Data Base contains primarily environmental data, although demographic, socioeconomic, energy and health data are available. The SEEDIS data base contains demographic, socioeconomic, health and air-quality data. Both UPGRADE [5, 6, 7] and Geoecology [8] have recently published system documentation.

3.2. Geoecology Data Base

As discussed above, the Geoecology Data Base [8] is typical of the integrated systems having an extensive data base for regional studies. A standard set of 3071 county units for the conterminous United States is used with some data available for subcounty units within larger, more diverse, eastern counties. The Geoecology Data Base contains selected data on terrain and soils, water resources, forestry, vegetation, agriculture, land use, wildlife, air quality, climate, natural areas and endangered species. Additional files on human population, health, energy and socioeconomic factors are also included. Data are stored in metric SI units.

The data base can be accessed in either batch or interactive modes utilizing the ORNL/IBM computer system. The Statistical Analysis System [1] is used for data management, retrieval, analysis and display, including both x-y plots and maps. Maps are also generated from the data base using independent computer-mapping programs. Currently the Data Base consists of approximately 125 SAS data sets containing over 2000 data elements and uses 40 million bytes of online storage. Copies of the Geoecology Data Base have been distributed on magnetic tape either as SAS format files or as EBCDIC files. The Geoecology Data Base became operational in 1975 with primary funding from the Office of Environmental Assessments, U.S. Department of Energy.

3.3. Spatial Cells

In general, the most commonly used spatial cell in the United States for health, socioeconomic, environmental and energy data is the county. Counties can be used to integrate data, to aggregate data into larger regions, and to display data as thematic maps. There are approximately 3100 counties depending on how independent cities, Alaska, and Hawaii are treated. Counties average 250 000 km^2. Some advantages of county units are: well-established and relatively permanent geopolitical units; common unit for state and national data collection; relatively uniform in size in the east; often used as management units for policy; and county outlines available in computerized form. Disadvantages include: names or boundaries change occasionally; different county identifier codes are used by various agencies; large and heterogeneous counties do occur; and independent cities representing urbanized administrative areas are not treated uniformly.

3.4. Data Resources

Many federal agencies collect county-level data to fulfill their specified missions. Inventories of these spatial data files, including data set descriptions and contact names, are available from many agencies [see 9, 10, 11, 12]. In addition, the proceedings of the 1980 Integrated County-level-data User's Workshop [4] contain an inventory of county-level data.

UPGRADE, DIDS, SEEDIS and Geoecology are routinely expanding their data bases by rigorously selecting and screening data from larger data bases (e.g. the EPA STORET water-quality data base) to create standardized files. Data extraction, editing and reformatting are tedious tasks. SEEDIS has created a very useful air-quality data base from EPA/SAROAD data, while BNL has developed emissions data from the EPA/NEDS file. UPGRADE has extracted several water-quality data sets from STORET files. Geoecology has estimated county-level climatic parameters from the NOAA weather-station files.

There is considerable interaction between the integrated systems groups. Copies of many of the Geoecology files have been distributed to the other national laboratories and to UPGRADE. ORNL has received data files from ANL, BNL, LBL and UPGRADE which have been incorporated into the Geoecology Data Base.

4. ASSESSMENT OF ATMOSPHERIC POLLUTANTS

Currently, one of the most popular topics for regional analysis is assessing impacts of acid rain on human health and the environment. Acid rain is the product of many sources, both natural and anthropogenic, with the precursors often being transported over state and national boundaries. Coal-fired power plants emit major amounts of sulfur dioxide and particulates, while other industries and automobiles contribute significant amounts of these plus nitrogen oxides and hydrocarbons. Regional studies of "acid rain" generally consider impacts of these pollutants along with acid deposition, either in dry or wet forms. Because acid rain is related to energy production and covers large geographic areas, several studies at ORNL have used a regional approach to assess potential impacts of these atmospheric pollutants.

4.1. Sulfur Dioxide and Particulates

A study of energy development on rural health in the southeast was conducted by Parzyck and others [13] of the ORNL Health and Safety Research Division. This study considered the location of energy facilities in the southeast and ambient levels of sulfur dioxide and particulates. The study integrated data on energy-facility siting, air quality, population and mortality with a series of computer-generated maps as the primary output. Sulfur dioxide levels were generally higher in states where the distribution of coal-fired power plants was localized, whereas total suspended particulates were not correlated

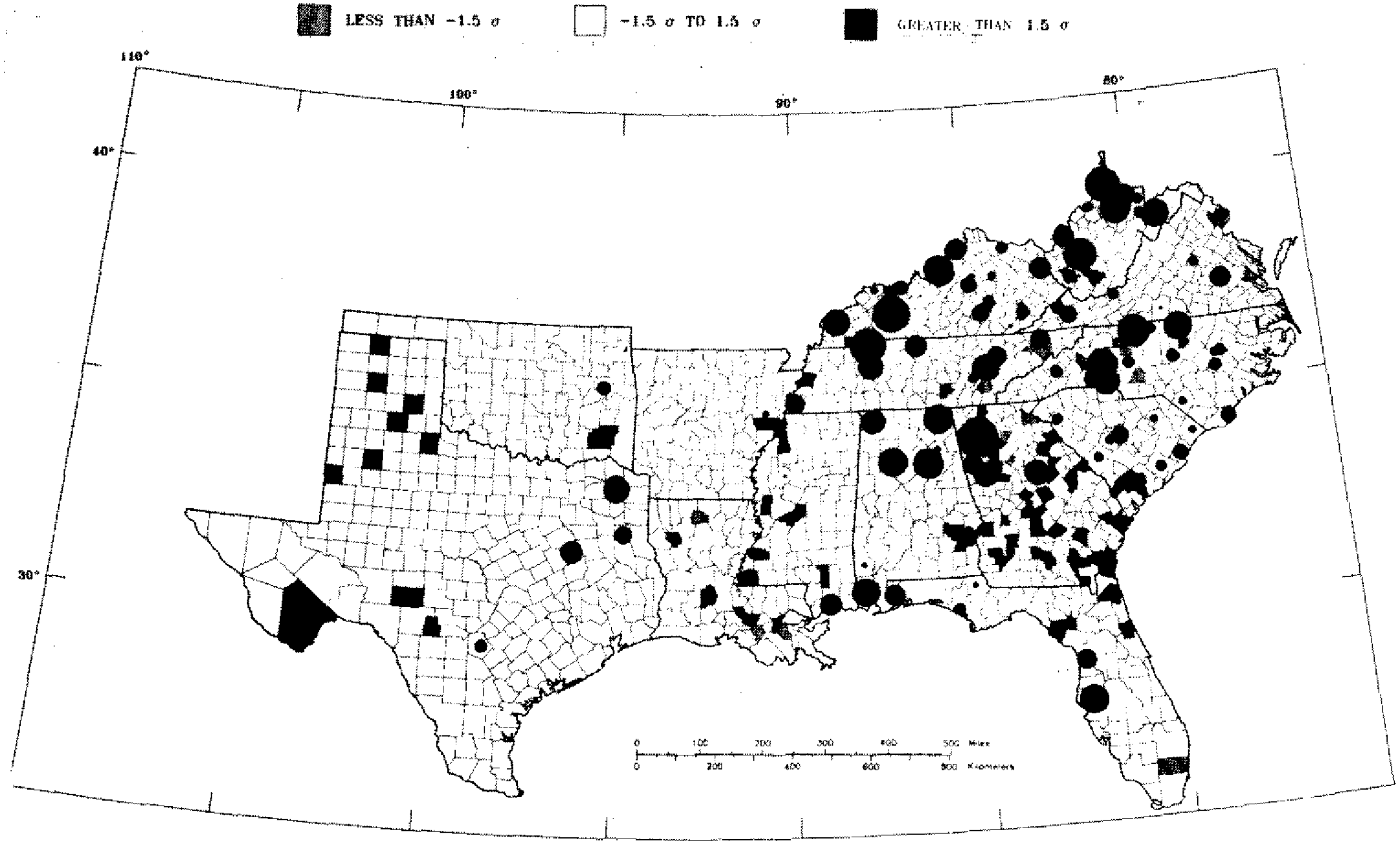

FIG. 1. County-level chronic respiratory disease rates adjusted for population density as standard deviations from the mean of each density class. Locations of coal-fired power plants represented by circles proportional to the generating capacity of the plant.

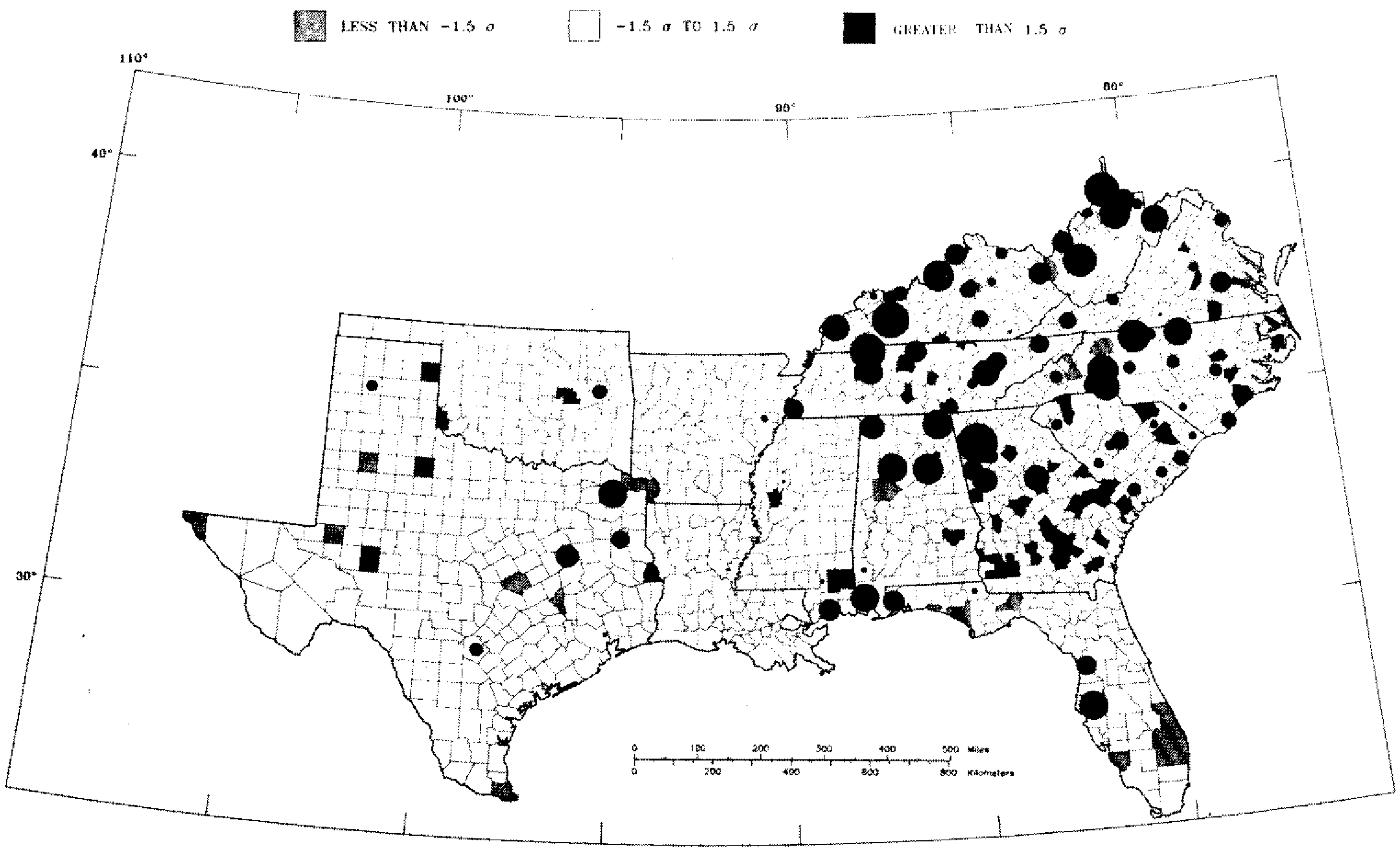

FIG.2. County-level acute ischemic heart disease mortality rates adjusted for population density as standard deviations from the mean of each density class. Locations of coal-fired power plants represented by circles proportional to the generating capacity of the plant.

well with energy facilities. Mortality rates for chronic respiratory disease and acute ischemic heart disease were mapped relative to the location of coal-fired power plants (Figs 1 and 2). Mortality rates were expressed in standard deviations calculated by combining counties of similar population density.

In general, no strong relationship between either chronic respiratory disease or acute ischemic heart disease and coal-fired power plant locations were found [13]. As the authors point out, these results do not eliminate the possibility of such a relationship but represent a first step in examining potential energy-related health impacts.

4.2. Acid Rain

One of the indirect impacts of acid rain on human health relates to the release and transfer of toxic metals from soils and bedrock into watersheds and the subsequent transfer of these metals into drinking water. Toxic metals include aluminum, cadmium, copper, lead and mercury. Human exposures to these metals can result from consuming contaminated water or fish.

A regional approach was used to identify sensitive areas in which soil and bedrock characteristics are such that acid inputs may be transferred to aquatic systems [14]. Sensitive areas were identified from an evaluation of five soil-chemistry parameters, bedrock sensitivity, land capability and land use. In addition to integrating both soil and bedrock factors, the analysis considered sulfate adsorption capacity of soils, which has been shown to prevent transfer of acid to aquatic systems [15]. The classification criteria were developed by the Canada/United States Transboundary Working Group [16]. The analysis utilized the Geoecology Data Base to integrate the data according to these criteria in a timely manner to be used by ongoing scientific-political deliberations.

In the eastern 27 states, 13% of the counties had moderate and 4% had high sensitivities (low buffering capacity) to acid rain (Fig. 3). The map includes previously identified areas in the northeast and upper midwest but did not include other areas found on other single-factor sensitivity maps. The next steps in the analysis will validate the map and incorporate current and projected atmospheric loadings to identify sensitive areas that are, or will be, exposed to high levels of acid deposition.

5. SUMMARY

Assessing regional-scale energy-related impacts on human health and the environment can be addressed utilizing available integrated data bases with computer-oriented analysis and display techniques.

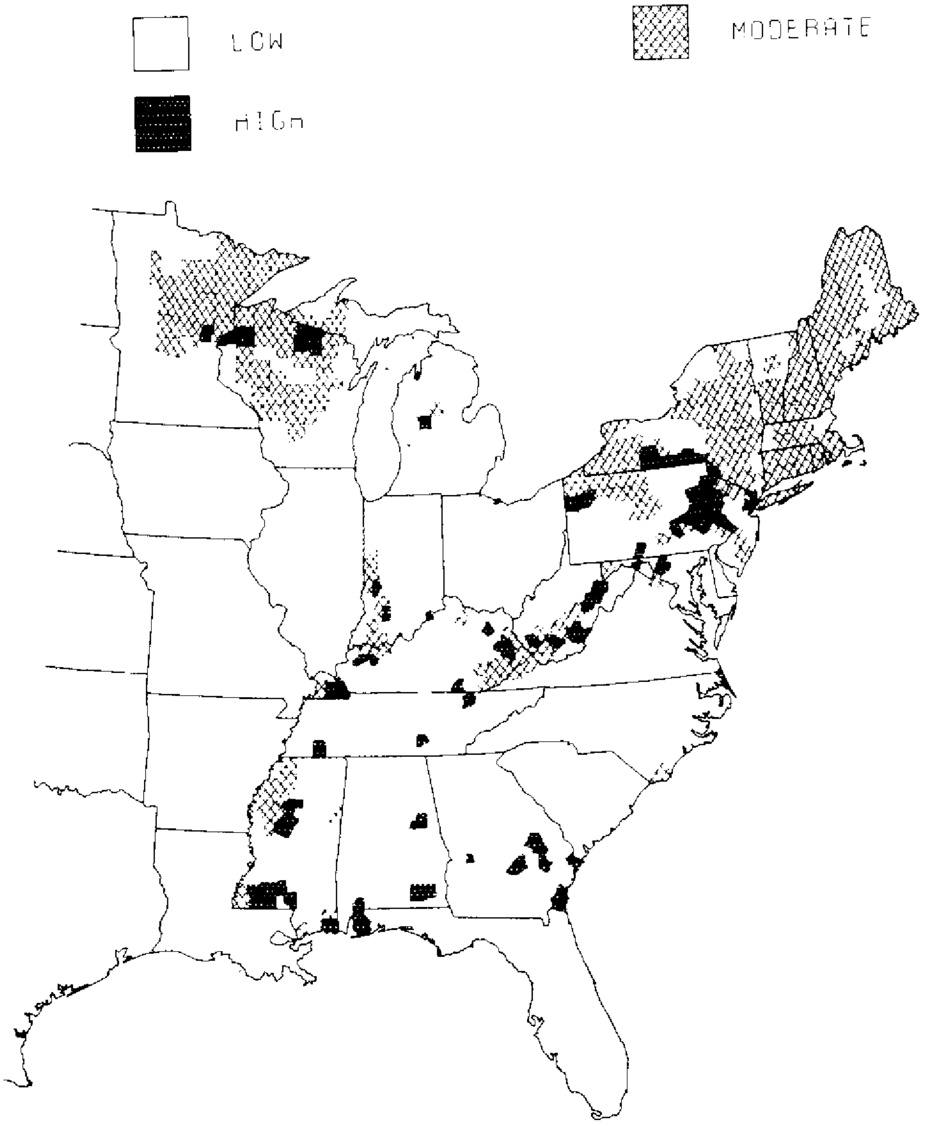

FIG.3. Soil sensitivity to acid rain with high and moderate potential to transfer acid to aquatic systems. Sensitivity ratings are based on exchangeable base content, sulphate adsorption capacity, bedrock sensitivity and land capability (surrogate for soil depth).

Regional studies often generate hypotheses and can be used to design more detailed studies. Regional, county-level data bases are available from integrated systems such as UPGRADE, SEEDIS or Geoecology. These systems have extracted and edited data from source files for integration and analysis. Data represent county aggregates or averages and, as such, are limited in use for epidemiological studies.

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DISCUSSION

R.M. BARKHUDAROV: As you said, your system assesses the effects of heavy metals. What effects do you have in mind and do they include carcinogenic ones?

R.J. OLSON: It has been suggested that acid rain is associated with the leaching of heavy metals from soil and bedrock. Our study did not investigate the effects of these metals but identified areas in which acid rain leaching may result in higher levels of these metals in streams and drinking water.

USE OF A MORTALITY-RATIO MATRIX AS A HEALTH INDEX

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Abstract

USE OF A MORTALITY-RATIO MATRIX AS A HEALTH INDEX.

There are no reliable data on population size and age distribution for small geographical areas such as counties, cities and populations around power plants. This problem is especially acute during intercensal periods. The health effects of energy use has usually been assessed by studying changes in mortality rates of exposed populations. Estimation of such rates depends on death certificates and census data. Because of the uncertainties in projections from census data for small geographical areas, the corresponding mortality rates are likewise uncertain. As an alternative health index, the authors propose the use of a mortality-ratio matrix, comprising a cross-tabulation of deaths by age and by selected disease categories, normalized for each age. The set of shapes of the distributions of the age-specific mortality fractions for different causes could be useful in the study of the comparative health of different populations. The authors use fractions rather than rates because of the above uncertainties in enumeration data. Age- and sex-specific mortality fractions have been computed for several groups of causes for selected countries and times. To quantify differences in the shapes of the distributions, standard descriptive statistical parameters were calculated. For some causes, changes in shape of the distributions are expected to reflect the combined effect of changes in intensity and duration of exposure to the causal agent and variations in the susceptibility of the population. Assuming that the number of deaths is sufficient for reliable estimation of the ratios, such changes in shape will be independent of age-distribution and size of population at risk. To apply this method to the population at risk in a specific area, e.g. the population around a power plant, it is important that the enumeration of deaths be complete and accurate. For certain chronic causes, selective migration may be a source of error.

1. INTRODUCTION

A census is taken every ten years. Between enumerations, the population size is based on estimates. These techniques are unreliable for small geographical aggregates owing to population mobility. Even during census years the undercount presents problems in small geographical areas for particular ethnic groups. In contrast, death data is recorded on a day-to-day basis, and the undercount is negligible.

It has been usual to assess the health effects of energy use by studying changes in mortality rates of exposed populations. Estimation of such rates depends on census data or estimates and on death certificates. Because of the uncertainties in estimates from census data for small geographical areas, the corresponding mortality rates are likewise uncertain. One way to avoid such uncertainties is to use cause-specific death ratios [1].

In previous work [2, 3] we have found that age is an important factor in assessing health effects of energy use. Age, sex and cause-specific mortality ratios show both beneficial and detrimental effects of industrial development and its associated increase in energy use. We propose to use a mortality ratio matrix, comprising a cross-tabulation of deaths by age and by selected disease categories, normalized for each age. Each element of this matrix will be referred to as the *relative importance* for a given disease, country, age and year.

The sex- and age-specific relative importance for different causes provides an alternative to mortality rates for the study of the comparative health of different populations. Age-specific ratios have the advantage of being independent of the age distribution of the population, which in small geographical areas is difficult to estimate.

2. METHODS

For three selected countries (USA, Japan and Chile) data on mortality from *Causes of Death* by S. Preston et al. [4] was used. These countries were chosen because they have very different histories of development but each one has data covering the period 1900 to 1964. Assuming that accidental deaths are independent of the prevalence of any disease, we have restricted our study to the ten non-accidental categories in Preston's data.

For each sex, year and country, a cross-tabulation was made between 19 age intervals and 10 non-accidental causes of death, each cell of which contains the value $P_{a,d}$. This is the percentage of non-accidental deaths in the age interval a attributable to cause d. We have kept the same age intervals and cause categories as in the original data. The set of 19 values for a given cause defines a distribution which describes the relative importance of that cause as a function of age. For each such distribution we have calculated the following descriptive statistics: the maximum per cent, the average per cent over the 0-to-90-year age span, the mean age, the standard deviation, the skewness and the kurtosis.

This set of values serves as a mathematical description of the probability distribution of the relative importance, for any given cause of death, as a function of age by sex, year and country. In particular, the average per cent measures the over-all level of relative importance of the cause; the skewness measures which side of the distribution is more gradual; and the kurtosis measures whether the distribution is more peaked or less so than the normal distribution.

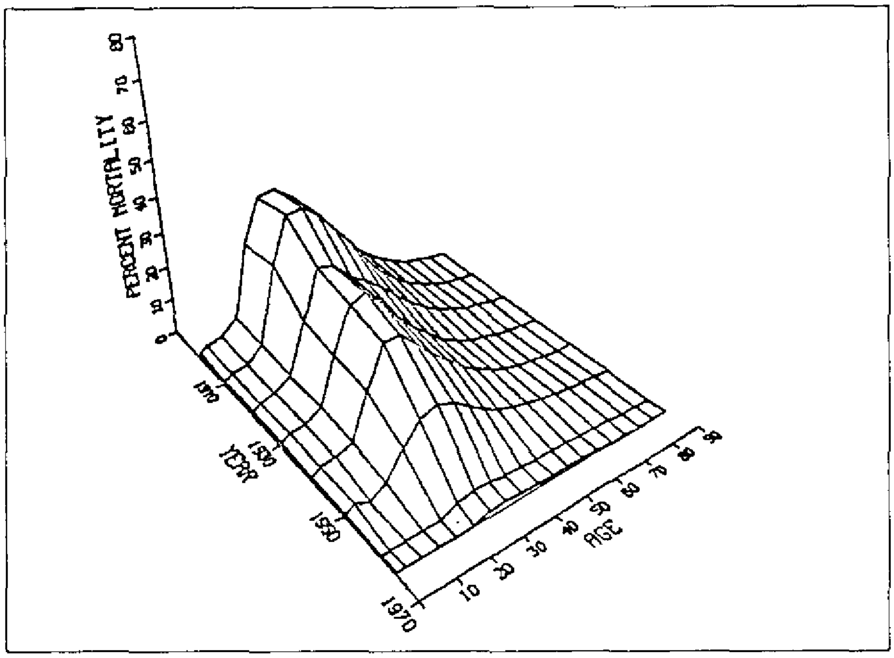


FIG.1. Plot of surface representing evolution of relative importance of respiratory TBC, 1900- 1964, for US males, age 0 to 90.

3. RESULTS

The complete set of percentages and parameters for Preston's ten disease categories as well as those for accidental deaths, by country and year, for the USA, Japan and Chile, from 1900 to 1964, for both sexes, and for 19 age intervals extending from 0 to 90 years may be found in Ref. [4]. Space does not permit reproduction in these Proceedings.

We have selected for further analysis four important disease categories:

(1) Respiratory tuberculosis (TBC)
(2) Other infectious and parasitic diseases
(3) Malignant and benign neoplasms
(4) Cardiovascular diseases

These categories were chosen because of their importance as causes of death and their strikingly different historical trends. The relative importance of each of these categories as a function of age and time is shown in Figs 1–6, and the associated descriptive parameters are given in Tables I–IV.

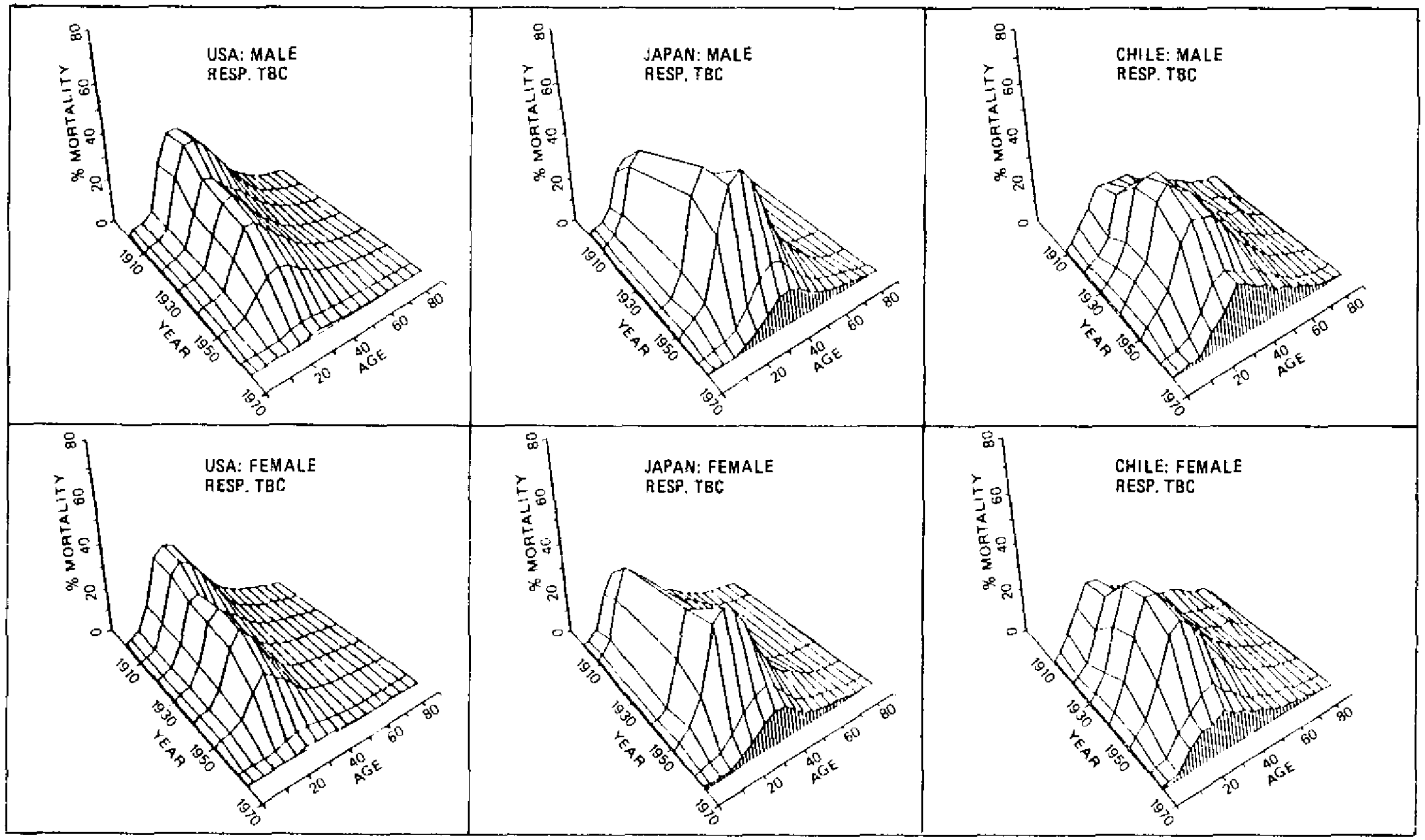


FIG.2. Plots of six surfaces representing evolution of the relative importance of respiratory TBC, 1900 1964, for USA, Japan and Chile, both sexes, age 0 to 90.

TABLE I. DESCRIPTIVE PARAMETERS FOR CATEGORY 1, RESPIRATORY TUBERCULOSIS

		USA		JAPAN		CHILE	
		MALE	FEMALE	MALE	FEMALE	MALE	FEMALE
1900	AVG%,0-90Y	15.39	14.22	10.36	9.70	--	--
	MEAN AGE	35.3	32.5	33.5	30.1	--	--
	S.D.	15.4	15.7	15.1	15.2	--	--
	SKEWNESS	.72	.91	.69	.82	--	--
	KURTOSIS	.31	.64	.11	.26	--	--
1910	AVG%,0-90Y	15.27	13.10	12.85	12.01	14.16	14.74
	MEAN AGE	34.7	31.7	32.4	29.3	39.1	36.7
	S.D.	14.7	14.8	14.8	14.7	20.7	20.5
	SKEWNESS	.71	.95	.73	.79	.45	.57
	KURTOSIS	.36	.86	.24	.20	-.68	-.58
1920	AVG%,0-90Y	11.80	10.23	--	--	13.88	14.64
	MEAN AGE	35.1	31.6	--	--	37.9	36.0
	S.D.	15.1	15.5	--	--	19.1	19.2
	SKEWNESS	.69	.96	--	--	.59	.72
	KURTOSIS	.25	.70	--	--	-.44	-.26
1930	AVG%,0-90Y	10.34	9.43	--	--	18.65	17.73
	MEAN AGE	33.6	29.8	--	--	35.9	33.3
	S.D.	14.4	14.6	--	--	17.9	18.1
	SKEWNESS	.80	1.16	--	--	.65	.81
	KURTOSIS	.69	1.53	--	--	-.21	.01
1940	AVG%,0-90Y	9.20	8.32	15.22	12.34	19.02	18.23
	MEAN AGE	33.4	28.7	30.2	28.0	34.2	31.2
	S.D.	14.4	13.7	12.6	13.0	16.8	16.9
	SKEWNESS	.79	1.27	.90	.97	.63	.78
	KURTOSIS	.67	2.19	.89	.84	-.04	.13
1950	AVG%,0-90Y	5.86	5.74	19.32	16.06	16.81	15.44
	MEAN AGE	35.1	28.5	33.5	30.3	34.6	31.3
	S.D.	15.4	13.6	13.8	13.7	16.7	16.2
	SKEWNESS	.81	1.22	.64	.77	.64	.87
	KURTOSIS	.54	2.41	.30	.45	.06	.49
1960	AVG%,0-90Y	1.14	.91	8.87	7.10	10.21	8.49
	MEAN AGE	44.0	38.0	39.8	36.0	38.2	33.6
	S.D.	17.0	14.1	15.2	14.4	16.4	16.1
	SKEWNESS	.69	1.39	.59	.69	.62	.84
	KURTOSIS	-.43	1.72	.21	.50	.07	.41
1964	AVG%,0-90Y	.68	.43	5.79	4.54	9.22	6.89
	MEAN AGE	48.3	43.5	43.8	39.9	39.8	34.8
	S.D.	16.7	15.3	15.8	14.0	16.3	17.0
	SKEWNESS	.58	1.06	.54	.91	.54	.75
	KURTOSIS	-.70	.39	.01	.58	-.15	.11

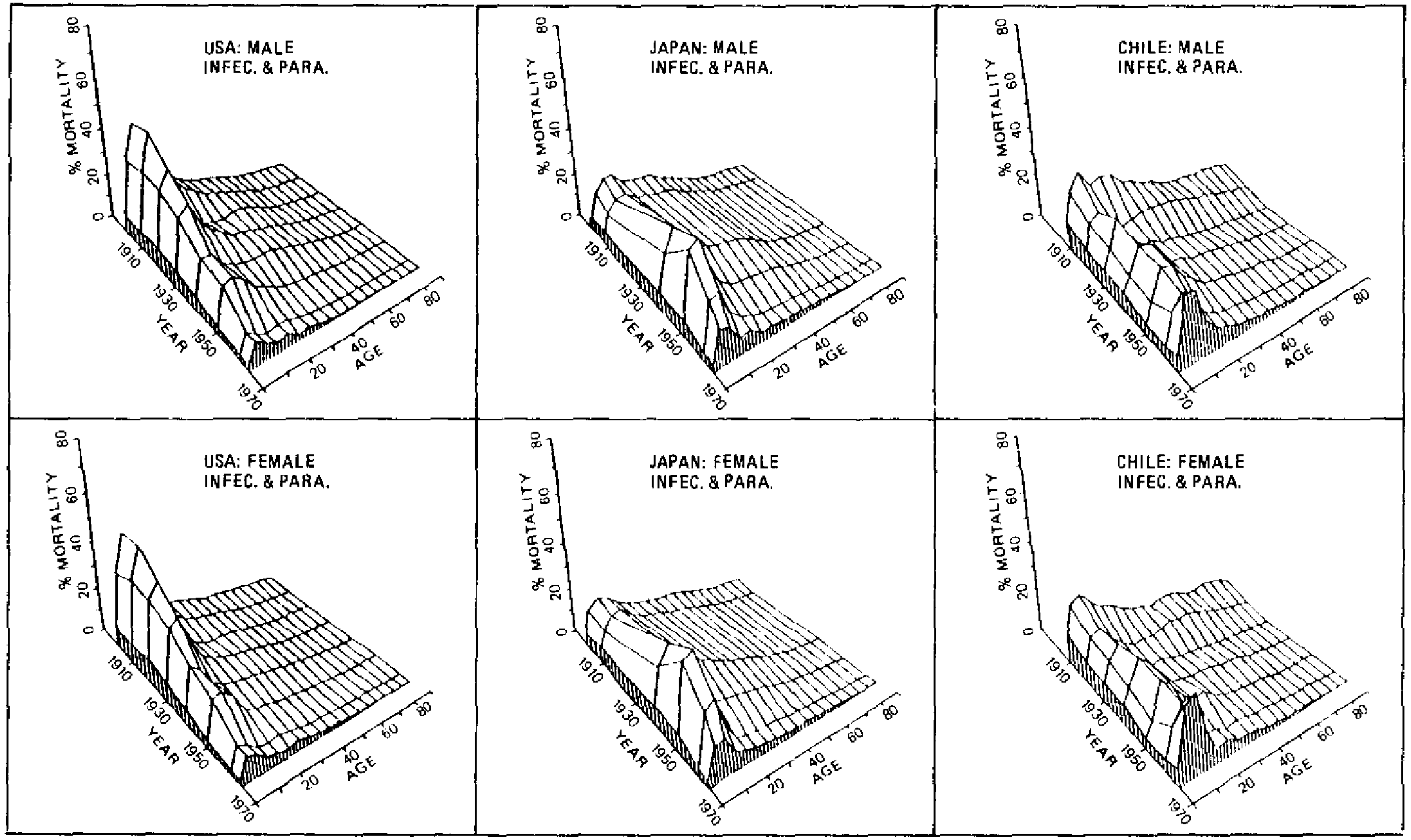


FIG.3. Plots of six surfaces representing evolution of relative importance of other infectious and parasitic diseases, 1900–1964, for USA, Japan and Chile, both sexes, age 0 to 90.

TABLE II. DESCRIPTIVE PARAMETERS FOR CATEGORY 2, OTHER INFECTIOUS AND PARASITIC DISEASES

		USA		JAPAN		CHILE	
		MALE	FEMALE	MALE	FEMALE	MALE	FEMALE
1900	AVG%,0-90Y	13.18	11.48	9.29	8.08	--	--
	MEAN AGE	25.1	23.5	27.0	28.5	--	--
	S.D.	21.0	21.5	18.9	20.8	--	--
	SKEWNESS	1.10	1.25	.86	.73	--	--
	KURTOSIS	.38	.65	.22	-.36	--	--
1910	AVG%,0-90Y	13.30	11.16	8.37	7.56	15.57	13.38
	MEAN AGE	23.8	21.2	27.1	26.6	34.5	35.4
	S.D.	19.7	19.8	18.3	18.9	24.5	25.8
	SKEWNESS	1.14	1.46	.77	.83	.58	.47
	KURTOSIS	.68	1.48	.12	.03	-.80	-1.04
1920	AVG%,0-90Y	10.75	8.89	--	--	11.18	9.31
	MEAN AGE	24.0	20.1	--	--	35.3	34.8
	S.D.	20.4	19.5	--	--	23.5	24.6
	SKEWNESS	.99	1.46	--	--	.39	.35
	KURTOSIS	.11	1.44	--	--	-.88	-1.04
1930	AVG%,0-90Y	10.25	8.14	--	--	7.12	6.05
	MEAN AGE	23.8	20.6	--	--	25.2	24.1
	S.D.	19.2	18.8	--	--	20.7	21.0
	SKEWNESS	1.01	1.37	--	--	1.01	1.10
	KURTOSIS	.36	1.35	--	--	.17	.31
1940	AVG%,0-90Y	7.51	5.80	10.98	10.72	7.80	6.88
	MEAN AGE	25.9	21.5	25.0	24.7	25.1	23.5
	S.D.	19.5	18.5	18.2	18.5	19.6	19.2
	SKEWNESS	.75	1.12	.87	.92	1.04	1.08
	KURTOSIS	-.22	.69	.19	.26	.36	.43
1950	AVG%,0-90Y	5.61	4.65	9.36	9.31	7.03	6.41
	MEAN AGE	20.2	17.7	21.8	21.5	23.2	22.0
	S.D.	17.4	16.3	18.8	18.9	18.2	17.3
	SKEWNESS	1.44	1.65	1.08	1.15	1.23	1.35
	KURTOSIS	1.69	2.68	.43	.62	.96	1.53
1960	AVG%,0-90Y	2.28	2.26	5.44	5.41	5.36	5.00
	MEAN AGE	21.4	20.5	19.9	18.3	21.0	19.4
	S.D.	18.9	18.3	18.4	18.1	19.4	18.2
	SKEWNESS	1.35	1.35	1.35	1.55	1.49	1.50
	KURTOSIS	1.34	1.43	1.22	1.81	1.65	1.74
1964	AVG%,0-90Y	2.20	2.27	3.60	3.66	5.52	5.80
	MEAN AGE	20.0	20.2	21.8	21.1	15.8	14.4
	S.D.	19.1	18.6	19.7	19.8	16.3	14.6
	SKEWNESS	1.55	1.38	1.16	1.26	1.96	2.09
	KURTOSIS	1.92	1.46	.54	.74	3.92	4.82

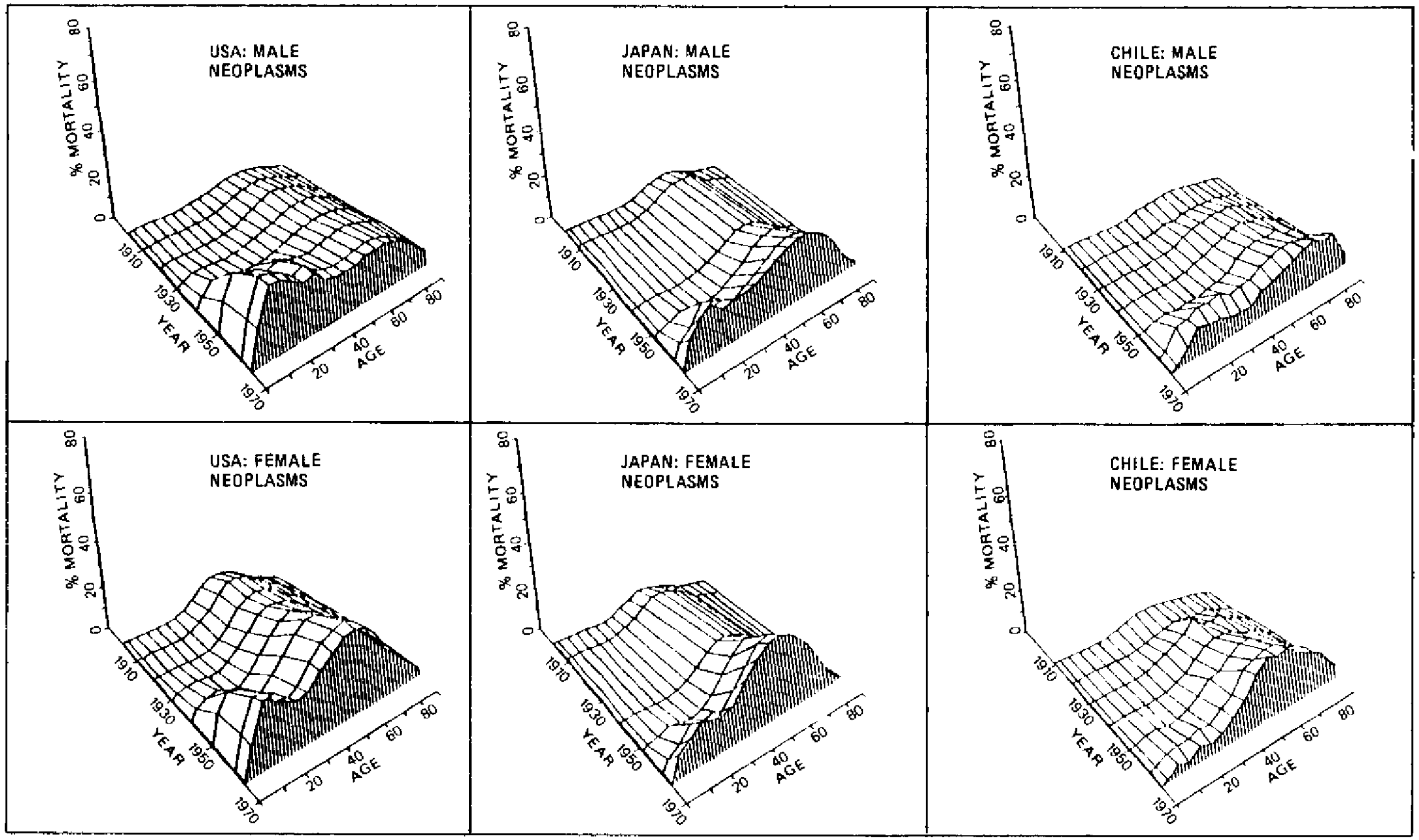


FIG. 4. Plots of six surfaces representing evolution of relative importance of malignant and benign neoplasms, 1900–1964, for USA, Japan and Chile, both sexes, age 0 to 90.

TABLE III. DESCRIPTIVE PARAMETERS FOR CATEGORY 3, MALIGNANT AND BENIGN NEOPLASMS

		USA		JAPAN		CHILE	
		MALE	FEMALE	MALE	FEMALE	MALE	FEMALE
1900	AVG%,0-90Y	3.41	6.58	2.49	3.00	--	--
	MEAN AGE	58.3	55.2	59.0	55.3	--	--
	S.D.	17.0	15.0	12.0	12.3	--	--
	SKEWNESS	-.67	-.10	.10	.37	--	--
	KURTOSIS	.42	-0.00	.28	-.10	--	--
1910	AVG%,0-90Y	4.71	8.67	3.92	4.53	2.50	3.66
	MEAN AGE	57.4	55.2	58.7	55.2	59.9	56.8
	S.D.	18.2	15.7	13.1	12.7	14.9	15.4
	SKEWNESS	-.69	-.21	-.12	.28	-.25	.05
	KURTOSIS	.26	.07	.29	-.25	.03	-.70
1920	AVG%,0-90Y	5.90	9.14	--	--	2.51	3.37
	MEAN AGE	58.6	56.3	--	--	58.8	56.7
	S.D.	17.5	15.8	--	--	17.6	16.5
	SKEWNESS	-.73	-.24	--	--	-.71	-.45
	KURTOSIS	.52	.12	--	--	.28	.11
1930	AVG%,0-90Y	7.75	11.49	--	--	4.56	6.34
	MEAN AGE	54.5	53.5	--	--	59.7	57.8
	S.D.	21.1	18.0	--	--	16.2	15.5
	SKEWNESS	-.60	-.34	--	--	-.53	-.23
	KURTOSIS	-.38	-.06	--	--	.13	-.18
1940	AVG%,0-90Y	10.29	14.55	5.04	6.33	5.45	7.10
	MEAN AGE	50.0	50.1	55.8	53.2	59.1	57.8
	S.D.	23.4	19.7	15.2	14.4	17.6	15.9
	SKEWNESS	-.37	-.32	-.70	-.37	-.77	-.36
	KURTOSIS	-.91	-.30	.97	.62	.47	.11
1950	AVG%,0-90Y	16.26	20.65	8.75	10.40	7.29	9.97
	MEAN AGE	42.8	45.0	52.7	51.3	57.0	55.8
	S.D.	24.7	21.6	17.8	16.1	19.2	17.2
	SKEWNESS	.06	-.14	-.68	-.43	-.71	-.45
	KURTOSIS	-1.21	-.74	.19	.30	.06	.12
1960	AVG%,0-90Y	20.17	24.27	15.68	17.38	13.09	13.23
	MEAN AGE	40.5	42.8	46.1	46.1	49.9	53.3
	S.D.	24.4	21.7	21.2	19.2	23.1	18.7
	SKEWNESS	.20	-.04	-.27	-.25	-.21	-.38
	KURTOSIS	-1.17	-.84	-.85	-.50	-1.03	-.23
1964	AVG%,0-90Y	20.92	25.42	19.78	21.62	12.48	15.48
	MEAN AGE	40.6	42.1	43.5	43.6	51.9	54.2
	S.D.	24.4	21.9	22.2	20.1	22.8	18.7
	SKEWNESS	.19	.01	-.11	-.15	-.41	-.41
	KURTOSIS	-1.18	-.90	-1.02	-.73	-.93	-.24

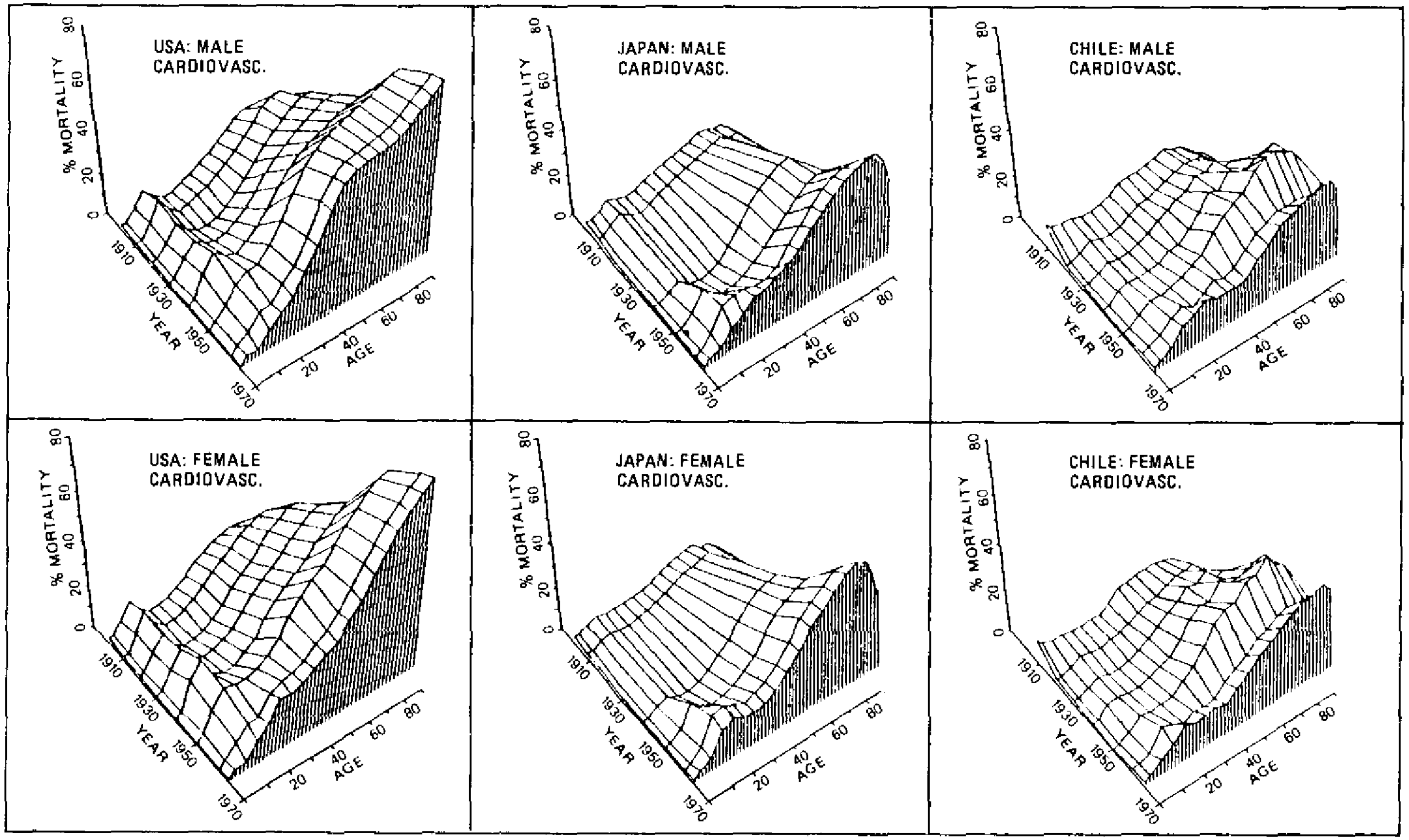


FIG.5. Plots of six surfaces representing evolution of relative importance of cardiovascular diseases, 1900 1964, for USA, Japan and Chile, both sexes, age 0 to 90.

TABLE IV. DESCRIPTIVE PARAMETERS FOR CATEGORY 4, CARDIOVASCULAR DISEASES

		USA		JAPAN		CHILE	
		MALE	FEMALE	MALE	FEMALE	MALE	FEMALE
1900	AVG%,0-90Y	19.52	18.93	15.95	14.52	--	--
	MEAN AGE	57.5	57.0	55.5	55.3	--	--
	S.D.	22.0	22.0	22.2	22.2	--	--
	SKEWNESS	-.70	-.67	-.53	-.53	--	--
	KURTOSIS	-.41	-.43	-.61	-.61	--	--
1910	AVG%,0-90Y	23.71	23.31	16.47	14.90	15.37	15.61
	MEAN AGE	59.5	58.9	56.4	56.1	52.1	53.4
	S.D.	22.2	22.3	21.8	21.4	23.3	22.7
	SKEWNESS	-.78	-.75	-.58	-.56	-.45	-.52
	KURTOSIS	-.32	-.35	-.50	-.48	-.72	-.56
1920	AVG%,0-90Y	24.13	24.34	--	--	14.43	15.26
	MEAN AGE	61.8	61.8	--	--	56.1	57.4
	S.D.	21.4	21.5	--	--	20.9	21.0
	SKEWNESS	-.91	-.92	--	--	-.52	-.59
	KURTOSIS	.06	.09	--	--	-.52	-.45
1930	AVG%,0-90Y	29.02	27.71	--	--	16.96	17.40
	MEAN AGE	60.1	60.7	--	--	58.7	59.8
	S.D.	21.5	21.9	--	--	20.3	19.8
	SKEWNESS	-.73	-.81	--	--	-.59	-.65
	KURTOSIS	-.29	-.21	--	--	-.35	-.18
1940	AVG%,0-90Y	34.12	31.18	17.72	17.34	20.00	21.08
	MEAN AGE	58.5	59.8	60.5	59.0	61.1	61.2
	S.D.	21.7	22.5	17.4	17.8	20.3	20.2
	SKEWNESS	-.60	-.75	-.84	-.73	-.70	-.71
	KURTOSIS	-.50	-.35	.72	.32	-.18	-.13
1950	AVG%,0-90Y	43.11	39.40	20.17	21.89	24.88	25.33
	MEAN AGE	58.7	59.9	58.3	57.8	60.4	60.9
	S.D.	20.8	21.8	20.6	20.1	20.1	19.4
	SKEWNESS	-.49	-.68	-.76	-.68	-.69	-.68
	KURTOSIS	-.60	-.44	-.15	-.21	-.21	-.16
1960	AVG%,0-90Y	43.38	38.46	28.26	28.45	20.59	20.14
	MEAN AGE	58.7	60.9	56.9	57.0	57.4	58.7
	S.D.	20.7	21.5	21.3	21.4	22.2	22.0
	SKEWNESS	-.45	-.69	-.57	-.62	-.55	-.67
	KURTOSIS	-.63	-.46	-.57	-.53	-.63	-.50
1964	AVG%,0-90Y	42.67	37.86	30.48	29.80	23.36	24.40
	MEAN AGE	59.0	61.2	58.1	58.4	57.5	58.6
	S.D.	20.4	21.2	20.9	21.4	22.7	22.4
	SKEWNESS	-.43	-.68	-.56	-.68	-.57	-.64
	KURTOSIS	-.66	-.49	-.55	-.42	-.73	-.61

Figure 1 shows an XYZ plot in which points on the X-axis represent ages from 0 to 90, points on the Y-axis represent calendar years from 1900 to 1964, and points on the Z-axis represent the percentages of total non-accidental deaths due to respiratory TBC for US males.

From Fig.2 and Table I the contrasts may be seen among the evolutions in the three countries of the relative importance of respiratory TBC. In the USA the curves for this disease declined steadily from 1910 to 1964, with a corresponding decrease in the average level from about 14% to less than 1% for both sexes, with little change in the other parameters. In Japan the disease was quite important in 1950, with an average level of about 18%, and decreased rapidly to an average level of 5% in 1964, about equal to that for the USA in 1950. In Chile the disease shows a relatively slow decline, from a high average level of about 19% in 1940 to a level of about 8% in 1964.

In Fig.3 and Table II we present a statistical description of the evolution of infectious and parasitic diseases. In the USA this group of diseases has been declining slowly from an average of about 12% in 1900 to 2.2% in 1964. This change has affected both sexes, with little change in the other descriptive parameters. In Japan the average level of 8.7% in 1900 increased to 10.8% in 1940 and declined to 3.6% in 1964. There is only a 1.4% difference between the values for the USA and Japan in 1964. For Chile the decrease has been slower, starting with 14% in 1910 and reaching a 5.2% level in 1960. A marked increase in kurtosis appeared in 1964, due to the decrease in the proportion of deaths from this cause after 10 years of age.

Figure 4 and Table III show the differences in the evolutions of malignant and benign neoplasms in the three countries. In the USA this group of diseases has been increasing from an average level of about 5% in 1900 to 23% in 1964. This change has affected both sexes, and marked changes in the other descriptive parameters have occurred: the mean age decreased from 56.8 to 41.4 years; the standard deviation increased from 16 to 22 years; skewness changed from -0.4 to 0.10, reflecting the shift to younger ages; and the almost normal kurtosis of the data (0.2 in 1900) was lost around 1940 with the values approaching −1.1 in 1964.

In Japan the same trend began in 1940: the mean age decreased from 54.5 to 43.4 years in 1964; the standard deviation increased from 14.8 to 21.2 years; the skewness changed from −0.55 to −0.13 with the shift to younger ages; the bimodality appeared around 1950, with values of the kurtosis changing from 0.8 in 1900 to −0.9 in 1964. In 1900 the average level was 5% in the USA and 2.6% in Japan; in 1940 the levels were 12.5 and 5.7%, and in 1964 they were 23.2 and 20.7%, respectively. There was only a 2.5% difference between the levels for the USA and Japan in 1964.

In Chile the same phenomena took place at a much slower pace beginning in 1930: the mean age went from 58.8 to 53.1 years; the standard deviation increased from 15.9 to 20.8 years; the skewness did not change; the bimodality appeared around 1960, corresponding to a change in kurtosis from 0.9 in 1950 to -0.6 in 1964. In 1910 the average level was 6.7% in the USA and 3.1% in Chile; in 1940 the levels were 12.5 and 6.3%, and in 1964 they were 23.2 and 14.0%, respectively. There was a 9.2% difference between the levels in the USA and Chile in 1964.

Figure 5 and Table IV present the different evolutions of cardiovascular diseases in the three countries. This category is composed of a great variety of diseases, some of which, such as rheumatic fever, have infectious causes, and are correlated negatively with industrialization and with age. Others, such as cardiac ischemic disease, correlate positively with age and with industrial development.

In the USA the relative importance of this group of diseases has been increasing steadily from 1900 to 1950, from an average level of 19.2% in 1900 to 41.3% in 1950, and remained constant through 1964. From 1900 to 1940 the descriptive parameters are nearly equal for both sexes. From 1940 to 1964 a progressive difference in shape developed, with males showing an increase in the relative importance at earlier ages.

This phenomenon can be seen in Fig.5 as a 'bump' around the age of 40 for males that is not found in the female data. This is a unique feature of the USA data that is not present in the Japanese or Chilean data.

For Japan the relative importance evolved in a different way. There was very little change up to 1940; the average level was about 15% in 1900 and about 17.5% in 1940. From 1940 to 1964 there was an increase in the relative importance of both infectious and degenerative components.

In Chile the evolution in the relative importance of cardiovascular diseases has been irregular, with a maximum around 1950 at about the age of 65 for both sexes. As in Japan, from 1960 to 1964 there was an increase in the relative importance of both infectious and degenerative components (Fig.6).

4. CONCLUSIONS

Our use of age- and cause-specific death ratios for three selected countries has shown changes in the health of these populations that are consistent with their histories of development.

The changes in relative importance of respiratory TBC and other infectious and parasitic diseases clearly reflect the impact of the changes in quality of life in the three countries.

The increases seen in the levels of neoplasms and cardiovascular diseases agree with previously found correlations with industrial development.

Before attempting any analysis of the health effects of pollution, an assessment must be made of the actual level of health already attained in a community.

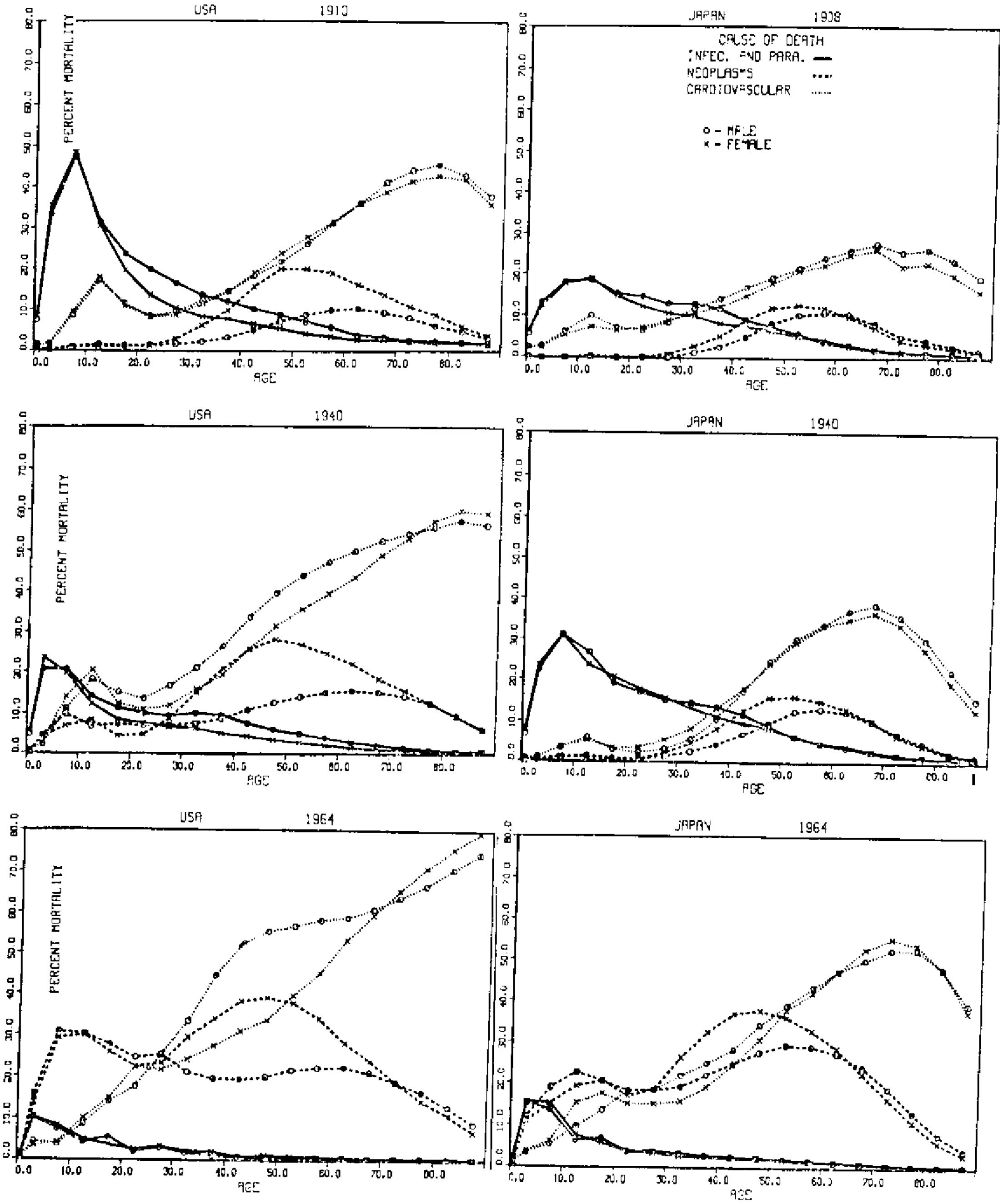
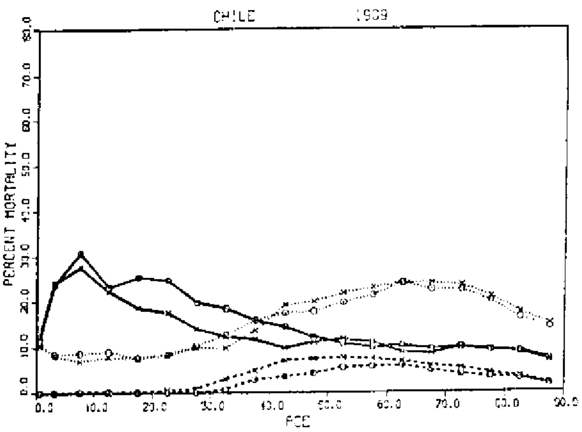
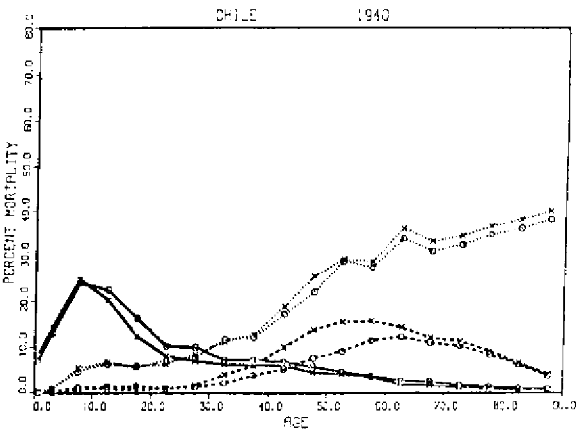
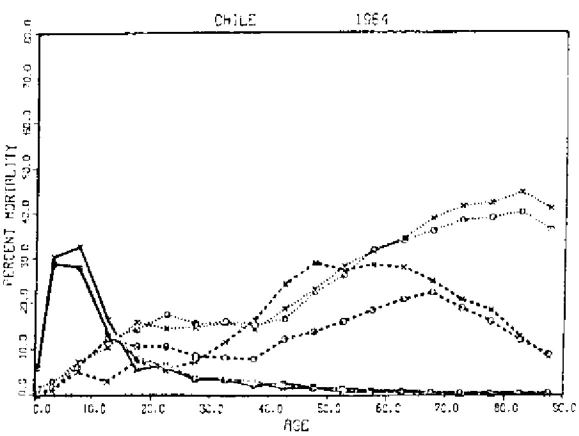


FIG. 6. Evolution of age-specific relative importance of infections and parasitic diseases, neoplasms and cardiovascular diseases in three countries, 1900–1964.


CHILE
PERCENT MORTALITY
AGE


CHILE 1940
PERCENT MORTALITY
AGE


CHILE
PERCENT MORTALITY
AGE

We must remember that both positive and negative effects are to be expected from changes in industrial development due to changes in energy policies.

To evaluate this necessary health base-line, the relative importance of different causes can be of particular value when an estimate of the size and age distribution of the exposed population is either difficult or impossible.

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DISCUSSION

K. SUNDARAM: I was very impressed with your presentation, which indicated a changing pattern of disease during the period of study in the USA, Japan and Chile. You also said that a developing country such as Chile is following the USA disease pattern, only many years behind. Could this be due to differences in the efficiency of diagnosis?

S.R. BOZZO: I do not think that in countries like Chile with good statistical data – a fact acknowledged by WHO – neoplasms or cardiovascular disease would be overlooked in 1965.

K. SUNDARAM: Would the same pattern remain if you expressed cancer and cardiovascular disease as a death-rate rather than as a percentage? This might help to provide a better insight into the effects of the changing environment on human health.

S.R. BOZZO: What we are doing is to compare the statistical differences between disease-specific mortality rates and disease-specific death ratios. We believe that a very good correlation (~1.0) will be obtained. Both values will be interchangeable.

J. SINNAEVE: In Fig.4, concerning malignant and benign neoplasms, an increase in fatalities in the 10–20-year age range was observed towards the end of the period studied. Could you comment on this phenomenon?

S.R. BOZZO: As causes of death such as infectious diseases disappear in that age span, other causes such as neoplasm become increasingly significant.

ENERGY ANALYSIS OF THE COAL FUEL CYCLE: COMMUNITY HEALTH AND RESOURCE CHANGE IN AN APPALACHIAN COAL COUNTY

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Abstract

ENERGY ANALYSIS OF THE COAL FUEL CYCLE: COMMUNITY HEALTH AND RESOURCE CHANGE IN AN APPALACHIAN COAL COUNTY.

In spite of steadily expanding coal development in this decade in the USA, there has been little systematic assessment of occupational and public health implications of increased production in specific regions of the USA. Preliminary analysis of a prototype Appalachian area is presented. Anderson County, Tennessee, the prototype area chosen for evaluation, lies in the Upper East Tennessee Coalfield. This county is uniquely suited for study since every process of the coal fuel cycle (extraction, transport, combustion, power production and waste disposal) takes place within the county boundary. By extensive exploitation of both surface and underground methods of extraction, this county has maintained a leading position in Tennessee's coal production for several years. Concepts of energy analysis and systematized data presentation were used to convert information gathered from diverse sources into comparable energy units (kcal). Concepts and methodology implemented in the analysis can be applied most appropriately to existing conditions in other countries of the Appalachian Coal Basin. Findings are presented for calendar year 1978. For the year of study, the major energy loss to the county was depletion of the coal resource base by use of inefficient mining techniques (a loss of 10.5×10^{12} kcal fuel equivalents). Another loss is to community health, which is depleted by lost productivity of, and compensation payments to, victims of mining accidents and occupational disease such as 'black lung' (15×10^9 kcal). Another countywide depletion process is roadbed and bridge deterioration caused by large volumes of heavy coal-haul vehicular traffic (10×10^9 kcal). These losses are being borne mainly by residents of the Appalachian host region, with little systematic compensation by consumers of the coal resource. It is expected that these losses will increase in magnitude as national coal use increases.

* Operated by Union Carbide Corporation under Contract W-7405-eng-26 with the US Department of Energy.

1. INTRODUCTION

The principal objective of performing an energy analysis is assessment of energy flows embedded in the production of goods and services [1]. A systematic assessment of this type is particularly useful in understanding fuel cycles and can be used as a tool for long-term planning [2]. The system of symbols and diagrammatic data presentation developed by Odum and Odum [2] to facilitate communication of energy concepts is incorporated into the present analysis. By analyzing a prototypical area in detail, the author wishes to provide an improved perspective of regional issues that should be resolved prior to nationally mandated increases in production.

Coal production throughout the Appalachian Basin has been expanding steadily since the Arab Oil Embargo of 1973. Approximately 60% of the nation's coal is mined in Appalachian states [3]. Current energy-use projections indicate that eastern bituminous coals will be in great demand for electricity production into the next century. To reduce the adverse effects of future coal industry expansion, systematic assessment of net energy and occupational or public health implications is needed. Long-term insights into fuel-cycle efficiencies may be obtained by applying basic principles of energy analysis to delineate transfers, subsidies and losses [1].

Anderson County, the prototype chosen for analysis, lies in the Upper East Tennessee Coalfield (Fig. 1). This county is uniquely suited for study since every process of the coal fuel cycle (extraction, transport, combustion, power production and waste disposal) takes place within the county boundary. By extensive exploitation of both surface and underground methods of extraction, Anderson County has maintained a leading position in Tennessee's coal production for several years.

The political county boundary, rather than a natural one of geology or topography, was chosen because relevant data are available only for political units. Data sources include accident records on file in the county sheriff's office, traffic volume counts from the state Department of Transportation, occupational accident and injury records from the Mine Safety and Health Administration (MSHA), U. S. Department of Labor (DOL) compensation records for black-lung victims, weigh bills from U. S. Department of Energy (DOE) steam plants, and statistics released by the Fuels Planning Branch of the Tennessee Valley Authority (TVA). Concepts and methodology implemented in the analysis can be applied generally to any coalfield, but the results would be used most appropriately to describe existing conditions in the Appalachian Coal Basin. Findings are presented for calendar year 1978, the most recent year for which comparable data are available.

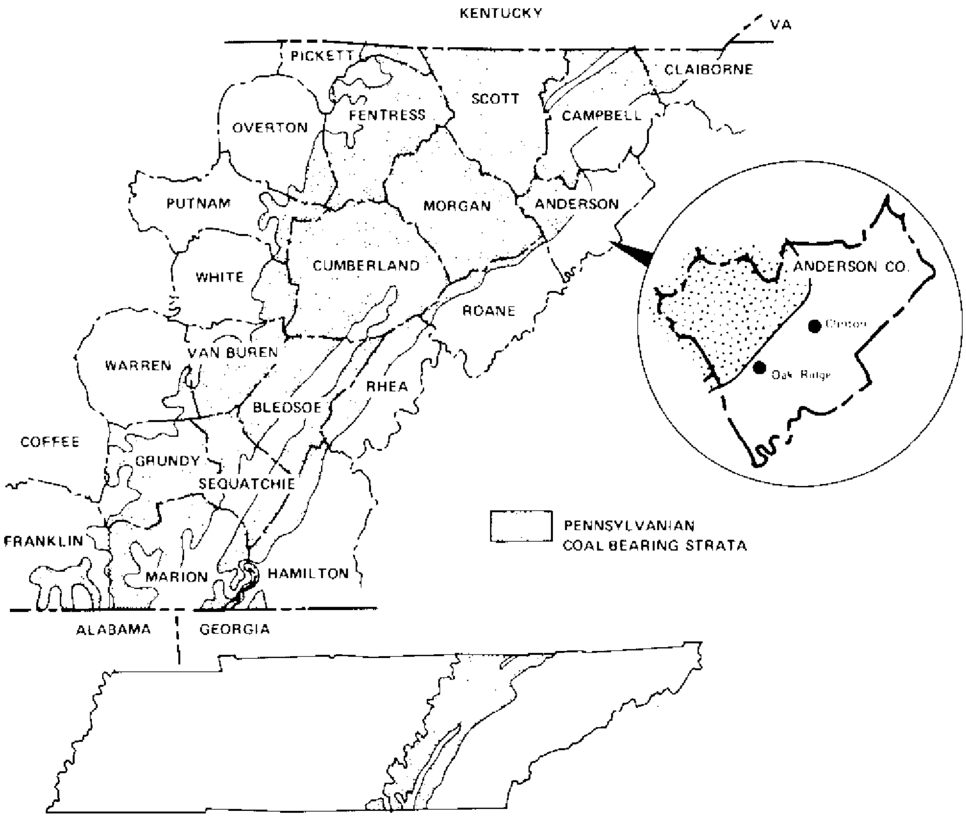

FIG.1. Coal-producing counties of the Appalachian Basin in Tennessee (after Luther [7]).

2. RESOURCE DESCRIPTION

Anderson County covers an area of 86 800 ha, approximately 33% of which overlies bituminous coal-bearing strata (Fig. 1). The acreage of currently known, commercially held mineral lands equals 25 438 ha [4]. In 1974, measured area disturbed by surface mining equaled 2876.5 ha [5]. By averaging seam thickness into production figures for 1975 and 1977 [6], an additional 134.5 ha were calculated to be disturbed by the beginning of the calendar year 1978. These values represent an estimate of area disturbed only by the physical removal of coal (i.e. not additional off-site disruptions such as haul roads, landslides, erosion, etc.).

The size of the Anderson County coal reserve was last measured by the Tennessee state Division of Geology in

1959 [7]. Later estimates, corrected for production and reserve depletion since 1959, have been used to calculate current remaining strippable reserves [defined as those found within 37 m (120 ft) of the surface] of 294.6×10^{12} kcal (42.7×10^{6} short tons) and deep minable reserves of 590.6×10^{12} kcal (85.6×10^{6} tons) [8]. Of course, these estimates do not discriminate among coal beds underlying municipalities, major waterways, or public lands, where extraction is not practical. Neither do these estimates incorporate the amount of reserve accessibility lost by the form of extraction used.

Depletion factors of 1.25 for surface-mined coal and 2.00 for deep-mined coal used in this analysis have been developed by the U.S. Bureau of Mines to account for national reserve losses incurred during mining. These factors are dimensionless multipliers of production and represent the coal lost to future recovery by underground room-and-pillar methods, blasting fracture, and other extraction inefficiencies [9]. County-specific depletion factors could not be estimated because of the current lack of quantifiable information.

3. EXTRACTION AND PROCESSING

In 1978, 13 surface and 30 underground mines in Anderson County reported annual production figures of 4.7×10^{12} kcal (0.7×10^{6} ton) and 9.3×10^{12} kcal (1.4×10^{6} ton), respectively [10]. Since these two basic extraction methods are quite different in terms of labor intensity, capital expenditure and occupational hazards, they will be compared and contrasted.

Energy inputs required for operation and maintenance include fuel-based energies and labor. Available data characterizing these inputs for local mines were sketchy and had to be augmented by assumptions regarding equipment supplies, overhead, and fixed costs [11]. By the use of appropriate kcal/$ conversion factors [12] corrected for inflation since 1974, and the calculational procedure outlined in Gregg [13], energy input estimates were derived.

Occupational health and safety aspects of coal extraction are significant. The severity of on-the-job accidents makes coal-mining one of the most hazardous major occupations in the United States [14]. Mortality ratios indicate that coalminers are exposed to life-threatening situations comparable to those confronted by fire-fighters and police officers [15]. What data are available outline lost work days (LWD) from accidents as reported to the MSHA by mine operators. There is some question as to the accuracy of these data, but they comprise all that is readily available [16].

The mining population is also unusually susceptible to respiratory disease, including coal-workers' pneumoconiosis (CWP), emphysema, influenza, tuberculosis and bronchitis [17]. Estimates vary on the number of black-lung-disabled coal-miners and their survivors currently living in the county. By assuming identical numbers of claimants for black-lung benefits in 1980 and 1978, 682 living and disabled miners and dependents were estimated to reside in the study area [18]. If it is further assumed that each claimant received minimum monthly benefits of $219.90 [19], and that the energy equivalent of such payment represented a form of insurance compensation, expenditures represent 7241.3×10^6 kcal in 1978.

A sample of 32 complete files from a local DOL Black Lung Field Office revealed a mean age of 69 for East Tennessee miners at the time of compensation award. Actuarial tables developed by Metropolitan Life Insurance Company [20] indicate a life expectancy of 11.4 years for white males of age 69. If we assume that these figures are characteristic of Anderson County populations and that each of the estimated 682 claimants was a black-lung victim who could have performed 20 h/week of light activity (890.6 kcal fuel equivalents/h) [13] if good health had been retained, we estimate a loss-of-work function to equal 6922.3×10^6 kcal. This value represents loss to the community. The summed black-lung cost estimated in this analysis includes neither expenditure for medical treatment and hospitalization nor an evaluation of medically diagnosed black-lung victims unable to receive compensation. These data were not included in available files.

The energy sum of black-lung payments and loss of work is 14163.6×10^6 kcal. Occupational disease is a major consumer of energy for workers and communities directly affected by this fuel cycle. If our assumptions are reasonable, the annual energy cost for supporting disease-disabled miners exceeds that of operator payments into the Black Lung Benefits Trust Fund by a factor of 5. (The Fund was established by the Black Lung Benefits Act of 1972 to compensate victims of CWP). This factor would be even greater if hospitalization and treatment costs were included. Such a comparison is supported by data from the Social Security Administration and DOL verifying that only 0.03% of all black-lung compensation between 1970 and 1977 was paid by the coal industry [21].

4. TRANSPORT

Coal-related transport in Appalachia occurs either on railways that are marginally maintained or on roads that were never designed to support large volumes of overweight traffic. Frequent travel by 3- and 4-axle dump trucks and 5-axle tractor trailers has resulted in reduced pavement life and stress to spanning structures between bridge piers [22].

As in most Appalachian counties, the sites of coal extraction in Anderson County are topographically removed from market locations. This isolation requires development of transport systems from the mine site to local processing facilities and remote markets. Trucks are used for short hauls to local tipples and washers and local markets; railcars are used for transport to more distant combustion facilities. Most coals in the Upper East Tennessee Coalfield are steam-grade; principal coal-haul routes in Anderson County reflect contract purchases by the TVA, DOE, and out-of-state power utilities.

Estimates of energy costs required to keep these routes in minimal repair (860×10^6 kcal) or maintain them in good condition (10276×10^6 kcal) have been made on the basis of existing truck volume counts, regional road damage figures, and interviews with the Anderson County Road Commission [22-24]. These values have been converted to fuel energy equivalents by assuming appropriate conversion coefficients [12]. If bridge upkeep expenditures were included (20 of them exist on the designated rural haul routes), the value would be at least 2 orders of magnitude greater. Repair of roadbed deterioration is clearly a countywide energy-consuming process.

Although significant, road maintenance expenditures are not the only transport factors requiring evaluation. Travel over damaged pavement can increase fuel consumption by 34% [21]. Overloading trucks also increases truck fuel consumption. Passenger vehicle and heavy truck traffic are often incompatible on twisting, narrow, Appalachian roads. Summarization of 1978 traffic accident descriptions on file in the Anderson County Sheriff's office included sideswipes involving at least one passenger vehicle and a tractor trailer or flatbed truck hauling mining equipment, a head-on collision between a passenger vehicle and a semitractor trailer, a truck-train collision, and the running off the road of vehicles transporting either fuel or explosives. Estimated personal injury and personal property damage were calculated to equal an additional 31.0×10^6 kcal.

5. COMBUSTION

Combustion facilities having a direct impact on Anderson County systems are the Bull Run Steam Plant, a 950-MW(e) (4571×10^6 kWh) TVA facility on the Clinch River; the Kingston Steam Plant, a 1700-MW(e) (9694×10^6 kWh) TVA facility downstream in Roane County; and the Y-12 Steam Plant, a

2.6×10^6 kg/d [approximately equivalent to 51 MW(e)] steam facility operated by DOE [25].

The two facilities operating within the county boundary do not consume significant portions of Anderson County coal. Bull Run contracts with a single eastern Kentucky firm for 10.1×10^{12} kcal/a (1.83×10^6 ton) to be shipped by rail. The majority of Y-12's coal supply originates from strip mines in nearby Campbell County. However, both Bull Run and Y-12 employ local personnel and emit combustion products into local and regional airsheds. For the year prior to that of our assessment, the Kingston facility received approximately 54% of its annual 24.9×10^{12} kcal (4.4×10^6 ton) by truck from Anderson County sources [26]. We have assumed that 1978 contracts followed the same proportionality. Airshed deterioration caused by emission of particulates and SO_2 from Kingston has been cited in a recent suit in which TVA was charged with violation of the Clean Air Act by neighboring states, citizen's groups, and the Environmental Protection Agency [27].

Known occupational safety costs for operation and maintenance of steam power facilities are relatively small (4.2×10^6 kcal). Accidents were normalized to reflect the percentage of coal originating from Anderson County mines, and accumulated LWDs were converted to kcal.

6. DISCUSSION

All quantified energy transfers are summarized in Fig. 2. Potential coal-bearing lands and estimated recoverable reserves are considered energy storage units within the coal fuel cycle. Work is performed when decisions to extract are made by the industry and mineral owners, who are often the same individuals. Such externally made decisions act as a control to initiate and determine the magnitude of work to be performed. A secondary result of the decision is to convert potential coal-bearing lands to mined-out properties. These lands may eventually become productive if proper reclamation procedures are applied and revegetation is successful. Some sites have been abandoned and will pass through successional stages until some climax community develops. The area and productivity of both types are at present undetermined. Inventory work has been initiated by the Federal Office of Surface Mining under mandate by the Surface Mining and Control Act of 1977.

The fuel-based energies (14.7×10^{10} kcal) of heavy equipment, liquid fuel, spare parts, etc.,[1] interact with the human energies (4.8×10^{10} kcal) of labor and management to extract saleable coals (14.0×10^{12} kcal). Estimated depletion

[1] (and industry consumption of electricity (14.6×10^6 kcal))

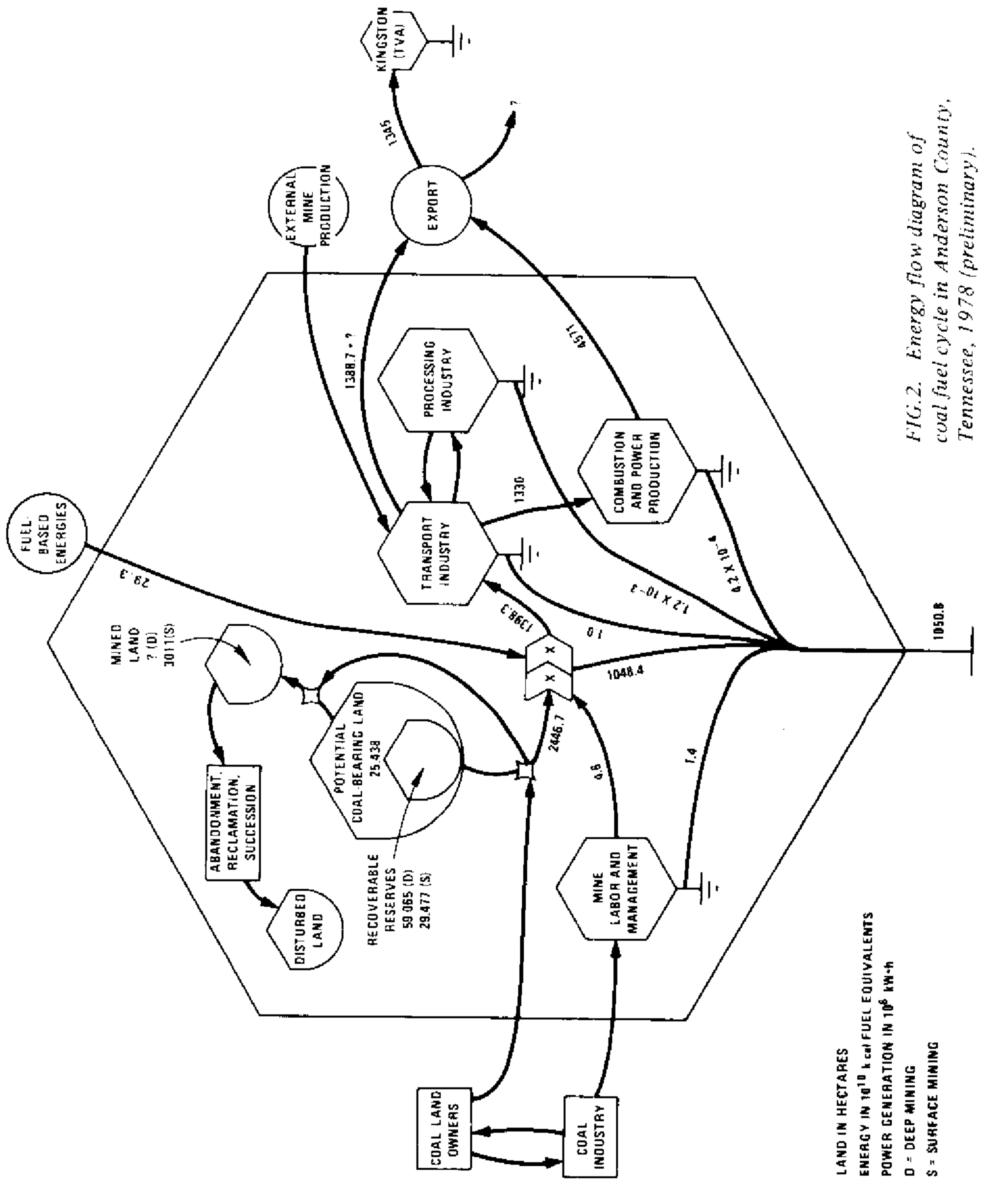

FIG.2. Energy flow diagram of coal fuel cycle in Anderson County, Tennessee, 1978 (preliminary).

of the resource base by inefficient recovery techniques appears to be the greatest caloric loss of the entire system (10.5×10^{12} kcal). If the depletion factors developed by the U.S. Bureau of Mines [9] are correct for existing Anderson County extraction, losses to future recovery are equivalent to approximately 75% of the caloric energy included in coals sent to market. Verification of these assumptions should be the basis for field evaluation.

Losses in productivity of the work force caused by accidents, and the inability of former miners to support themselves because of respiratory disease, are included in the partial degradation value of 1.4×10^{10} kcal. Additional information on costs of medical care and the incidence of chronic occupational health impairment would make this value more accurate. Additional outputs of the extraction process include changes in watersheds, mine drainage, blasting damage to aquifers and homes, and loss of agricultural productivity in bottomlands frequented by mine-induced flooding. More analysis will be needed to collect these data.

The transport industry receives coal from sources both inside and outside the county boundary, and uses designated corridors for intercounty and intracounty shipping. The largest single buyer of locally produced coal is the Kingston Steam Plant, and industry use of roadways reflects this market relationship. The total quantity of energy transported to county combustion facilities, of course, includes supplies brought in from outside sources. The degree and quantity of coal movement between processing and transport are current unknowns, as are the energy investment and energy consumption represented by truck fleets, unit trains, crushers and washers used by these two segments of the cycle. The partial degradation loss of 1.0×10^{10} kcal for transport represents the sum of lost workdays and property damage resulting from coal-related accidents on local roadways, plus an estimate of road repairs and construction needed to bring coal transport corridors up to federal standards. The quantity of unsaleable waste produced by processing has not been determined to date.

Of the two facilities in the county producing steam or electricity, only one exports energy. The Bull Run Steam Plant transmitted 4571×10^6 kWh to the TVA power grid[2] in 1978. All steam produced at the Y-12 DOE facility is consumed within the government reservation. The partial degradation value of 4.2×10^6 kcal represents lost workdays to the employees in accidents attributable to the burning of local coals or accidents affecting the local work force. Potential public health changes were not estimated.

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[2] (390×10^{10} kcal)

The main incentive to the initiation and maintenance of the coal fuel cycle is money, which is the principal input to the system encouraging current growth and development. Quantitative information on monetary flows is extremely difficult to locate and is considered too incomplete for inclusion at present.

7. SUMMARY AND CONCLUSIONS

Preliminary results from an energy analysis of the coal fuel cycle in an Appalachian coal county have provided a systematic assessment of hidden energy subsidies in extraction, transport, processing and combustion. Current results indicate that the system operates at an annual energy deficit of approximately 350×10^{10} kcal. A major loss is depletion of the coal resource base by use of inefficient mining techniques. Although of smaller magnitude, reductions in work force and community productivity from occupational accidents, disease, and road maintenance requirements for transport also appear to be significant. Further assessment is needed to verify assumptions and characterize additional data bases.

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DISCUSSION

M. EL DESOUKY: Do you not think that expenditure on the reclamation of mining sites should be included in the cost incurred by the community?

A.P. WATSON: Yes, this value should definitely be included, as should the loss of agricultural land and damage to homes caused by flooding and landslides produced by mining. However, a federal act concerning surface mine control and reclamation (PL 95-87) was not passed by Congress until the late summer of 1977. To date, an inventory of abandoned mined lands requiring reclamation has not been completed, nor has the State of Tennessee maintained adequate records on the total area and cost of reclamation. I am still attempting to quantify this variable, but it is at present unknown. The current analysis was for the year 1978, too soon for inventory results to be included.

I.M. TORRENS: If the net energy balance for extracting the coal in Anderson County is negative from your analysis, and some of the input is expensive oil and labour as against a relatively cheaper coal output, how can this economic activity continue?

A.P. WATSON: The flow of dollars runs countercurrent to the flow of energy in most human systems subject to this type of analysis. If money is the desired product, then many processes that are energy-inefficient can be subsidized. This appears to be the case in the coal fuel cycle example. The greatest entropy (degraded energy) loss is the coal left in the seam that cannot be reasonably removed. Until now, this loss has been discounted and not included in the cost of extraction.

I.M. TORRENS: How much of the energy input consists of unquantifiable or social costs not paid for by the mine operator?

A.P. WATSON: I am assuming that you include in your term 'social costs' the loss of agricultural land and property damage produced by floods and landslides, destruction of local water supplies, all occupational respiratory diseases and their treatment, blasting damage to buildings and so on. Very little of this cost is supported by the industry in the form of wages or compensation payments. I am not sure that these factors will always remain unquantifiable; the data are just very difficult to obtain.

There are a limited number of examples where operators of mines or blending facilities have recognized the larger problem and attempted to mitigate it. One operator of a coal washer in Anderson County added a surcharge of 10 cents per ton on all deliveries to his plant, which he then turned over to the County for road maintenance. However, the sum total of his contribution could pay for only very basic maintenance on the road leading to his plant. The three bridges on that road (declared unsafe for passage by school buses) will require several million dollars to replace under federal subsidy.

Some operators also dredge streams filled by mine sediment (to reduce flooding) and oil the roads to keep down dust. However, this is not common.

E. SIDDALL: The coal deposit in Anderson County is of high grade and is mined very efficiently. It is a major energy source which would be the envy of many countries in the world. If your calculations show that its exploitation results in a net import of energy into the country, does this not indicate that your method of calculation must be wrong?

A.P. WATSON: On the contrary. The estimate simply quantifies the well-known fact that mineral source areas become depleted through time. Expenditure of increasing quantities of energy gradually becomes necessary to extract less accessible reserves. In addition, available data lead me to disagree with your statement that coal in this region is mined in an efficient manner.

Session III

RADIOACTIVITY IN COAL

Chairman
Z. JAWOROWSKY
Poland

ARE HEAVY METALS AND RADIUM IN MAN RELATED TO EMISSIONS FROM COAL BURNING?

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Abstract

ARE HEAVY METALS AND RADIUM IN MAN RELATED TO EMISSIONS FROM COAL BURNING?

Concentrations of ^{226}Ra, Cd, Pb and Zn in bones of contemporary residents of four regions of Poland are not geographically related to the current industrial dust emissions. Contrary to the levels in atmospheric precipitation, concentrations of ^{226}Ra and Pb in human bones were found to be lower in the contemporary than in the pre-industrial population.

1. INTRODUCTION

During the past 100 years the content of mineral dust in precipitation in southern Poland increased by approximately a factor of 150, and the content of ^{226}Ra and Pb increased by a factor of 50 and 16 respectively [1, 2]. This was probably caused by the fallout from the large industrial centres in Silesia and Czechoslovakia. An effect of this fallout in the industrial southern and south-western regions of Poland is that the concentrations of ^{226}Ra, Cd, Pb and Zn in soil and of ^{226}Ra and Pb in plants are higher than in the northern and eastern rural areas [3–5]. Also, the concentration of ^{226}Ra in the pine trees in southern Poland increased about tenfold during the past century [6].

However, as will be seen in this paper, geographical and historical investigations reveal that the increased environmental pollution did not increase the concentrations of ^{226}Ra and several stable heavy metals in the bones of the inhabitants of Poland.

2. MATERIALS AND METHODS

To study the geographical distribution of radionuclides and heavy metals in the Polish population, we collected samples of ribs from 202 newborns, infants, children and adults deceased in 1969 and 1970 in two highly polluted (Wrocław and Kraków) and two less polluted (Białystok and Gdańsk) regions of Poland. These two types of regions differ by more than an order of magnitude in the degree of air pollution, as indicated by the industrial dust emission (Table I).

TABLE I. MEAN CONTENT OF Pb, Zn, Cd AND ^{226}Ra IN BONES OF CONTEMPORARY RESIDENTS OF FOUR REGIONS OF POLAND *(No. of samples in parentheses)*

Region and industrial dust emission per year[a]	Pb	Zn	Cd	^{226}Ra	
	($\mu g \cdot g^{-1}$ d.w.[b] ∓ SE[c])			($pCi \cdot g^{-1}$ Ca ∓ SE)[e]	($pCi \cdot g^{-1}$ d.w. ∓ SE)
Białystok 15 000 t	9.3 ∓ 1.3 (45)	n.d.[d]	n.d.	0.038 ∓ 0.005 (45)	0.0060 ∓ 0.0009 (45)
Gdańsk 39 000 t	17.8 ∓ 2.5 (51)	112.1 ∓ 6.3 (47)	5.3 ∓ 0.8 (51)	0.032 ∓ 0.005 (50)	0.0057 ∓ 0.0005 (49)
Kraków 500 000 t	3.5 ∓ 1.0 (41)	95.3 ∓ 9.0 (41)	n.d.	0.029 ∓ 0.003 (41)	0.0049 ∓ 0.0006 (39)
Wrocław 680 000 t	16.2 ∓ 4.7 (46)	89.4 ∓ 6.7 (15)	3.7 ∓ 0.4 (41)	0.084 ∓ 0.011 (44)	0.0125 ∓ 0.0015 (44)

[a] After Ref.[7].
[b] d.w. = dry weight.
[c] SE = standard error.
[d] n.d. = not determined.
[e] 1 Ci = 3.7×10^{10} Bq.

Two or three ribs were taken from each body and pooled in one sample, except ribs taken in Gdańsk from 20 newborns (10 males and 10 females) which were pooled in two samples, because of the low mass of each specimen. This resulted in 184 samples divided into four sets. They were representative of a mixed urban and rural population of the particular region.

To study temporal variations, we collected various types of bones from the skeletons of adults living in the Kraków region in southern Poland between the 11th and 19th Centuries buried in graves located in four churches in the Kraków area. Most of these skeletons were preserved in wooden or stone coffins which were stored in dry underground crypts, but some of them were buried directly in the loess under the floor of a church. The majority of the bone samples would be representative of the rich townspeople, noblemen and clergy. Bone samples from the skeletons of adults and children living in the 3rd Century were collected from a cave in Kroczyce, about 70 km north-west of Kraków. These bones were lying on a dry floor of limestone rock. Part of this collection had been analysed and reported earlier [2]. Six bone samples were collected from Egyptian mummies, stored in the National Museum in Warsaw, the age of which was assumed as 12th Century BC. In various parts of Georgia, USSR, we collected bones from graves located directly in the ground, the age of which ranged from 40th Century BC to 19th Century AD. In the catacombs under the San Francisco Convent in Lima, Peru, we collected femur samples from persons deceased between 1700 and 1800.

After external impurities and the remains of soft tissue had been removed by scrubbing, the bones were dried in an oven at 105°C to a constant weight. Samples weighing on average 15 g were placed in quartz beakers, wet-ashed with concentrated nitric and perchloric acid, and dissolved in hydrochloric acid. ^{226}Ra was determined by the radon emanation method and the results were corrected for radioactive decay. Pb was determined by standard dithizone colorimetry and Cd and Zn by atomic absorption spectrophotometry. The analytical procedures used are described in detail in Ref. [8]. Detection limits per 15-g sample for the analytical methods used are 0.02 pCi[1] for ^{226}Ra and 0.15 μg for Pb, Cd and Zn. The mean analytical error for ^{226}Ra was about ∓10% and about ∓5% for the metals.

3. RESULTS AND DISCUSSION

All results were analysed using the graphic statistical method [9]. We found that most of the results have the log-normal distribution. In these cases we calculated the arithmetic from the geometric means [10]. As may be seen in

[1] 1 Ci = 3.7 $\times 10^{10}$ Bq.

Table I, the content of metals in the bones of contemporary residents of Poland does not seem to be related to the level of environmental pollution in particular regions. The lowest mean content of Pb (3.5 $\mu g \cdot g^{-1}$ of dried bone) was found in the heavily industrialized and polluted region of Kraków, whereas in the typically rural Białystok region, with 33 times lower industrial dust emission and considerably lower automotive traffic, the mean lead content in the bones was 2.7 times higher. The highest concentrations of Pb, Zn and Cd were found in bones from the Gdańsk region in northern Poland, which is less industrialized than the southern part of the country, and has a low level of heavy metals in vegetation [5]. The mean concentrations of Pb, Zn and Cd in the bones of contemporary residents of four regions studied (of 12.1, 102.1 and 4.6 $\mu g \cdot g^{-1}$ of dried bone, respectively) do not differ substantially from the values reported from other countries and are assumed to be typical for 'standard man' [11].

The whole country's mean concentration of ^{226}Ra found in the bones of contemporary residents of Poland of 0.007 $pCi \cdot g^{-1}$ of dried bone is similar to the world mean of 0.0085 $pCi \cdot g^{-1}$ in areas with normal radiation background [12]. The highest mean concentration of ^{226}Ra in bones was found in the Wrocław region, with the highest level of natural terrestrial radiation background in Poland [13]. It seems that the high content of natural radionuclides in the geological structure in this region is responsible for the elevated ^{226}Ra level in bones, rather than the industrial dust emission which is similar to the Kraków region, where the ^{226}Ra bone level is the lowest in the country.

The lack of geographical relationship between the levels of ^{226}Ra and heavy metals in bones and the industrial pollution of the atmosphere suggests that the inhalatory intake of these contaminants is overwhelmed by some other factors, presumably related to water and food supply.

The concentrations of lead in the bones of residents of southern Poland deceased between the 3rd and 20th Centuries reveal a striking feature. The low concentration of the 3rd Century increased to very high levels in mediaeval times and remained high until the end of the 19th Century (Table II). In the 20th Century the concentration dropped again to a level only twice as high as 1700 years ago (Table III).

The mean lead concentration of 2.9 $\mu g \cdot g^{-1}$ of dried bone in the 3rd-Century samples probably represents the natural level of lead in human bones in this region, as metallic lead was not in use in Poland at this time [14]. In the bone from the 11th Century, the lead concentration was 92.5 $\mu g \cdot g^{-1}$ of dried bone, i.e. 30 times higher than the mean in 3rd-Century samples. Still higher lead concentrations were found in the bones from the 13th and 17th to the 19th Centuries, reaching the highest level of 267.4 $\mu g \cdot g^{-1}$ in the 17th-Century samples.

The mean lead concentration in the oldest, i.e. 3rd-Century, bones from Poland is 2.6 times lower than in the Egyptian bones from the 12th Century BC. Relatively low levels of lead were found in the bones from Georgia, USSR.

TABLE II. MEAN CONTENT OF Pb AND ^{226}Ra IN BONES OF RESIDENTS OF KRAKÓW REGION BETWEEN THE 3rd AND 19th CENTURIES *(No. of samples in parentheses)*

Century	Pb ($\mu g \cdot g^{-1}$ d.w.[a] ∓ SE[b])	^{226}Ra ($pCi \cdot g^{-1}$ Ca ∓ SE)	^{226}Ra ($pCi \cdot g^{-1}$ d.w. ∓ SE)
3rd	2.9 ∓ 0.6 (18)	0.167 ∓ 0.026 (13)	0.030 ∓ 0.005 (13)
11th	92.5 (1)	Not determined	Not determined
13th	83.3 ∓ 64.8 (11)	0.303 ∓ 0.770 (3)	0.043 ∓ 0.025 (33)
14th	6.4 ∓ 1.4 (3)	0.141 ∓ 0.120 (2)	0.041 ∓ 0.049 (2)
17th	267.4 ∓ 101.0 (6)	0.077 ∓ 0.026 (5)	0.013 ∓ 0.006 (5)
18th	53.1 ∓ 8.9 (14)	0.052 ∓ 0.011 (9)	0.010 ∓ 0.002 (9)
19th	87.2 ∓ 42.4 (13)	0.075 ∓ 0.009 (9)	0.014 ∓ 0.002 (9)

[a] d.w. = dry weight.
[b] SE = standard error.

These samples were collected in countryside graveyards and probably represent the rural population, i.e. different from the Polish mediaeval set of samples collected in the city of Kraków. In the 18th Century in Lima, Peru, the mean lead concentration in bones (46.9 $\mu g \cdot g^{-1}$) was similar to the levels in Kraków at the same time (53.1 $\mu g \cdot g^{-1}$). This probably reflects similarities in the household and medical exposures to lead in both communities.

Unlike lead, the concentration of ^{226}Ra in the bones of Polish population did not change much through eighteen centuries. The mean concentration of this radionuclide in contemporary bones in southern Poland is about six times lower than in the 3rd Century. In the bones from Egypt and Peru this concentration was also higher than the present global mean of 0.0085 $pCi \cdot g^{-1}$ [12].

TABLE III. MEAN CONTENT OF Pb AND ^{226}Ra IN BONES OF RESIDENTS OF SOUTHERN POLAND, EGYPT, USSR AND PERU *(No. of samples in parentheses)*

Country	Century	Pb	^{226}Ra	
		($\mu g \cdot g^{-1}$ d.w.[a] ∓ SE[b])	($pCi \cdot g^{-1}$ Ca ∓ SE)	($pCi \cdot g^{-1}$ d.w. ∓ SE)
Poland	3rd	2.9 ∓ 0.6 (18)	0.167 ∓ 0.026 (13)	0.030 ∓ 0.005 (13)
Poland	11th to 19th	68.5 ∓ 13.5 (48)	0.067 ∓ 0.006 (28)	0.013 ∓ 0.002 (28)
Poland	20th	3.5 ∓ 1.0 (41)	0.029 ∓ 0.003 (41)	0.005 ∓ 0.001 (41)
Egypt	12th BC	7.4 ∓ 0.6 (6)	0.078 ∓ 0.013 (6)	0.015 ∓ 0.002 (6)
USSR	40th BC to 19th AD	5.0 ∓ 1.3 (19)	0.861 ∓ 0.800 (21)	0.217 ∓ 0.207 (21)
Peru	18th	46.9 ∓ 3.9 (59)	0.050 ∓ 0.005 (59)	0.013 ∓ 0.001 (59)

[a] d.w. = dry weight.
[b] SE = standard error.

The highest content of ^{226}Ra in the bones from Georgia may have resulted from the contact of skeletons with the ground. The decrease of ^{226}Ra content in the bones of the contemporary population probably results from the introduction of drinking-water treatment systems which remove 70–99% of ^{226}Ra [15] and from decreases in the consumption of bread and cereals which are principal sources of ^{226}Ra intake in the Polish diet [16, 17].

During the 20th Century, when the lead content in atmospheric precipitations in Poland increased dramatically, the concentration of lead in human bones decreased, paradoxically, to a level not much higher than natural background. Also, the level of ^{226}Ra in the Polish population did not increase by the recent rise of ^{226}Ra content in precipitation. The same factor which caused the increase of lead content in the atmosphere, i.e. the development of industry, improved living and hygienic conditions, and resulted in decreasing the lead level in the population by eliminating lead from cooking utensils, tableware, storage containers, water pipelines, etc., which were once the source of epidemic contaminations [18]. Probably even more important for the observed effect was the elimination of lead compounds from therapeutic use. From time immemorial until recently, lead compounds were administered orally for treatment of pneumonia, lung haemorrhage, haemorrhagic nephritis, diarrhoea and a score of other diseases [19, 20], and externally in the form of ointments, plasters, douches, mouth-washes, baby and toilet powders, cosmetics, etc. [18]. This indicates that the old household and medical sources of lead exposure contributed much more to the contamination of the general population than the new environmental sources. This rather optimistic situation may well be long-lasting. Even with continued pollution of the environment at the current rate, it seems unlikely that we might in the future lose much of what has been gained in the recent past by the improvement of hygienic conditions.

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DISCUSSION

L.D. HAMILTON: Have you carried out any measurements in polluted areas on tissues other than bone – for example, cadmium in kidneys?

Z. JAWOROWSKI: Yes, we are currently measuring ^{226}Ra, ^{210}Pb, U, Th, Pb, Cd and Zn in lungs, kidneys, livers and bones of inhabitants of Silesian urban areas. We plan to continue this study for the next three or four years.

R. WILSON: Have you any information on acid rain during the period of study and the acidity levels in lakes?

Z. JAWOROWSKI: In the Himalayas and Ruwenzori and near Mt. McKinley in Alaska we measured the acidity of glacier ice and found pH values of 4.0 to 4.5 in many samples from the pre-industrial period.

B.L. TRACY: You have shown that there is no apparent correlation between industrial emissions and radium and lead in human bone. Could one possible explanation be that these elements in fly-ash and dust are highly insoluble and are hence not available for biological uptake?

Z. JAWOROWSKI: Yes, I think it could.

L.J. RAMBERG: I should like to make a comment. A comparison of heavy metals in man – between regions or in time – must take into consideration the fact that the food distribution system (the ratio of locally produced food to the total amount ingested) may well be different between regions or have changed during history. It must also take into account the fact that the solubility (leachability) and uptake by vegetation of many heavy metals is strongly pH-dependent. Differences in soil properties between regions are common and may well also occur with time as a result of, for example, changes in the use of fertilizers and lime.

A.C.M.J. BOUVILLE: With regard to the radioactive substances emitted from the burning of coal, you studied ^{210}Pb, ^{226}Ra and U. Would it not be of greater value to measure thorium isotopes (^{230}Th or ^{232}Th) in human organs and tissues? Since the main pathway to man for thorium is inhalation, this element might prove to be a better index of radioactive contamination produced by the burning of coal.

Z. JAWOROWSKI: You are quite right, and we are now measuring thorium in soft tissues.

M. EL DESOUKY: In your presentation you talked about mineral dust – which is a rather comprehensive term. What was the exact composition of this dust?

Z. JAWOROWSKI: This is mineral dust which was measured following dry-ashing at 450°C of residues left after the evaporation of ice samples.

ASSESSMENT OF THE IMPACT OF RADIONUCLIDES IN COAL ASH

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Abstract

ASSESSMENT OF THE IMPACT OF RADIONUCLIDES IN COAL ASH.

An assessment of the potential environmental and health impacts of radionuclides in the coal fuel cycle is being conducted at Mound. This paper describes studies evaluating the potential for migration of radionuclides from ash disposal sites. Studies at a power plant burning western-US coal dealt with an assessment of potential radiation doses from coal ash ponds and leachate discharges of radionuclides from the ponds. Emanation of ^{222}Rn from the ash is relatively low. The emanation of ^{222}Rn from the ash pond (^{226}Ra at 4.5 $pCi \cdot g^{-1}$) is predicted to be about six times less than from soil (^{226}Ra at 1 $pCi \cdot g^{-1}$). Ash with ^{226}Ra at 25 $pCi \cdot g^{-1}$ would approximate emanation of ^{222}Rn from soil. At 1000 m from the centre of the ash pond area, ^{222}Rn from the ash pond is predicted to be 1000 to 6000 times less than background (0.1 to 0.5 $pCi \cdot ltr^{-1}$). Pathways exist for transport of radionuclides leached from ash into the aquifer beneath the holding ponds, but concentrations of radionuclides in water leaving the ponds are lower than concentrations in groundwater which is upgradient of the ponds. Leachability of the ash is quite low, on the order of 0.002% in one month, and flow of ash-sluicing water (3% of the volume of the ponds each day) has actually diluted normal background concentrations of radionuclides in the aquifer between the ponds and the adjacent river.

INTRODUCTION

Coal is expected to play an increasingly important role in meeting energy needs of the United States as we move to reduce our dependence on imported fossil fuels. The combustion of coal releases trace elements, including naturally occurring radionuclides, to the atmosphere as vapors and particles; and these particles have relatively greater concentrations of certain trace constituents than the feed coal [1-4]. Although the environmental impact of these residuals on ecosystems is not certain, control of these residuals appears to be a reasonable goal within acceptable cost limits.

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* Operated by Monsanto Research Corporation for the US Department of Energy under Contract No. DE-AC04-76-DP00053.

A project was initiated at Mound Facility to assess the fate of radionuclides in coal and their associated health and environmental effects. Phase I included in the experimental design a plan to broadly survey pathways of radionuclides in the coal fuel cycle for western U. S. coal.

Samples of coal collected in Phase I from 19 active western mines that produce 65% of the coal mined in the province had an average concentration (95% confidence limits) for uranium-238, uranium-234, and radium-226 of 0.31 ($\pm$0.10) pCi/g [5].[1] The data for uranium-238 are somewhat below the national average (0.60 pCi/g) [6]. Pathways of radionuclides in a coal-fired steam electric generating plant (1000 MWe) were investigated through analysis of coal, bottom ash, fly ash, stack effluents, airborne particulates, soil and vegetation. Bottom ash and fly ash contained relatively higher concentrations of uranium-238, uranium-234, thorium-230, lead-210, and polonium-210 than did the feed coal. Although some small fraction of the radionuclides apparently bypassed the electrostatic precipitator as vapors or in association with very fine particles as do other constituents of coal, the electrostatic precipitator effectively controlled emission of radionuclides associated with fly ash. Atmospheric dispersion calculations, using data on stack effluents, indicated maximum depositions over a 20-yr period to be 0.1 to 1.0% of measured background concentrations [7].

Coal ash is presently accumulating in the United States at a rate in excess of 60 million tons annually [8]. Roughly 10% of this is being used in a variety of products such as concrete, aggregate in stabilizing roadways, and a filler in putty, paint and wallpaper. Fly ash may also be used as a "dewatering" agent for waste slurries from flue gas desulfurization systems. The vast majority of waste products from flue gas desulfurization systems will be directly disposed of in ponds or used in landfill [9]. The potential of such practices for enhancement of radiation doses to man warrants further evaluation.

The scope of Mound's assessment of the radionuclide concentration in coal refuse and the potential migration of radionuclides from ash disposal sites has remained limited to western

[1] 1 Ci = 3.7×10^{10} Bq.

U. S. coal. Los Alamos National Laboratory is evaluating eastern U. S. coal. Mound's efforts are reported here for the following tasks:

1. Evaluation of potential for airborne radiation doses from coal ash.

2. Evaluation of leachate discharges of radionuclides from coal ash ponds to groundwater and surface water.

POTENTIAL RADIATION DOSES FROM COAL ASH

In this study of ash ponds at the George Neal Power Station near Sioux City, Iowa, U.S.A. (Figure 1), two pathways are considered which may potentially result in radiation doses to individuals in the vicinity of a site where ash has been deposited. The first pathway is the emanation and subsequent dispersion of radon-222 from the ash. Upon decay, radium-226 forms radon-222, which is an inert, radioactive gas. The radon can diffuse through voids within and between particles of ash, and then emanate from the surface of the ash pile. Upon dispersion, the radon and its short-lived decay products may be inhaled, and thus produce a radiation dose in the lungs of individuals in the vicinity of the ash pile. Also, the decay products of radon may deposit on the ground and subsequently decay to lead-210, bismuth-210 and polonium-210. These radionuclides may then be taken up by food crops grown in the area, and individuals may receive radiation doses from the ingestion of these foods. Because ash contains concentrations of radium-226 that are slightly greater than those generally found in soil, the doses resulting from the emanation and dispersion of radon-222 may be larger than those resulting from background radon-222.

The second pathway is the suspension and dispersion of the ash itself. The dried areas of the ash ponds at the Neal Station seemed to have formed a crust on the surface that eliminated or retarded the suspension of the ash by wind. However, it seems likely that if the ash were to remain unstablized over a long period of time this crust would break up as the result of various weathering processes, and the ash could then be suspended and dispersed by wind. Thus, the ash is a potential source of radiation doses to individuals in the vicinity, through inhalation of suspended ash and ingestion of foods which may take up deposited radionuclides. It should be pointed out that the intent here is to estimate the potential impact of the ash if it is left in place over a long period of time in an unstabilized condition (so that suspension is possible). Dispersion of

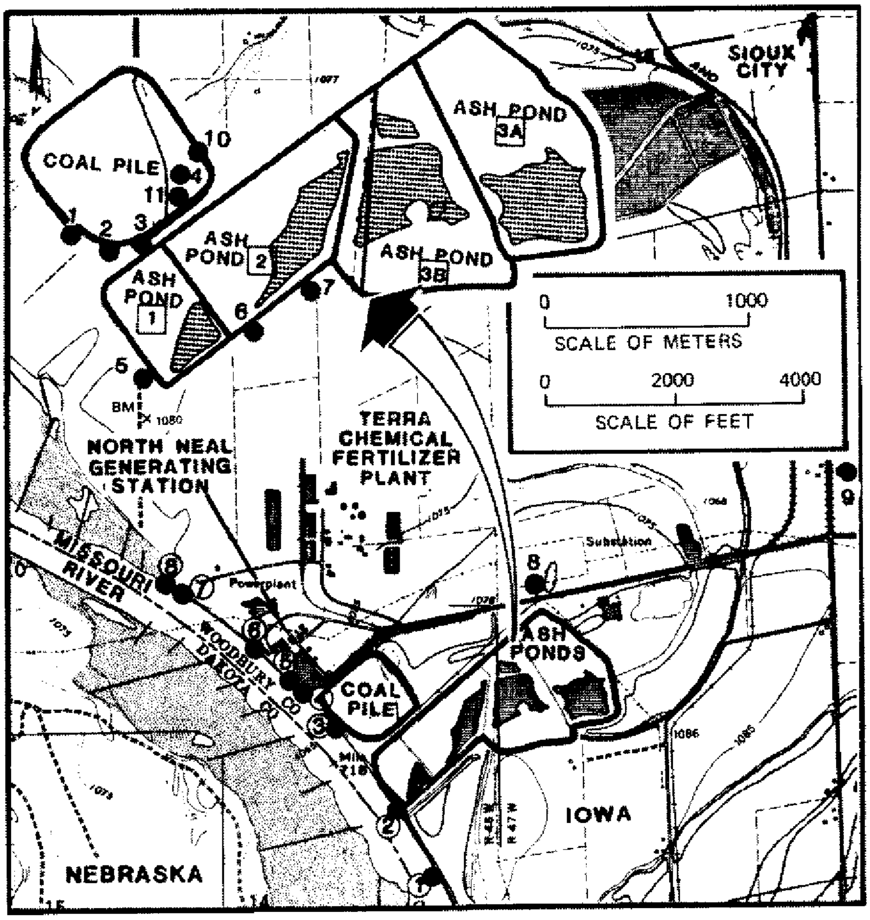

FIG.1. The 11 groundwater monitoring wells (●), the 8 surface water sampling locations (○), and the 4 ash ponds (□). The scale refers to the base map only; its values must be divided by 2 for use with the enlargements of the coal pile and the ash ponds.

ash resulting from short-term disturbances, such as digging and moving the ash, is not considered here.

Physical characteristics of the ash, emanation of radon-222, and derivation of source terms for dispersion modeling are described elsewhere [10].

Atmospheric dispersion calculations were performed to estimate ground-level air concentrations and deposition rates of radionuclides as well as doses from inhalation and ingestion pathways, resulting from the release of 1.42 Ci of radon-222 per year from the ash piles at the Neal Station, the ash containing radium-226 at 4.5 pCi/g. A computer code [11], which is currently being used by the Tennessee Valley Authority for uranium mining and milling licensing calculations, was used to perform the calculation. This computer code uses a sector-average Gaussian dispersion plume model and uses methodologies consistent with U.S.N.R.C. Regulatory Guides 1.109 [12] and 1.111 [13]. Further, the code explicitly calculates decay and growth of all radionuclides in the uranium-238 decay series.

The meteorological data used in the dispersion calculations were obtained from the National Climatic Center in Asheville, North Carolina. These data were based on observations covering the 5-yr period, January 1970 through December 1974, at the Sioux City, Iowa, weather station.

For each receptor location, the following were calculated: ground-level air concentration of radon-222, polonium-210, lead-214, bismuth-214 and polonium-214; ground-level air concentration of the short-lived decay products of radon-222 in working levels (WL); deposition rates of polonium-218, lead-214, bismuth-214 and polonium-214; annual average inhalation dose to the bronchial epithelium; and annual average ingestion dose to bone, total body, kidneys, liver and G.I. (gastrointestinal) tract.

The location for which the calculated doses were the largest was at 1000 m in the WNW direction, hereafter referred to as "1000 m WNW". A summary of the calculated concentrations, deposition rates and doses for this location is presented in Table I.

There are no regulatory guidelines or limits which directly apply to release of radionuclides from coal-fired power plants. However, the concentrations of radon-222 (6.9×10^{-5} pCi/liter) and its decay products (8.0×10^{-8} WL) are small in comparison with the limits listed in 10CFR20 for release to uncontrolled areas: 1 pCi/liter for radon-222 or 1/30 WL (0.033WL) for the short-lived decay products of radon-222. In fact, the radon-222 concentration in Table I is small in comparison with background concentrations which are typically on the order of 0.1 to 0.2 pCi/liter [14]. Also, the doses are small in comparison with proposed limits under 40CFR190 for uranium fuel-cycle facilities of 25 mrem to total body or any organ other than thyroid.

The effects resulting from radon-222 emanating from the ash piles at the Neal Station are insignificantly small, even if they were an addition to background. However, the ash is physically replacing the soil that would be there if the ash were not.

TABLE I. ANNUAL AVERAGE CONCENTRATIONS, DEPOSITION RATES AND DOSES AT 1000 m WNW

Parameter	Calculated Value
Rn-222 conc.	6.9×10^{-5} pCi/liter[a]
Po-218 conc.	4.9×10^{-5} pCi/liter
Pb-214 conc.	5.4×10^{-6} pCi/liter
Bi-214, Po-214 conc.	5.9×10^{-7} pCi/liter
Combined Radon Decay Products	8.0×10^{-8} WL[b]
Po-218 dep. rate	3.0×10^{-10} μCi/m^2 -sec
Pb-214 dep. rate	2.6×10^{-11} μCi/m^2 -sec
Bi-214, Po-214 dep. rate	2.3×10^{-12} μCi/m^2 -sec
Inhalation dose to bronchial epithelium	8.5×10^{-3} mrem
Ingestion dose to:	
Bone	3.9×10^{-4} mrem
Total Body	1.6×10^{-5} mrem
Kidneys	3.8×10^{-4} mrem
Liver	1.3×10^{-4} mrem
G. I. Tract	3.4×10^{-6} mrem

[a] 1 curie = 3.7×10^{10} Bq.

[b] WL = working level.

According to Harley [14], a reasonable average value for radon-222 emanation from soil is 1600 pCi/cm^2-yr or about 0.5 pCi/m^2-sec. If it is assumed that this value is applicable to the soil in the vicinity of the Neal Station, and that the derived value for ash of 0.09 pCi/m^2-sec is accurate, then the ash piles at the Neal Station may be emanating less radon-222 than the soil without the ash. Another way of stating this is that if the radium-226 concentration in the ash were increased to about 25 pCi/g, then the radon-222 emanating characteristics of the ash would be essentially the same as those of soil, and the effects of radon-222 emanation would be indistinguishable from background.

The value of 0.09 $pCi/m^2 sec$ was derived considering only the ash as a source of radon-222; therefore, the actual radon flux from the surface of the ash pile may be somewhat larger than this value because of radon-222 emanating from the soil beneath the ash piles. However, the radon-222 flux from the surface of the ash piles should be lower than that from the surrounding soil. It is therefore concluded that radon emanation from the ash piles at the Neal Station is of no present or potential environmental concern.

Atmospheric dispersion and dose calculations were also performed for the suspension of ash [10] in the same manner as was described in the previous section for the dispersion of radon-222.

The largest doses calculated were for location 1000 m SSE. The largest inhalation dose was approximately 1.8 mrem to bone. The breakdown of doses by radionuclide indicates that, for kidneys, liver, bone and lungs, the isotopes of thorium contribute a very large percentage of the total dose. For total body and the G.I. tract, radium-226 is the largest contributor to the total dose, with the thorium isotopes making up a large percentage of the remainder.

The largest ingestion dose was approximately 40 mrem to bone. The breakdown of doses by radionuclide indicates that, for every organ, radium-226 and lead-210 contribute a very large percentage of the total dose.

It must be reemphasized that the methodology used to calculate suspension source terms is based on models which have not been verified. The intent was to arrive at a rough, order-of-magnitude estimate of potential effects should the ash ponds be allowed to be suspended and dispersed by the wind. One should keep this in mind when interpreting the above results.

The largest calculated dose was to bone and resulted almost entirely from deposition of radium-226. The concentration of radium-226 in the ash was measured to be about 4.5 pCi/g. The average concentration of radium-226 in soil is reported to be on the order of 0.7 pCi/g [15]. Data presented in the Phase I report [1] indicate that the radium-226 concentration in soil in the vicinity of the Neal Station should be in the range of about 0.6 to 2.0 pCi/g. Thus, application of the models used here to the suspension, dispersion and deposition of soil would lead to predicted ingestion doses on the order of 10 to 45% of those predicted for ash. Therefore, the dose methodology used here would predict that background ingestion doses may be an appreciable fraction of the 25-mrem limit proposed under 40CFR190. Further, it was assumed that the radionuclides in the ash were available

TABLE II. LEACHABILITY OF NEAL STATION COAL AND ASH WITH WATER

Sample	Radionuclide	Concentration (pCi/g)	Total Activity In Samples (pCi)	Total Activity In Leachates (pCi)	Percent Leached
Neal Station	U-238	0.54 ± 0.02	41	0.36	0.88
Coal	U-234	0.53 ± 0.02	40	0.65	1.6
	U-235	0.022 ± 0.003	1.7	0.025	1.5
	Th-231	0.28 ± 0.006	21	< 0.05	< 0.2
	Th-230	0.63 ± 0.10	47	0.06	0.13
	Pb-210	0.40 ± 0.12	30	4.7 ± 0.3	16
Neal Station	U-238	3.93 ± 0.09	2950	0.035	0.0012
Ash, Pond #3b	U-234	3.93 ± 0.09	2950	0.066	0.0022
(32302)	U-235	0.18 ± 0.02	135	< 0.015	< 0.01
	Th-232	2.04 ± 0.11	1530	0.057	0.0037
	Th-230	4.67 ± 0.11	3500	0.11	0.0031
	Pb-210	2.98 ± 0.23	2230	< 1.5	< 0.07
Neal Station	U-238	4.01 ± 0.13	2400	0.035	0.0015
Ash, Pond #3b	U-234	4.03 ± 0.13	2420	0.047	0.0019
(32303)	U-235	0.22 ± 0.02	132	< 0.012	< 0.01
	Th-232	1.86 ± 0.10	1120	0.017	0.0015
	Th-230	4.44 ± 0.19	2660	0.11	0.0041
	Pb-210	3.08 ± 0.35	1850	< 1.2	< 0.07
Neal Station	U-238	4.41 ± 0.14	2200	0.023	0.0010
Ash, Pond #3b	U-234	4.44 ± 0.14	2220	0.044	0.0020
(32304)	U-235	0.22 ± 0.03	110	< 0.013	< 0.01
	Th-232	1.93 ± 0.11	965	0.015	0.0016
	Th-230	4.57 ± 0.15	2300	0.055	0.0024
	Pb-210	3.62 ± 0.23	1810	< 1.3	< 0.07

for uptake by plants. Data presented in Table II indicate that the radionuclides are not easily leached from the ash and therefore may not be taken up by plants. Thus, the ingestion doses would be greatly reduced. For these reasons, it is considered that the potential impact through the ingestion pathway resulting from the suspension of ash is not significant. However, it is also recommended that a prudent approach should be taken for the long-term storage of the ash, and that the ash should be stabilized to preclude suspension and dispersion.

LIQUID DISCHARGES FROM ASH PONDS

The potential for release of radionuclides from coal ash ponds was evaluated through a series of ash leachate studies and a survey of geohydrology at the George Neal Steam Electric Generating Station, Sioux City, Iowa (Figure 1).

Leaching experiments were performed on material from Ash Pond #3B (Figure 1) of the Neal Station and coal from the Neal Station.

Distilled water and 75- to 150-g samples were placed in 1-liter Erlenmeyer flasks and shaken on a Burrell Wrist-Action Shaker or stirred with a magnetic stirrer. Replicate leachates were combined and divided into two equal samples for duplicate analyses [10].

Measured concentrations of radionuclides leached from ash and coal are given in Table II. The data show that very little of the elements of interest was found in the leachates. However, radionuclide concentrations were significantly higher in coal leachates than in ash leachates. The data also suggest that uranium-234 is more leachable from most of the samples than uranium-238 is. This suggests that the radionuclide decay process may cause activities of daughters to be less bound to the sample matrix than the parent radionuclide.

The field study involved an assessment of surface and groundwater transport of selected radionuclides from ash ponds of the Neal Station to the environs. Specific objectives included: (1) identify radionuclide levels in the surface water and groundwater regimes at the site, (2) assess the hydrologic and geologic conditions at the Neal Station from available historical information, and (3) formulate a preliminary working model describing the operating mechanism for movement of radionuclides from the ash pond system. Samples of water from ash ponds, test wells

TABLE III. COMPARISON OF CONCENTRATIONS OF TRACE ELEMENTS IN GROUND WATER AT THE NEAL STATION – JULY 1979 AND JUNE 1980

Parameter	Environment[a] River 1979	River 1980	Coal Pile 1979	Coal Pile 1980	Ash Pond 1979	Ash Pond 1980	Background 1979	Background 1980	EPA Drinking Water Standard
Radium-226[b]	<0.10	0.07	<0.10 to 0.11	0.11	<0.10 to 0.14	0.14	0.17	0.19	5.0
Gross Alpha[b]	2.73	1.74	6.90	3.18	1.76	1.5	3.08	5.75	15.0
Gross Beta[b]	4.11	6.21	13.4	9.40	3.17	9.10	3.30	5.00	50.0
Silver	0.01	0.02	0.06	0.04	0.13	0.21	0.14	0.16	0.05
Aluminum	0.06	0.53	0.22	0.69	0.25	1.64	0.39	1.74	
Barium	0.05	0.06	0.06	0.06	0.07	0.11	0.19	0.18	1.0
Beryllium	0	0	0.01	0	0	0	0	0	
Boron	0.14	0.14	0.40	0.38	0.55	0.56	0.22	0.21	
Calcium	56.2	58.5	259.0	201.0	40.0	76.0	157.0	139.9	
Cadmium	0	0.01	0	0.01	0	0.04	0	0.03	0.01
Cobalt	0.01	0.02	0.50	0.02	0.02	0.05	0.04	0.03	
Chromium	0.02	0.05	0.07	0.10	0.03	0.14	0.06	0.12	0.05
Copper	0.01	0.05	0.05	0.06	0.05	0.16	0.05	0.11	1.0
Iron	0.05	0.09	0.62	2.30	0.22	0.35	1.55	1.04	0.3
Magnesium	23.3	23.9	86.4	68.9	10.0	15.6	46.9	41.4	
Manganese	0.02	0.02	4.46	3.67	0.53	1.18	2.72	2.40	0.05
Molybdenum	0.02	0.04	0.05	0.08	0.09	0.15	0.07	0.09	
Sodium	60.4	63.9	87.1	82.7	94.6	61.6	26.9	22.1	
Nickel	0.14	0.64	0.53	1.16	0.45	1.95	0.63	1.61	
Lead	0.03	0.18	0.14	0.29	0.17	0.64	0.20	0.40	
Phosphorus	0.08	0.40	0.27	0.79	0.80	1.28	0.47	0.90	
Antimony	0.02	0.14	0.09	0.24	0.14	0.47	0.16	0.34	
Silicon	3.54	1.50	10.3	0	8.42	0	13.3	0	
Tin	0.03	0.12	0.12	0.22	0.06	0.18	0.12	0.24	
Strontium	0.59	0.49	2.09	1.39	0.39	0.59	1.40	1.12	
Titanium	0	0.01	0.01	0.01	0.01	0.04	0.01	0.03	
Vanadium	0.13	0.20	0.48	0.53	0.08	0.25	0.27	0.38	
Zinc	0.01	0.02	0.04	0.36	0.01	0.36	0.03	0.34	5.0
Potassium	4.78	5.62	7.43	7.53	5.41	7.74	6.72	7.27	

[a] Values for River, Coal Pile, Ash Pond and Background are averages.

[b] These parameters are given in pCi/l (1 Ci = 3.7×10^{10}Bq). All others are in mg/l.

(Figure 1), and the Missouri River (Figure 1) were analyzed for radionuclides as described in Styron et al. [10] for stable trace elements by Inductively Coupled Argon Plasma Emission Spectroscopy.

Samples were collected from nine groundwater monitoring wells (Figure 1) and eight river sampling locations (Figure 1) on July 16-18, 1979, and June 23-24, 1980. The following data were gathered at each monitoring well: date, time, weather, well number, water level before and after pumping, depth to well point, pH and general appearance of the water. All samples, except for fractions designated for gross alpha and gross beta analyses, were filtered and acidified at the power plant.

Data are grouped according to the particular types of environment found in the study area. The environments identified and data points (sampling locations) representing these environments include:

Environment	Data Points
Missouri River	River sample locations 1 - 8
Coal pile - groundwater	Groundwater monitoring wells 1, 2, 3, 4, 10 and 11
Ash pond - groundwater	Groundwater monitoring wells 5, 6 and 7
Background - groundwater	Groundwater monitoring wells 8 and 9

Data on concentrations of radium-226, gross alpha, gross beta, and trace elements are summarized in Table III. Groundwater samples from background wells (8 and 9) had the highest concentration of radium-226. Groundwater down-gradient of the ash ponds and coal pile had concentrations of radium-226 that fell between the concentrations found in background groundwater and surface water of the Missouri River. Gross alpha activity was also lower in ash pond and coal pile groundwater than background groundwater in 1980. The trend for gross beta activity was somewhat different. Background wells had the lowest value; the river, next lowest; and the coal pile and ash pond wells, the highest. Except for sodium, phosphorus and potassium, most stable elements that were measured were found at the same or lower concentrations in groundwater from the ash pond environment than in water from background wells.

In order to examine the groundwater gradients near the ash ponds, water levels were measured in nine shallow monitoring wells.

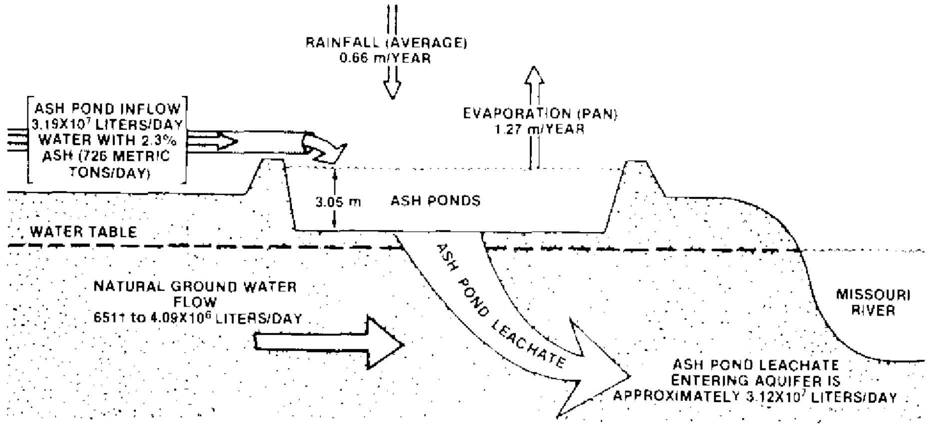

FIG. 2. Water balance for the ash ponds at the North Neal Generating Station. The surface area of the ash ponds (1, 2, 3A and 3B) is 4.23 X 10^5 m^2; the capacity of the ponds, which have an average depth of 3.05 m, is 1.29 X 10^6 m^3.

Water levels measured on July 16 and July 18, 1979, and June 23, 1980, indicate that the predominant groundwater gradient is toward the river. Groundwater gradients are slightly steeper near the river. However, they do not exceed one-half percent.

During the site investigation, no discharge through the dikes or indication of surface runoff were observed. Additionally, it was reported by personnel from Iowa Public Service Company that discharge from the ash pond has never been observed. The conclusion is evident that, except for the small amount of evaporative loss, almost all the 3.19 x 10^7 liter/day (8 424 000 gal/day) input, i.e. 3.12 x 10^7 liter/day (8 240 000 gal/day), flows directly out the bottom of the ash pond and ultimately enters the groundwater regimes, which will discharge into the river. A diagrammatic sketch representing the operative mechanism is shown in Figure 2.

In this context, we considered a possible scenario in which the burning of coal would tend to concentrate uranium and its daughter products in the ash. In this scenario, the ash disposed of in containment ponds may be selectively leached and "excessive" amounts of radionuclides made available to the groundwater for subsequent transport.

Groundwater samples representing the background environment had the highest concentration of radium-226 whereas the ash pond environment had the lowest. This observation is diametrically opposed to the proposed scenario.

Studies on leachability of radionuclides from Neal Station fly ash point strongly to limited solubility of radionuclides bound in fly ash particles. It is plausible that river water, which is relatively low in concentration of radium-226 (0.074 pCi/liter), used to sluice ash to the holding ponds, in passing from the ponds to the aquifer dilutes natural background (0.216 pCi/liter) to levels observed for the ash pond groundwater environment (0.112 pCi/liter). The short residence time of sluicing water in the pond and the low leachability of radionuclides from ash indicate that little of the radionuclides in ash is transferred to groundwater.

Comparison of concentrations of stable trace elements in the ash pond and background groundwater environments suggest that dilution by ash pond water may also be lowering natural levels of barium, calcium, iron, magnesium, manganese and strontium. The concentration of sodium in water down-gradient of the ash ponds is higher than in the background environment, but the levels are not greatly different from those found in river (sluicing) water. The highest concentration of boron is associated with ash pond groundwater.

In summary, pathways exist for the transport of radionuclides into the hydrologic regime, but concentrations of radionuclides in water leaving the ponds are less than concentrations in groundwater upgradient (Wells 8 and 9) of the ponds. Radionuclide levels observed in monitoring wells are not significantly different from levels observed in the background environment (Wells 8 and 9).

CONCLUSIONS

1. Modeling of the emanation and dispersion of radon-222 from the ash ponds at the Neal Station indicated that there is no significant potential environmental impact. In fact, because of the very small radon emanation power of the ash, the radon flux from the surface of the ash pond is predicted to be less than from typical soil.

2. Modeling of the suspension and dispersion of the ash itself indicated no significant potential impact from the radionuclides in the ash. The modeling did indicate, however, that annual doses from ingestion of foods grown in the immediate vicinity of the ash ponds could conceivably be of the same order of magnitude as

the 25-mrem limit proposed under 40CFR190 for uranium fuel cycle facilities. Therefore, it is recommended that the ash be stabilized for long-term storage in order to prevent the ash from being suspended and dispersed.

3. Pathways exist for transport of radionuclides leached from ash into the aquifer beneath holding ponds at the Neal Station, but concentrations of radionuclides in water leaving the ponds are lower than naturally occurring concentrations in groundwater upgradient of the ponds. Percolation of ash pond water into the aquifer appears to have diluted normal background concentrations of radionuclides.

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DISCUSSION

K.E. BRODERSEN: You mentioned that the radon emission from fly-ash was less than from soil. I take it that these measurements were made on samples with similar moisture content, as the results could not otherwise be compared.

C.E. STYRON: The measurements were made on dried samples.

K.E. BRODERSEN: The conclusion you drew from these measurements was that radon emission is neither an actual nor a potential problem. Does that mean that you have investigated the aging behaviour of the ash, since its properties might change with time?

C.E. STYRON: We have not looked at the effects of the aging of the ash on emanation power.

M. EL DESOUKY: At one point in your discussion of the dispersion of radon gas and its decay product, you mention that the location for which the largest concentration was calculated was at 1000 m in the WNW direction. At another point, you say that the location of maximum suspension of ash was at 1000 m in the opposite direction. What is the reason for this?

C.E. STYRON: The difference in direction for our predictions of maximum concentration of ^{222}Rn and maximum suspension of fly-ash can be attributed to the mechanisms leading to the dispersion of these components. The maximum concentration of ^{222}Rn occurs under conditions of low wind speed. Maximum suspension of ash will occur under conditions of high wind speed. Low wind speed leads to maximum radon concentration in the WNW direction, while the higher wind speeds leading to the suspension of fly-ash are in the other direction.

J. SINNAEVE: You mention very low leaching rates both for laboratory studies and field observations. What type of water was used in the laboratory studies, and could you comment on the physicochemical properties of the river-water used in the field studies?

C.E. STYRON: Distilled water was used in laboratory studies. The pH of the river water was about 7.5, that of the sluicing water was 11.0, and that of the ash pond water was 8.0 to 9.0.

J. HIBBERT: Power-station ash is sometimes used as infill material in land-reclamation programmes. This land may be used subsequently for farming purposes, including cereal production or dairying. Have you any information on the uptake of isotopes by this route into the food chain or, alternatively, do you propose to undertake investigations in this area?

C.E. STYRON: I'm afraid not. We have not studied the uptake of radio-nuclides from fly-ash in this context, nor do we have any plans at the moment to undertake such an investigation.

Z. JAWOROWSKI *(Chairman)*: Following a fairly lengthy study in Poland, it was decided that fly-ash from lignite could be used as a fertilizer and it is now so used on a large scale.

RADIOLOGICAL IMPLICATIONS OF THERMAL POWER PRODUCTION

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Abstract

RADIOLOGICAL IMPLICATIONS OF THERMAL POWER PRODUCTION.

Coal combustion, along with nuclear energy, is destined to play an increased role in Canada's energy future. An evaluation of the health risks of coal utilization is incomplete without consideration of naturally occurring radionuclides released during combustion or leached at a later time from ash disposal sites. To provide insight into this matter, an investigation was undertaken at the 4000-MW(e) coal-burning station at Nanticoke, Ontario. The measured radionuclide concentrations in air, water, soil and vegetation within 20 km of the station did not show any enhancements that could be directly attributed to emissions from coal combustion. Predictions based on radionuclide balance at the station and atmospheric dispersion calculations indicated that the enhancements in the environmental concentrations would be less than 1% of the measured background values. Inhalation of airborne activity was found to be a more significant contributor to individual dose commitments than ingestion of deposited activity. Doses were two orders of magnitude lower than those resulting from routine emissions from a nuclear generating station of similar capacity. Potential health effects from these radiation exposures were also compared with the impact of other pollutants released in coal combustion. Of the radionuclides, only uranium isotopes showed any measurable leachability from fly ash. The use of fly ash as a component of building material is discussed.

1. INTRODUCTION

At present about 70% of Canada's electric power requirements are met by hydroelectric developments, the remainder being provided by fossil fuels and nuclear energy. Based on a projected annual growth rate of 5%, the demand for electric power will have more than doubled by the year 2000. Less than half of this increased demand can be met by present and proposed hydroelectric projects. Furthermore, future oil supplies in Canada are not well assured. Consequently, an increased emphasis is being placed on both coal and nuclear energy.

Ontario Hydro, the publicly-owned utility of Canada's most populous province, has undertaken major developments in both these areas. One of the most recent projects is the Nanticoke Thermal Generating Station, situated on the north shore of Lake Erie, about 100 km southwest of Toronto. With a total generating capacity of 4000 MW(e), it is the world's largest coal-burning generating station. The fuel currently being burned at the plant is a 50/50 mixture of U.S. Appalachian and western Canadian coal. The surrounding area is largely rural, and the land is used to grow a variety of vegetables, fodder crops and tobacco.

From its beginning in 1968, the station has been equipped with modern pollution control equipment. Flue gases are cleaned of particulate matter by electrostatic precipitators with a rated collection efficiency of 99.5%, and are dispensed to the atmosphere by two 200-metre-high stacks. Waste ash, consisting of 80% fly ash and 20% bottom ash, is pumped as a slurry to an 80-hectare on-site storage lagoon. Water from the lagoon is re-circulated in a closed loop, and the lagoon is lined in order to prevent contamination of ground water supplies. Although there are no immediate plans for the utilization of this ash, it is hoped that it may eventually be marketed for landfill, roadbed construction, and as a constituent of concrete.

Since the early seventies, an extensive program has been conducted under the umbrella of the Nanticoke Environmental Management Program. Its primary purpose is to study the environmental implications of large-scale power production from coal. Within this broad objective, a multitude of individual studies have been carried out by federal, provincial, industrial agencies and universities. This has involved plume-dispersion studies; measurements of sulfur dioxide, oxides of nitrogen, hydrocarbons, ozone, particulate matter and trace elements; determinations of soil acidity; and phytotoxicological studies with emphasis on cash crops.

In 1979 a health-oriented study of the impact of the Nanticoke environmental emissions was launched by the Department of National Health and Welfare. It was soon recognized that certain gaps in knowledge had to be filled. In particular, more work was needed on radiological implications. Some studies [1,2,3] have indicated that population exposures from radionuclides in the naturally occurring uranium and thorium series released by coal-burning power stations may be comparable to those from fission products released by nuclear power stations.

This paper describes investigations carried out in the latter half of 1979 to determine whether radionuclide emissions from the Nanticoke station had any measurable effects on environmental levels, and what the most significant pathways of these radionuclides to man would be. Estimates of radiation doses and associated health risks would then facilitate comparisons with other pollutants released in coal combustion and with other forms of energy conversion. The investigations would also help to identify areas where further research is needed.

2. DESIGN OF THE STUDY

The study has focussed on atmospheric emissions of radionuclides from the stacks and on leaching or emanation of these radionuclides from ash disposal sites. Atmospheric emission rates were estimated from measured concentrations in coal and fly ash and from coal consumption data at the station. Dispersion calculations were carried out to predict radionuclide concentrations in ambient air. The predicted values were then compared with the results of experimental measurements on air filters, precipitation, soil and vegetation collected in the plant vicinity. Radiation dose calculations were carried out for the most exposed individual. Leaching of radionuclides from ash was studied by taking samples of water from the storage lagoon and by carrying out solubility tests in the laboratory.

All samples were subjected to analyses by gamma-ray spectroscopy. In addition, the environmental samples were analyzed radiochemically for ^{226}Ra and fluorometrically for total uranium. Radiochemical determinations of ^{210}Pb and ^{222}Rn were carried out on the water samples.

Dispersion calculations were based on a Gaussian plume-dispersion model with parameters from Smith [4] and Hosker [5]. Meteorological data from a nearby station (Simcoe, Ontario) were analyzed by the STAR program [6] to give average annual and average monthly probabilities for wind direction, wind speed, and Pasquill stability category.

Dose calculations were carried out for a "worst case" individual who lives at the location of the highest predicted ground-level concentration of radionuclides and who consumes only food raised or grown in the immediate area. Inhalation, ingestion and "groundshine (i.e. exposure to deposited radioactivity) were considered. A breathing rate of 2.66×10^{-4} m/s [7] and a particulate deposition velocity of 0.01 m/s [8] were assumed. Factors for the uptake by plants of ^{226}Ra, ^{210}Pb and uranium were based on our recent measurements at Port Hope, Ontario [9]. Factors for surface deposition on crops and transfer to meat and milk were taken from USNRC Regulatory Guide 1.109 [10]. Annual consumptions of various food groups were based on Nutrition Canada data [11]. Inhalation and ingestion doses were based on ICRP-30 [12], and groundshine doses were calculated by factors from Kocher [13] for exposure to an assumed one-year's deposition of activity and time fraction spent outdoors of 100%. The radionuclide ^{40}K was not included in these dose calculations because of uncertainties in its metabolic behaviour.

TABLE I. RADIONUCLIDE CONCENTRATIONS (Bq/kg) IN COAL AND ASH FROM THE NANTICOKE GENERATING STATION

Nuclide	Feed Coal	Fly Ash	Bottom Ash
^{40}K	26.4±0.6	204±4	245±4
U-Series	12.4±0.3	92±3	91±3
Th-Series	7.53±0.17	58±2	61±2

3. RESULTS AND DISCUSSION

Radionuclide concentrations in coal and ash samples from the generating station are shown in Table I. Members of the uranium and thorium series were generally found to be in secular equilibrium. The activities of uranium and thorium in coal correspond to 1.0 ppm and 1.9 ppm respectively, which are comparable to values reported by Beck et al.[14], for U.S. coals. There appeared to be no fractionation in the uranium or thorium series between fly ash and bottom ash. The concentrations of members of the uranium and thorium series in fly ash and bottom ash are 2 to 4 times as high as in local soils. Hence the possibility exists that environmental concentrations could be enhanced by the disposal of ash.

Table II gives average emission rates and values of the highest predicted ground-level concentrations on an annual average basis. Also shown are averages of the observed concentrations at the position of the predicted highest values and at a "control" location upwind from the generating station. The predicted enhancements from plant emissions are less than 1% of background values and are thus unlikely to be detectable. This is borne out by the fact that the measured concentrations show no dependence on location relative to the plant. Measurements of precipitation, soil and vegetation also showed no enhancement in radionuclide concentrations that could be attributed to emissions from the generating station.

The highest predicted ground-level concentrations of radionuclides were used to calculate the 50-year committed dose equivalents shown in Table III. It can be seen that lung and bone surface would receive the greatest radiation doses and that these are largely due to inhalation of airborne radioactivity. The ingestion pathway is of lesser importance but

TABLE II. RADIONUCLIDE EMISSIONS AND CONCENTRATIONS IN AIR

Nuclide	Emission Rate (Bq/s)	Concentrations in ground-level air ($\mu Bq/m^3$)		
		Predicted Maximum (d)	Observed at Maximum location	Observed at Cont. Locat. (e)
^{40}K	10. (a)	0.049	38±9	32±10
U-Series to ^{226}Ra	4.4 (a)	0.022	4.2±0.4	3.7±0.4
^{222}Rn + daughters	1500. (b)	7.4	-	-
^{210}Pb-^{210}Po	13. (c)	0.066	510±50	680±60
Th-Series to ^{224}Ra	2.8 (a)	0.014	-	-
^{220}Rn + daughters	920. (b)	4.5	-	-

[a] Based on concentration in fly ash and annual particulate emission of 1.529×10^6 kg [17].

[b] Based on concentration in feed coal and annual coal consumption of 3.8104×10^9 kg [17].

[c] As in (a) except that an enrichment factor of 3.0 for ^{210}Pb and ^{210}Po has been assumed in escaping ash [21].

[d] Predicted maximum ground level concentration is expected to occur 10 km NE of station.

[e] Control location is 19 km WNW (upwind of station).

would be the chief contributor to kidney and liver doses, arising mainly from ^{210}Pb and ^{210}Po. The "groundshine" doses were generally insignificant compared to internal doses. The doses obtained here are somewhat lower than those calculated by other authors [3,15] for comparable generating stations. One factor contributing to this is the greater stack heights, and hence greater atmospheric dilution, at Nanticoke.

The effective or whole-body committed dose equivalent of 0.091 μSv can be compared with a value of 20 to 30 μSv/year estimated at the site boundary of the 2000-MWe Pickering Nuclear Generating Station near Toronto [16]. However, a complete comparison of the radiological effects of coal versus nuclear energy would entail an in-depth analysis of the entire fuel cycle of each energy form. Such an analysis is beyond the scope of this paper.

TABLE III. COMMITTED DOSE EQUIVALENTS AND CANCER RISKS TO VARIOUS ORGANS

Organ	Committed Dose Equivalents (μSv)				
	Inhalation	Ingestion	Groundshine	Total	Cancer Risk[a]
Lung	0.37	0.0002	0.0008	0.37	7.6E-10[b]
Bone Surface	0.78	0.18	0.0010	0.96	4.8E-10
Red Marrow	0.062	0.011	0.0009	0.074	1.5E-10
Kidney	0.006	0.030	0.0007	0.037	-
Liver	0.008	0.037	0.0008	0.046	-
GI tract	0.0	0.0006	0.0009	0.0015	-
Gonad	0.0	0.0004	0.0007	0.0011	-
Effective dose equivalent:				0.091	
Total Cancer risk:					1.4E-9

[a] Based on ICRP Publication 26, Pergamon, Oxford (1977).
[b] Contribution from radon and daughters is 3%.

The low risks associated with radionuclide emissions from the Nanticoke station can be compared with the effects of other pollutants released in the combustion of coal. For example, from plant emission data [17] the predicted maximum ground-level concentration of SO_2 is 24 $\mu g/m^3$. If we assume that 1.5% of this is converted to sulphate, and that a sulphate concentration of 1 $\mu g/m^3$ produces 3.7 deaths per 10^5 persons per year [18], then we obtain a risk of 1.4×10^{-5} per year, which is 4 orders of magnitude higher than the risks from all radionuclides combined.

Finally, we have examined the question of possible radiation effects from the disposal of waste ash. Lagoon water samples which had been in contact with the ash for several months showed uranium concentrations of 1.4 to 2.0 μg/ltr. This indicates a slight leachability for this element from fly ash. By comparison, the maximum acceptable concentration of uranium in Canadian drinking water supplies is 20 μg/ltr [19]. The concentrations of ^{226}Ra and ^{210}Pb were less than the detection limit of 0.01 Bq/ltr. ^{222}Rn was marginally detectable with a concentration of 0.10±0.04 Bq/ltr, which confirms the conclusion of others [15] that the emanation of radon from fly ash is extremely low.

Measurements carried out in the laboratory on fly ash suspensions maintained in a slightly acid solution (pH$\sim$5) for a period of one week indicated solubilities of less than 2×10^{-3} and 1×10^{-4} for ^{226}Ra and total uranium respectively. These results must be regarded as preliminary, but they do indicate that leaching of radionuclides from coal ash is not likely to be a serious health or environmental problem.

One aspect of ash disposal which should be investigated further is the use of fly ash in concrete for buildings. A recent OECD study [20] indicates that if natural radionuclides in building materials are increased by a factor of two above normal levels, then the effective dose equivalent could be increased by 300 to 400 μSv per year. This would be a significant enhancement in background radiation and could have undesirable consequences for human health.

4. CONCLUSIONS

Atmospheric emissions of radionuclides from the Nanticoke Thermal Generating Station are insignificant compared to background levels and to routine emissions from a nuclear generating station of similar capacity. Of far greater importance are the chemical toxicants emitted in the combustion of coal. Leaching of radionuclides from waste ash does not appear to be a serious problem, but the use of ash in concrete building materials could lead to enhanced radiation exposure. More research is needed on the total fuel cycles of coal and nuclear energy.

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DISCUSSION

Z. JAWOROWSKI *(Chairman)*: I should like to comment on your remark concerning building materials. One need not necessarily expect to find an increase in radioactivity when fly-ash is used in such materials. We in Poland use it on a greater scale than any other country; during a ten-year study, the highest radioactivity was found not in contemporary bricks or other materials containing fly-ash but in bricks dating back to the 12th Century.

J. SINNAEVE: Large discrepancies appear between the data reported by you, Dr. Tracy, and those reported by Dr. Styron (SM-254/30), particularly as regards comparisons between radionuclide concentrations in ashes and soil respectively. I should like to suggest that data on the physico-chemical properties and geomorphological characteristics of the soils in these site-specific studies be included in these papers.

R.M. BARKHUDAROV: I was impressed by the depth and thoroughness of your study, Dr. Tracy. I have two questions. Firstly, did you determine the dispersion of the fly-ash and did you study the enrichment coefficients for small ash particles? Secondly, how did you obtain the coefficients for radionuclide migration from soil to food?

B.L. TRACY: Thank you. The answer to your first question is that all our predicted ground-level concentrations were based on dispersion calculations for fly-ash particulates. We did not study the enrichment coefficients for small ash particles ourselves, but used values taken from the literature. We assumed an enrichment factor of 3.0 for ^{210}Pb and ^{210}Po, and 1.0 for all the other radionuclides.

As to your second query, the enrichment coefficients for radium and uranium were taken from our studies of produce grown in contaminated soil at Port Hope, Ontario. Other values were taken from USNRC Regulatory Guide 1.109.

IMPACT RADIOLOGIQUE DES REJETS ATMOSPHERIQUES D'UNE CENTRALE AU CHARBON

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Abstract–Résumé

RADIOLOGICAL IMPACT OF ATMOSPHERIC RELEASES FROM A COAL-FIRED POWER STATION.

As the first stage of a study carried out under contract with the Commission of the European Communities for the comparative assessment of the risks to which the individuals of a regional population are exposed, the paper seeks to evaluate atmospheric releases from a coal-fired thermal power station. The station is of traditional design with an installed capacity of 415 MW(e) and uses only lignite-type coal produced from a coal basin situated nearby. Gaseous effluents are released from four stacks. The area close to the station is rural in nature: there are a number of small farms, fairly abundant natural vegetation and some fairly well-populated zones with various industries. The main feature of the weather conditions is the strong prevailing winds in the optimum direction. A radiation measurement campaign involving the main ^{238}U and ^{232}Th daughter products was carried out focussing on: (1) the coal burnt in the power station; (2) the solid residues resulting from combustion (fly ash, wet ash); (3) gaseous effluents by means of direct sampling from the two release stacks. The information obtained on the releases has made it possible, with the help of dispersion and transfer models, to evaluate the atmospheric concentration of the different radionuclides released as well as their deposition and presence in the biotope in the plant vicinity. The effective dose equivalents received by persons living in the zone of maximum exposure and consuming food produced in that zone were assessed at approximately 7×10^{-5} Sv·a^{-1} at the end of the plant's operating period. Finally, the main radionuclides were measured at a number of points near the plant with the aim of verifying the model evaluations for a particular situation.

IMPACT RADIOLOGIQUE DES REJETS ATMOSPHERIQUES D'UNE CENTRALE AU CHARBON.

Dans le cadre d'une étude menée sous contrat avec la Commission des Communautés européennes et relative à l'évaluation comparative des risques auxquels sont soumis les individus d'une population régionale, une première étape a porté sur l'évaluation des conséquences des rejets atmosphériques d'une centrale thermique fonctionnant au charbon. Il s'agit d'une

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centrale de conception classique, d'une puissance installée de 415 MW_e (utilisant exclusivement du charbon de type lignite produit par un bassin minier situé à proximité. Les effluents gazeux sont rejetés par quatre cheminées. L'environnement proche de la centrale est de nature rurale: on y trouve quelques petites exploitations agricoles, une végétation naturelle assez abondante et quelques zones d'assez fort peuplement avec présence de diverses industries. La météorologie est caractérisée par une prédominance de vents forts de direction privilégiée. Une campagne de mesures radioactives prenant en compte les principaux descendants de l'uranium 238 et du thorium 232 a été réalisée en ce qui concerne: 1) le charbon brûlé dans la centrale, 2) les résidus solides de la combustion (cendres volantes, cendres humides), 3) les effluents gazeux par prélèvement direct dans deux des cheminées de rejet. La connaissance des rejets a permis, à l'aide de modèles de dispersion et de transfert, d'évaluer la concentration atmosphérique des différents radionucléides rejetés, leur dépôt et leur présence au niveau du biotope dans l'environnement de la centrale. Les équivalents de dose effectifs reçus par les personnes résidant dans la zone d'exposition maximale et consommant des produits alimentaires issus de cette zone, ont pu ainsi être évalués à environ $7 \cdot 10^{-5}$ $Sv \cdot a^{-1}$ à la fin de la période de fonctionnement de la centrale. Enfin, les mesures portant sur les principaux radionucléides ont été effectuées en un certain nombre de points de l'environnement de la centrale. Ces mesures avaient pour objet de vérifier, pour une situation particulière, les évaluations faites à partir des modèles.

1. INTRODUCTION

Le charbon, comme tous les matériaux présents dans la nature, contient à l'état de traces des radionucléides naturels d'origine terrestre. Par conséquent, la combustion du charbon conduit au rejet dans l'environnement de produits radioactifs et contribue à l'exposition humaine aux rayonnements ionisants.

L'évaluation de l'impact des rejets radioactifs des centrales au charbon a été évoquée pour la première fois en 1964 et a depuis fait l'objet de nombreuses études pour la plupart méthodologiques (voir par exemple les références de 1 à 4). Le travail présenté ici a un caractère plus concret puisqu'il étudie les conséquences des rejets d'une centrale réelle, implantée dans le Sud-Est de la France, et s'appuie autant que possible sur des mesures effectuées sur le matériau combustible, à l'intérieur de la centrale et dans son environnement.

Le découpage de l'étude est classique ; la description de la centrale et de son environnement précède l'estimation des rejets atmosphériques d'effluents radioactifs et de leurs conséquences, évaluées pour un groupe critique hypothétique et exprimées en équivalents de dose effectifs.

Ce document présente les premiers résultats de travaux, qui s'inscrivent dans le cadre d'une étude plus large menée sous contrat avec la Commission des Communautés Européennes et relative à l'évaluation comparative des risques auxquels sont soumis les individus d'une population régionale.

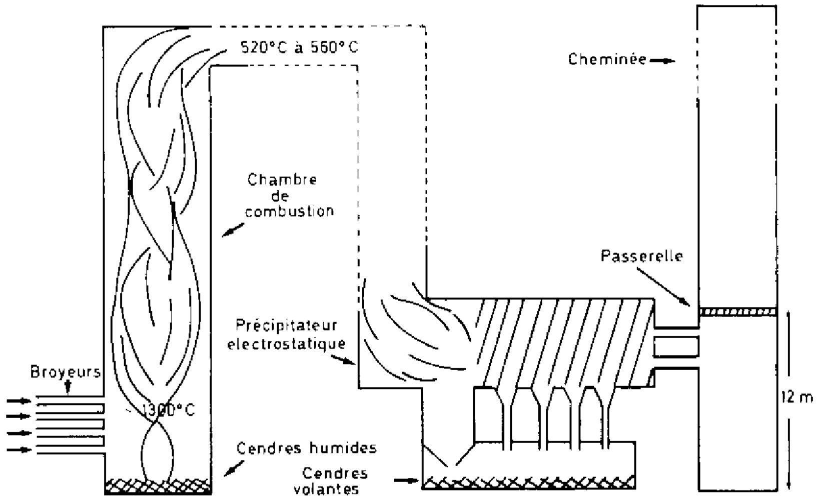

FIG.1. Schéma simplifié du fonctionnement de la centrale.

2. DESCRIPTION DE LA CENTRALE

La centrale thermique est construite sur le territoire de la commune de Gardanne à proximité immédiate du bassin d'où est issue toute la production de charbon nécessaire à son alimentation.

La figure 1 schématise de façon très simplifiée le fonctionnement de la centrale. La puissance totale électrique est de 415 MW répartie en 4 tranches : 3 groupes de 55 MW et un groupe plus récent de 250 MW.

Les effluents atmosphériques sont rejetés par 4 cheminées (trois de 65 m pour les groupes de 55 MW et une de 142 m pour celui de 250 MW). Le dépoussiérage des gaz est assuré par un filtrage électrostatique. Les cendres recueillies sont stockées dans plusieurs silos d'une capacité totale de 46 000 tonnes. Ces cendres sont utilisées par les cimenteries qui en apprécient les propriétés hydrauliques que leur confère leur teneur en chaux. Les cendres non utilisées sont stockées dans des terrils aux alentours de l'usine.

La consommation de charbon est de l'ordre de 800 tonnes par jour pour chacun des groupes de 55 MW et de 3 400 tonnes par jour pour celui de 250 MW. Ceci conduit à une consommation moyenne annuelle totale d'environ 1,3 $Mt \cdot a^{-1}$ de charbon pour une production électrique d'environ $2 \cdot 10^9$ kWh. Toutes ces observations sont regroupées dans le tableau I.

TABLEAU 1 - CARACTERISTIQUES DE LA CENTRALE THERMIQUE DE GARDANNE

N° tranche	Année de mise en service	Puissance	Cheminée		Gaz à la sortie			Quantité de charbon brûlé
			Hauteur	Diamètre à l'éjection	Température	Débit [a]	Humidité	
		MW	m	m	°C	$m^3 \cdot h^{-1}$	%	$g \cdot (kWh)^{-1}$
1	1953	55	65		125	25.10^4	11	780
2	1957	55	65		125	25.10^4	11	780
3	1958	55	65		125	25.10^4	11	780
4	1966	250	142	4,85	200	10^6	11	550
5	1984	600	300					

[a] ramené à la pression et température normales

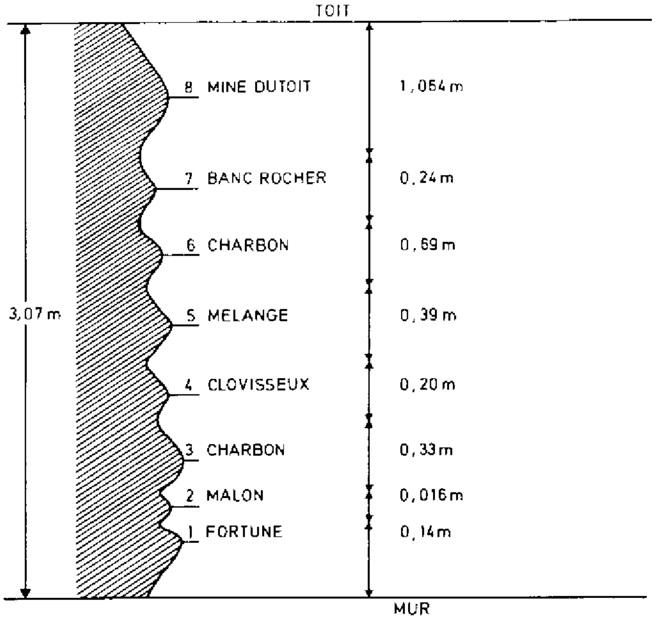

FIG.2. Coupe de la mine.

Le charbon brûlé à la centrale est du type lignite à pouvoir calorifique relativement bas et à forte teneur en cendres. La figure 2 présente une coupe de la mine au lieu dit l'Etoile ; on y distingue 8 couches de caractéristiques différentes. L'ensemble du minerai est envoyé vers les broyeurs puis la laverie. La gangue est séparée par densité de la fraction combustible, elle-même classée en deux qualités. La première qualité est vendue à l'extérieur des houillères, la deuxième bien inférieure constitue le combustible de la centrale proprement dit. On y trouve de 25 à 33 % de cendres et de 1,5 à 3,5 % de soufre.

Nous considèrerons deux types principaux de cendres :

- les cendres humides qui correspondent à la fraction incombustible la plus grossière récupérée au bas de la chaudière sous la grille réceptionnant le minerai à brûler ;
- les cendres volantes qui constituent la fraction incombustible la plus fine et qui sont entraînées par les gaz de combustion à une température de 520 à 570°C selon les foyers. La plus grande partie de ces cendres volantes sera retenue par les dépoussiéreurs. Le rejet atmosphérique sera alors constitué par les cendres volantes non retenues par les filtres.

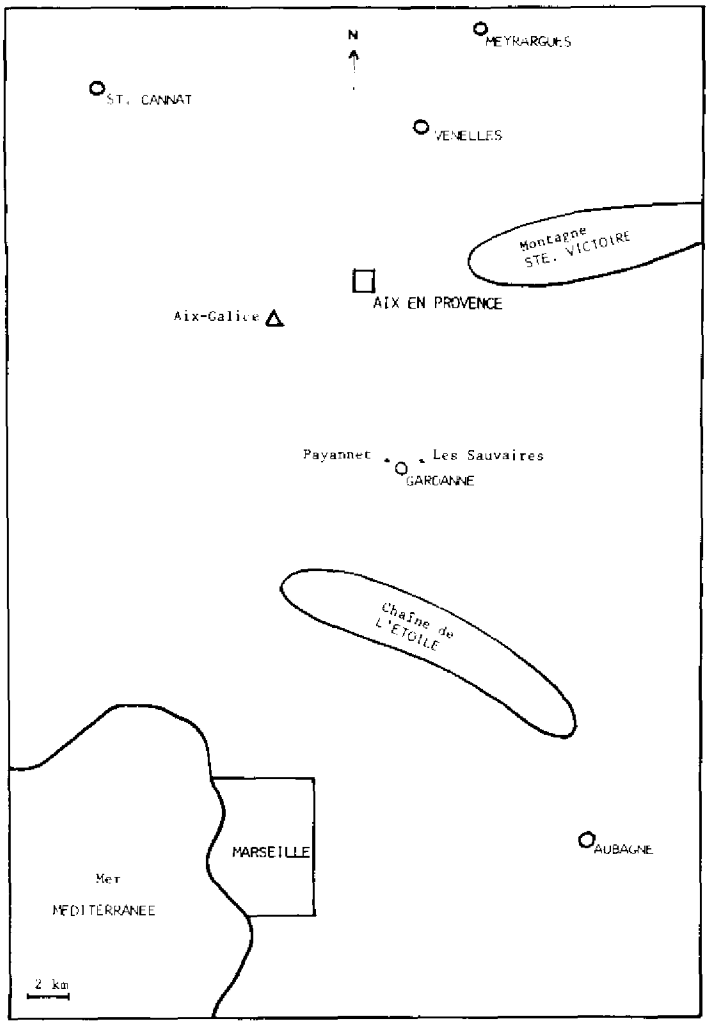

FIG.3. Site de Gardanne.

3. DESCRIPTION DE L'ENVIRONNEMENT DE LA CENTRALE

Gardanne est situé dans la partie Ouest des Bouches-du-Rhône à environ 20 km au Nord de Marseille, dans la plaine de l'Arc ; le relief est vallonné et limité au Nord-Ouest par la montagne Sainte Victoire et au Sud par le massif de l'Etoile (figure3).

Climatologie - Météorologie

Le climat de la région est de type méditerranéen. La température moyenne annuelle est élevée (15°C), la température maximale dépasse 25°C quatre vingt dix jours par an et le nombre de jours de gelées est faible (25 jours par an en moyenne).

Les précipitations sont de l'ordre de 620 $mm \cdot a^{-1}$ dans la plaine de l'Arc et de 750 $mm \cdot a^{-1}$ sur les reliefs avoisinants ; la hauteur d'eau moyenne décroît à 500 mm vers l'étang de Berre.

Les observations provenant d'une station météorologique proche indiquent une fréquence élevée de vents calmes (13,7 % de vents inférieurs à 1 $m \cdot s^{-1}$) et des vents dominants du Nord-Ouest et secondairement du Sud-Est. Il est possible que dans la plaine de l'Arc l'effet de vallée soit perceptible (orientation Ouest-Est).

Populations

Les alentours du site de Gardanne sont très peuplés mais de façon peu homogène en raison de la présence de zones urbanisées à forte concentration (Marseille : 2 millions d'habitants) coexistant avec des communes rurales peu peuplées. L'environnement proche reflète ces disparités. Si on excepte Aix-en-Provence et Marseille, les communes les plus peuplées sont : Gardanne (14 120 habitants), Allauch (11 149 habitants) et Septemes (10 827 habitants).

Agriculture

Gardanne se situe au Sud-Est de la région des côteaux de Provence. Du point de vue agricole, cette région est caractérisée par une polyculture où la vigne occupe une place importante, notamment sur les côteaux. Les exploitations sont en général de taille réduite, la plupart d'entre elles étant inférieures à un hectare. Les cultures sous serres sont largement pratiquées et compte tenu de la relative sécheresse estivale, l'irrigation est très développée.

Les productions animales concernent la production porcine, dont l'élevage tend à devenir semi-industriel, ovine et, près du littoral, bovine avec production laitière.

4. LES MESURES : RESULTATS ET DISCUSSION

Les mesures ont porté sur la radioactivité naturelle d'échantillons prélevés dans le combustible et les cendres (matières de base) ainsi que dans l'environnement de la centrale (air, sol, eau, produits végétaux et produits animaux). Les éléments mesurés sont : U-238, Ra-226, Th-232, Po-210, K-40. Dans certains cas, on a également mesuré la radioactivité artificielle.

Les poussières des cheminées ont été collectées sur filtre papier à l'aide d'une sonde isocinétique. Le débit était faible (1 $m^3 \cdot h^{-1}$) et la durée du prélèvement très courte (2 minutes).

TABLEAU II - CONCENTRATIONS EN RADIONUCLEIDES MESUREES DANS DU CHARBON DE DIFFERENTES COUCHES DE LA GALERIE ETOILE ET COMPARAISON AVEC QUELQUES DONNEES INTERNATIONALES

($Bq \cdot kg^{-1}$)

		K-40	Ra-226	U-238	Th-232
Fortune	(1)	-	-	-	-
Malon	(2)	1,5	15	11	< 4
Charbon	(3)	25	40	40	< 7
Clovisseux	(4)	20	40	40	3
Mélange	(5)	15	30	20	< 7
Charbon	(6)	7	40	40	< 4
Banc rocher	(7)	11	20	20	4
Mine du toît	(8)	1,5	15	11	< 4
Gamme de variation Etoile		1,5-25	15-40	11-40	3-7
Valeur moyenne adoptée		17	40	35	5
Production internationale		40-540	0,5-120	15-250	8-160
Moyenne Etats-Unis		52	-	18	21

Les poussières atmosphériques ont été collectées sur filtres ; les prélèvements ont duré une centaine d'heures avec un débit de 70 à 80 $m^3 \cdot h^{-1}$. Les filtres ont été fondus avec leurs poussières et amenés par pastillage à chaud à une dimension normalisée pour le porte-échantillon de la chaîne de mesure.

Les échantillons de charbon ont été broyés, séchés et amenés eux aussi aux dimensions compatibles avec la chaîne spectrométrique.

La plupart des mesures ont été effectuées par spectrométrie γ utilisant des détecteurs Ge-Li ; les mesures α (Po-210 essentiellement) ont consisté en un comptage total par un détecteur à scintillations.

Notons que certains échantillons ont dû être reconditionnés pour pouvoir être mesurés sur des chaînes différentes.

4.1. ANALYSE DES MATIERES DE BASE

Charbon

On a mesuré U-238, Ra-226, Th-232, K-40 dans les différentes couches (figure 2) du front de taille. Les résultats sont donnés dans le tableau II. La distribution des concentrations est assez

TABLEAU III - CONCENTRATIONS EN RADIONUCLEIDES MESUREES DANS LE CHARBON ET DIVERS TYPES DE CENDRES ET RAPPORTS DES CONCENTRATIONS CENDRES/CHARBON ($Bq \cdot kg^{-1}$)

	K-40	Ra-226	U-238	Th-232	Po-210
Cendres volantes					
Tranche 1	220	370	410	40	270
Tranche 4	-	520	430	-	-
Cendres compactes	220	370	370	30	230
Cendres foyers	110	280	270	25	-
Charbon	17	40	35	5	-
Cendres volantes/Charbon	13	9,2	11,7	8	-
Cendres compactes/Charbon	13	9,2	10,6	6	-
Cendres foyers/Charbon	6,5	7	7,7	5	-

homogène dans l'ensemble des couches à l'exception des couches extrêmes dans lesquelles les concentrations en K-40 et U-238 sont nettement plus faibles. Il convient de remarquer que le charbon utilisé dans la centrale est essentiellement celui qui provient des couches 3 à 6. Les teneurs moyennes en produits radioactifs naturels adoptées dans le tableau II sont les moyennes arithmétiques des concentrations mesurées dans ces 4 couches. On peut remarquer que les concentrations mesurées sont faibles par rapport aux valeurs habituellement rencontrées dans le monde. Les caractéristiques radioactives de ce charbon sont assez proches de celles de la moyenne de la production américaine. Le tableau II présente également quelques valeurs limites de la concentration en radionucléides de la production internationale de charbon.

Cendres

Le tableau III présente les résultats d'analyses effectuées sur les divers types de cendres. Comme nous l'avons indiqué précédemment, il existe deux sortes de cendres : les cendres volantes et les cendres humides, mais il existe aussi une variante des cendres volantes, celles que l'on retrouve dans les terrils où elles sont

TABLEAU IV - GRANULOMETRIE DES AEROSOLS PRELEVES DANS LA CHEMINEE DE LA TRANCHE 1

Dimensions (µm)	Pourcentage
< 1	4
1 - 2	9,5
2 - 3	17,7
3 - 5,5	19,6
5,5 - 9,2	17,4
> 9,2	31,6

rejetées après abondante humidification au canon à eau, opération qui limite la remise en suspension par érosion éolienne en accroissant la vitesse de sédimentation et en donnant une certaine cohésion au milieu. Dans le tableau III ce type de cendres sera appelé cendres compactes.

La teneur en cendres du charbon utilisé dans la centrale variant de 25 à 33 % on pourrait s'attendre à ce que les activités massiques dans les cendres soient supérieures aux activités massiques dans le charbon d'un facteur 3 à 4. Les résultats présentés au tableau III montrent que les rapports entre les concentrations des différents radionucléides dans les cendres et dans le charbon sont sensiblement plus élevés puisqu'ils varient entre 5 et 13.

Cette différence peut s'expliquer par l'échantillonnage, les cendres prélevées pouvant provenir d'un lot de charbon à plus faible teneur en cendres. Pour ce qui est des cendres volantes, les rapports plus élevés sont en partie dus à une granulométrie plus fine (tableau IV).

Estimation des rejets dans l'atmosphère

Les activités rejetées dans l'atmosphère ont été évaluées à partir des activités mesurées dans les cendres volantes. En effet, connaissant la concentration C des poussières dans les gaz et les débits D de rejet, il est aisé d'en déduire l'activité R rejetée par MWe produit. Celle-ci s'écrit :

$$R = \frac{A \;.\; C \;.\; D}{P} \quad k_u$$

TABLEAU V - DEBITS DE REJETS ESTIMES EN $kBq\cdot(MW.a)^{-1}$

	K-40	Ra-226	U-238	Th-232
Tranche 1	4.10^3	$6,8.10^3$	$7,5.10^3$	$7,3.10^2$
Tranche 4	$3,1.10^3$	$7,3.10^3$	6.10^3	$5,6.10^2$
Centrale	$3,5.10^3$	$7,1.10^3$	$6,6.10^3$	$6,3.10^2$

où R est exprimé en $kBq\cdot(MWe\cdot a)^{-1}$

A en $Bq\cdot kg^{-1}$,

C en $g\cdot m^{-3}$

D en $m^3\cdot s^{-1}$

P qui représente la puissance de la centrale, en MWe

k_u est un coefficient de conversion d'unité : il est égal à $31,5\ Ms\cdot a^{-1}$.

Pour les tranches 1 + 2 + 3, le taux de poussières est de 0,46 $g\cdot m^{-3}$; le débit de rejets gazeux et la puissance sont indiqués au tableau I. On obtient :

$$R_1 = C \times 0,46 \times \frac{75\ 10^4}{3600 \times 165} \times 31,5 = 18,3\ C$$

Pour la tranche 4, le taux de poussières est de 0,40 $g\cdot m^{-3}$. On obtient :

$$R_4 = C \times 0,40 \times \frac{10^6}{3600 \times 250} \times 31,5 = 13,9\ C$$

A partir des valeurs adoptées de C, on obtient les rejets estimés présentés au tableau V.

4.2. MESURES DANS L'ENVIRONNEMENT

Air

Deux sites de prélèvements ont été choisis, compte tenu de la rose des vents, à environ 1 km des points de rejets (figure 3) : les Sauvaires à l'Est de la centrale et le Payannet à l'Ouest. On y a installé des aspirateurs assurant un débit de 70 à 80 $m^3\cdot h^{-1}$ avec collection de poussières sur filtre microsorban.

On a mesuré les radionucléides naturels ainsi que les produits de fission (tableau VI). Ces derniers sont de l'ordre de grandeur

TABLEAU VI - CONCENTRATIONS ATMOSPHERIQUES MESUREES AU PAYANNET ET AUX SAUVAIRES ($mBq \cdot m^{-3}$)

Radionucléide	Le Payannet	Les Sauvaires
Be-7	6,3 ± 0,7	6,3 ± 0,7
Zr-95	0,67 ± 0,18	0,85 ± 0,11
Nb-95	1,2 ± 0,2	1,5 ± 0,1
Ru-103	0,81 ± 0,07	1,1 ± 0,1
Cs-137	0,07 ± 0,04	0,04 ± 0,02
Ce-141	0,67 ± 0,07	1,5 ± 0,2
Ce-144	0,15 ± 0,04	⩽ 0,04
K-40	3,7 ± 1,1	1,5 ± 0,5
U-238	0,52 ± 0,18	ND
Ra-226	0,67 ± 0,18	ND
Th-232	0,26 ± 0,15	0,11 ± 0,07

ND = non détecté

de ce que l'on trouve habituellement : par contre, les concentrations en U-238, Ra-226, Th-232, K-40 sont une centaine de fois plus élevée que les valeurs moyennes du bruit de fond ce qui indique une influence certaine de la centrale.

Eau

Deux pluviomètres avaient été installés dans l'enceinte de la centrale mais la trop faible pluviosité n'a pas permis de récupérer suffisamment d'eau pour la mesure. Par contre, les concentrations en radionucléides d'un échantillon d'eau prélevé dans un canal d'irrigation proche de la centrale (canal de Provence, projet Rigaud) ont pu être mesurées.

Les résultats sont les suivants :

Ra-226	$1,26.10^{-2} \pm 1,1.10^{-3}$	$Bq \cdot l^{-1}$
Th-232	$2,6 .10^{-3} \pm 6,3.10^{-4}$	$Bq \cdot l^{-1}$
U-238	$4,8 .10^{-3}$	$Bq \cdot l^{-1}$

Sols

Des prélèvements de sol ont été effectués en deux points situés à proximité des deux stations de prélèvements de poussières atmosphériques. La répartition de l'activité en fonction de la profondeur est présentée pour l'un d'entre eux dans le tableau ci-dessous : il s'agit d'un sol de vignoble, non irrigué, situé à proximité d'un terril

Profondeur (cm)	Concentration ($Bq \cdot kg^{-1}$)			
	Cs-137	Ra-226	Th-232	U-238
0 - 5	3,7	16	7,4	37
5 - 10	5,5	15	5,2	37
10 - 15	2,6	7,8	3,3	37

Légumes

Des prélèvements de légumes ont été effectués dans le secteur Ouest du Payannet, près de la zone où ont été prélevées des poussières atmosphériques. Bien qu'il existe de nombreuses fermes dans ce secteur, le choix des échantillons a été limité en raison de la saison. Les résultats obtenus, exprimés en $Bq \cdot kg^{-1}$ frais, sont les suivants :

Element	Be-7	Zr-95	Cs-137	K-40	U-238	Ra-226	Th-232
Salade	5,6	0,43	0,62	95,5	0,28	0,14	6
Pomme de terre	_	_	1,3	67	0,85	0,85	
Carotte	0,2	_	$3,7.10^{-3}$	0,96	$2,2.10^{-2}$	$3,4.10^{-2}$	$2,7.10^{-2}$

Bioindicateurs

De nombreux auteurs ont recommandé les aiguilles de pin comme des indicateurs biologiques de la radioactivité de l'air à cause de leur facteur de concentration élevé et de la proportion existant entre l'activité par gramme de cendres et l'activité déposée. L'étude de la cinétique de fixation des aiguilles de pin (émetteurs β et α) a montré qu'elle était lente, autrement dit qu'elles ne sont pas utilisables dans le cas de rejets accidentels mais conviennent très bien pour les faibles variations d'activités en fonctionnement normal.

Dans cette même station, des feuilles de chênes verts ont été recueillies à cause d'un dépôt blanchâtre particulièrement abondant. Ces dépôts sont en relation avec l'absorption au niveau des aiguilles de pin.

Les analyses de ces échantillons ont fourni les résultats suivants :

	Cs-137	Ra-226	Th-232	U-238
Aiguilles de pin ($Bq \cdot kg^{-1}$ frais)	0,02	0,02	0,09	-
Feuilles de chênes ($Bq \cdot kg^{-1}$ sec)	-	-	-	6,3
Crottes de lapin ($Bq \cdot kg^{-1}$ sec)				
Payannet				11
Témoin				< 0,11

5. EVALUATION DES CONSEQUENCES DES REJETS ATMOSPHERIQUES

Les conséquences des rejets atmosphériques ont été évaluées pour les individus du groupe critique. Les radionucléides considérés sont ceux des familles de l'uranium 238 et du thorium 232, dans lesquelles tous les descendants sont supposés être en équilibre avec leur précurseur à l'exception de radon 222 et de radon 220. Les rejets de radon 222 sont estimés à 3 $kBq \cdot s^{-1}$ sur la base de la teneur du charbon en radium 226 et de la quantité de charbon brûlé. Les rejets de radon 220 ont été négligés. Les estimations des conséquences des rejets tiennent compte de trois voies de transfert :

. l'inhalation,
. l'ingestion de légumes feuilles et de légumes racines,
. l'irradiation externe.

Elles sont effectuées lors de la dernière année de l'exploitation de la centrale (durée de fonctionnement adoptée : 30 ans) afin d'obtenir les expositions maximales, et sont exprimées en équivalents de dose effectifs annuels dans le but de permettre une comparaison aisée de l'importance relative des divers radionucléides et des voies de transfert.

De plus, étant donné que certains des radionucléides considérés ont des périodes très longues et s'accumulent dans le sol, il a paru intéressant d'évaluer également les équivalents de dose effectifs reçus annuellement par les individus du groupe critique après arrêt de la centrale.

5.1. INHALATION PENDANT LA PERIODE DE FONCTIONNEMENT

Le modèle de dispersion atmosphérique de DOURY [5] a été appliqué aux rejets estimés pour calculer le champ des moyennes annuelles des concentrations atmosphériques autour de la centrale. La zone de concentration maximale est ainsi trouvée à environ 300 m au Sud-Est du point de rejet.

Afin de déterminer les conditions d'utilisation du modèle de dispersion, celui-ci avait auparavant été appliqué à l'évaluation des concentrations atmosphériques au Payannet et aux Sauvaires lors de la campagne de mesures. La comparaison entre les concentrations mesurées et les concentrations calculées a montré qu'il était inutile de tenir compte d'une surélévation du panache au point de rejet et que les conditions de diffusion correspondant à une atmosphère instable étaient plus appropriées que celles correspondant à une atmosphère stable. Néanmoins, le rapport entre les concentrations calculées et les concentrations mesurées est au mieux égal à 3, les concentrations calculées étant systématiquement inférieures aux concentrations mesurées. Parmi les raisons pouvant expliquer cette divergence, on peut citer les difficultés d'application d'un modèle de dispersion atmosphérique dans un site à relief tourmenté et la présence de sources de pollution voisines (usine de bauxite) qui pourraient contribuer aux concentrations mesurées.

Les moyennes annuelles des concentrations atmosphériques calculées dans la zone d'exposition maximale sont présentées au tableau VII et comparées à des concentrations naturelles typiques [1]. Les concentrations calculées sont nettement supérieures aux concentrations naturelles pour les radionucléides compris entre U-238 et Ra-226, nettement inférieures pour Rn-222, Pb-210 et Po-210, et voisines pour les autres radionucléides.

Les équivalents de dose effectifs reçus annuellement par inhalation par les individus du groupe critique ont été estimés à partir des données de la CIPR [6] en supposant que les concentrations atmosphériques étaient identiques à l'intérieur et à l'extérieur des habitations. Le tableau VIII montre les résultats obtenus.

5.2. INGESTION PENDANT LA PERIODE DE FONCTIONNEMENT

On a supposé que les individus du groupe critique se nourrissaient de produits cultivés localement. Seuls les légumes feuilles (taux de consommation : 140 $kg \cdot a^{-1}$) et les légumes racines (taux de consommation : 110 $kg \cdot a^{-1}$) ont été pris en considération.

L'activité déposée au sol a été évaluée en prenant une vitesse de dépôt sec de $5 \cdot 10^{-3}$ $m \cdot s^{-1}$. Le dépôt humide a été négligé. La fraction du débit de dépôt retenue par les feuilles des légumes

TABLEAU VII- COMPARAISON DES CONCENTRATIONS CALCULEES DANS LA ZONE D'EXPOSITION MAXIMALE ET DE CONCENTRATIONS NATURELLES TYPIQUES

Radionucléide	Concentrations atmosphériques ($\mu Bq \cdot m^{-3}$)		Concentrations dans les sols ($Bq.kg^{-1}$)	
	Calculées	Naturelles	Calculées	Naturelles
K-40	21	40	0,20	370
U-238	37	3	0,35	25
Ra-226	37	3	0,35	25
Rn-222	1300	4 000 000	-	-
Pb-210	37	500	0,35	30
Po-210	37	100	0,35	30
Th-232	3,7	3	0,035	25
Ra-228	3,7	3	0,035	25
Th-228	3,7	3	0,035	25

a été prise égale à 0,25 ; on a admis de plus que la période de rétention sur les feuilles était de 30 jours et que les légumes feuilles étaient récoltés six mois par an.

En ce qui concerne l'activité transmise aux légumes par voie racinaire, celle-ci a été établie en tenant compte d'un temps d'accumulation dans les sols de 30 ans et en utilisant des facteurs de transfert publiés [7-9].

Les équivalents de dose effectifs correspondant à l'ingestion de légumes sont présentés au tableau VIII.

5.3. IRRADIATION EXTERNE PENDANT LA PERIODE DE FONCTIONNEMENT

Pour calculer l'irradiation externe délivrée aux individus du groupe critique du fait de l'activité contenue dans le sol lors de la dernière années d'exploitation, on a supposé que l'activité déposée restait en surface pendant un an puis était mélangée de manière homogène sur une épaisseur de 30 cm. Les doses d'irradiation externe présentent ainsi deux composantes, qui sont calculées de manière indépendante [10]. Le tableau VIII présente les résultats obtenus.

Au sujet des concentrations dans les sols à la fin de la période d'exploitation, il faut noter que celles-ci sont, pour tous les radionucléides considérés, très inférieures aux concentrations naturelles typiques (tableau VII).

TABLEAU VIII - ESTIMATION DES EQUIVALENTS DE DOSE EFFECTIFS RECUS PAR LE GROUPE CRITIQUE LORS DE LA 30ème ANNEE D'EXPLOITATION DE LA CENTRALE ($Sv.a^{-1}$)

Radionucléide	Inhalation	Ingestion Légumes-feuilles	Ingestion Légumes-racines	Irradiation externe
U-238	$9{,}5.10^{-6}$	$2{,}5.10^{-7}$	$2{,}4.10^{-9}$	
Th-234	$2{,}6.10^{-9}$	$1{,}4.10^{-8}$	$1{,}3.10^{-10}$	
U-234	$1{,}1.10^{-5}$	$2{,}8.10^{-7}$	$2{,}7.10^{-9}$	
Th-230	$2{,}1.10^{-5}$	$5{,}7.10^{-7}$	$5{,}6.10^{-10}$	
Ra-226	$6{,}3.10^{-7}$	$1{,}5.10^{-6}$	$9{,}7.10^{-8}$	$5{,}3.10^{-7}$
Rn-222	$8{,}8.10^{-9}$	-	-	
Pb-210	$1{,}0.10^{-6}$	$6{,}5.10^{-6}$	$3{,}6.10^{-6}$	
Bi-210	$1{,}5.10^{-8}$	$1{,}2.10^{-8}$	$4{,}3.10^{-9}$	
Po-210	$6{,}3.10^{-7}$	$3{,}2.10^{-6}$	$1{,}2.10^{-6}$	
Th-232	$9{,}2.10^{-6}$	$2{,}9.10^{-7}$	$2{,}8.10^{-10}$	
Ra-228	$3{,}4.10^{-8}$	$1{,}6.10^{-7}$	$1{,}1.10^{-8}$	$8{,}0.10^{-8}$
Th-228	$2{,}5.10^{-6}$	$4{,}8.10^{-8}$	$3{,}2.10^{-9}$	
Ra-224	$2{,}3.10^{-8}$	$4{,}2.10^{-8}$	$2{,}8.10^{-9}$	
TOTAL	$5{,}5.10^{-5}$	$1{,}3.10^{-5}$	$4{,}9.10^{-6}$	$6{,}1.10^{-7}$

5.4. RECAPITULATION

L'examen du tableau VIII montre qu'avec les modèles utilisés, c'est l'inhalation qui est la voie de transfert prépondérante pendant la période de fonctionnement de la centrale et que les radionucléides les plus importants sont Th-230, U-234, U-238 et Th-232. La dose reçue par ingestion serait environ trois fois plus faible que celle reçue par inhalation tandis que l'irradiation externe serait négligeable. L'équivalent de dose effectif total délivré en une année serait d'environ $7 \cdot 10^{-5}$ Sv, ce qui est très inférieur aux limites recommandées par la CIPR.

TABLEAU IX - ESTIMATION DES EQUIVALENTS DE DOSE EFFECTIFS RECUS PAR LE GROUPE CRITIQUE APRES L'ARRET DE LA CENTRALE ($Sv.a^{-1}$)

Radionucléide	Inhalation	Ingestion Légumes-feuilles	Ingestion Légumes-racines	Irradiation externe
U-238	$4,5.10^{-9}$	$4,3.10^{-10}$	$2,4.10^{-9}$	$4,2.10^{-7}$
Th-234	$1,2.10^{-12}$	$2,4.10^{-11}$	$1,3.10^{-10}$	
U-234	$5,2.10^{-9}$	$4,8.10^{-10}$	$2,7.10^{-9}$	
Th-230	$9,9.10^{-9}$	$3,4.10^{-10}$	$5,6.10^{-10}$	
Ra-226	$3,0.10^{-10}$	$2,6.10^{-7}$	$9,7.10^{-8}$	
Rn-222	-	-	-	
Pb-210	$4,7.10^{-10}$	3.10^{-6}	$3,6.10^{-6}$	
Bi-210	$7,1.10^{-12}$	$5,5.10^{-9}$	$4,3.10^{-9}$	
Po-210	$3,0.10^{-10}$	$1,5.10^{-6}$	$1,2.10^{-6}$	
Th-232	$4,4.10^{-9}$	$1,8.10^{-10}$	$2,8.10^{-10}$	$6,4.10^{-8}$
Ra-228	$1,6.10^{-11}$	$2,8.10^{-8}$	$1,1.10^{-8}$	
Th-228	$1,2.10^{-9}$	$8,5.10^{-9}$	$3,2.10^{-9}$	
Ra-224	$1,1.10^{-11}$	$7,4.10^{-9}$	$2,8.10^{-9}$	
TOTAL	$2,6.10^{-8}$	$4,8.10^{-6}$	$4,9.10^{-6}$	$4,8.10^{-7}$

5.5. DOSES REÇUES APRES L'ARRET DE LA CENTRALE

Etant donné que certains des radionucléides considérés ont des périodes radioactives très longues, les individus du groupe critique continueront d'être exposés longtemps après l'arrêt de la centrale en raison de l'accumulation de ces radionucléides dans les sols et de leur lente élimination.

L'évaluation des équivalents de dose effectifs annuels correspondants a été faite en supposant :

- pour l'inhalation, que la teneur de l'air en poussières provenant du sol était de 50 $\mu g \cdot m^{-3}$;
- pour l'irradiation externe, que la composante due au dépôt en surface avait disparu.

Les résultats sont présentés au tableau IX. Ils montrent une prédominance de l'ingestion sur l'irradiation externe et l'inhalation. L'équivalent de dose effectif total reçu en une année est évalué à 10^{-5} Sv environ.

6. CONCLUSION

L'analyse des résultats bruts obtenus au cours de cette première étude consacrée aux rejets atmosphériques d'éléments radioactifs naturels par une centrale électrique de 415 MWe fonctionnant au charbon montre que l'exposition maximale résultante pour les personnes du public se situe aux environs du pourcent des limites recommandées pour ces personnes par la Commission Internationale de Protection Radiologique : elle est donc du même ordre de grandeur que celle que l'on évalue généralement dans le cas d'une centrale électro-nucléaire de 900 MWe.

En dehors des incertitudes qui s'attachent aux évaluations présentées et des hypothèses conservatives prises, cette comparaison n'est en fait que très partielle. D'une part elle ne tient pas compte de l'exposition entraînée par les autres étapes du cycle du combustible : extraction et traitement du minerai, préparation du combustible, traitement des combustibles nucléaires irradiés, stockage des déchets.

D'autre part, elle ne concerne que les conséquences des rejets radioactifs alors que les rejets de polluants conventionnels sont susceptibles de créer d'autres nuisances, notamment dans le cas de l'utilisation du charbon. Enfin, les résultats obtenus doivent être interprétés en tenant compte de la nature différente des radionucléides en cause.

Comme celà a été indiqué, cette étude a constitué une première approche d'un programme beaucoup plus large englobant d'autres activités industrielles qu'elles soient ou non génératrices d'énergie.

Elle a permis de cerner les difficultés susceptibles d'être rencontrées, de mettre au point les techniques de mesures et de définir une méthodologie qui sera plus largement développée dans l'avenir.

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Nous tenons à remercier la Direction de la Centrale de Gardanne et en particulier M. FILLIOL, de nous avoir permis d'effectuer des prélèvements à l'intérieur de l'établissement et de nous avoir communiqué des informations sur le fonctionnement de la Centrale.

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DISCUSSION

B.L. TRACY: The results you presented for emissions and ground-level concentrations are higher than values reported recently in North America, and I should like to know the reason for this difference. What is the collection efficiency of the electrostatic dust collectors? Also, did you use theoretical predictions or experimental measurements of field concentrations as input to your dose calculations?

R. COULON: The difference to which you refer is very probably due to the fact that the power plant studied is relatively old and that the technology used in it is less sophisticated than that of more modern plants. The efficiency of the electrostatic dust precipitator was about 95%.

With regard to your second question, public exposure was assessed not on the basis of concentrations measured in the environment but by means of concentrations calculated on the basis of the activities released, after applying dispersion and transfer models.

COMPARISON OF RADIATION EXPOSURE FROM COAL-FIRED AND NUCLEAR POWER PLANTS IN THE FEDERAL REPUBLIC OF GERMANY

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Abstract

COMPARISON OF RADIATION EXPOSURE FROM COAL-FIRED AND NUCLEAR POWER PLANTS IN THE FEDERAL REPUBLIC OF GERMANY.

Several studies have been published in recent years on radiation exposure of the public from coal-fired power plants in comparison with nuclear power plants. The results differ remarkably, mainly owing to the different assumptions about the discharge rate and the models and parameters necessary for such calculations. A comprehensive study of radioactive emissions from two typical modern coal-fired plants was performed in the Federal Republic of Germany in 1979/80. Measurements included the specific activities of all relevant radionuclides in coal and fly-ash. On the basis of these data the possible radiation exposure of the public by inhalation and ingestion was calculated and compared with the results for a modern PWR and BWR. The models and parameters of the regulatory guide for calculating radiation exposure by emission of radionuclides from nuclear power plants in the FRG were used, but emphasis was placed on applying realistic assumptions or parameters where justifiable. In particular, results with dose factors of ICRP 2 were compared with those of ICRP 26. The results indicate for the point of maximum exposure an effective committed dose equivalent in the range of $0.1-1$ $mrem \cdot a^{-1}$ per $GW(e) \cdot a$ electric energy generated, for the modern nuclear power plants under consideration as well as for the pit-coal-fired plant. In the case of coal-fired plants, the dose to the lungs and bone surfaces predominate whereas, for nuclear power plants, the main contributions are external γ-radiation from the cloud and the dose to the thyroid from radioiodine. It is concluded that the radiation exposure to the public from all these types of power plants is small compared with the variation of normal natural radiation exposure and corresponds to less than 1% of the average natural radiation exposure.

1. INTRODUCTION

The operation of coal-fired and nuclear power plants results in radiation exposure of the public. For the former it is caused by natural radionuclides;

for the latter by newly created nuclides. With both types of power plant, the first concern with regard to radiation exposure of the population in the vicinity is the operational discharge of airborne radioactive effluents.

In the past decade, general interest has focussed on releases of artificial radioactive substances from the new nuclear power plants. The public is concerned about possible health effects of the artificial radionuclides.

Following the growing awareness that all impacts on the environment and public health must be examined, comparison studies of radiation exposure of the public from coal-fired power plants and nuclear power plants have been undertaken and discussed in recent years. These studies were soon countered by the ethical argument that one case cannot be justified by favourable comparison with another. This is even more true if the comparison is incomplete, e.g. with regard to other important environmental, occupational and public health effects.

This paper is restricted to a comparison of radiation exposure of the neighbouring population as the result of the operational discharge of airborne radioactive effluents from coal-fired and nuclear power plants. The calculations were carried out for a chosen site with the same models and parameters, as far as possible, taking into account the characteristics of modern types of coal-fired and nuclear power plants now operating in the FRG.

2. AIRBORNE RADIOACTIVE EFFLUENTS FROM COAL-FIRED POWER PLANTS

2.1. Investigations in the Federal Republic of Germany

The 1977 report of the United Nations Scientific Committee on the Effects of Atomic Radiation (UNSCEAR) [1] summarizes published data on the specific activity of natural radionuclides in coal and coal residues, including data from the FRG. It refers also to partitioning the elements in coal between bottom-ash, fly-ash and volatile vapours and their discharge into the atmosphere depending, for example, on the efficiency of the effluent-treatment systems employed.

Since then, a number of new investigations have been carried out. In the FRG, during the years 1978 1980, a measuring programme of natural radionuclides in coal and fly-ash from two modern coal-fired power plants, sponsored by the Federal Ministry of the Interior, was undertaken in co-operation with the TÜV-Rheinland, the Association for Radiation and Environmental Research, and the Federal Health Office, Neuherberg.

In particular, at the request of the Federal Ministry of the Interior, the German Radiation Protection Committee formed a working group to co-ordinate and evaluate the study. The working group particularly recommended that measurements of radionuclide composition and specific activity of coal and fly-ash discharged and particle size distribution should be carried out for this study.

TABLE I. AVERAGE SPECIFIC ACTIVITY OF PIT COAL, BROWN COAL AND FLY-ASH FROM COAL-FIRED PLANTS

Radionuclide	Average specific activity ($nCi \cdot kg^{-1}$)			
	Pit-coal-fired plant		Brown-coal-fired plant	
	Coal	Fly-ash	Coal	Fly-ash
U-238 (U-234, Th-230)	<1	8	0.4	3
Ra-226	0.5	8	<0.3	2
Pb-210	0.7	80	0.3	5
Po-210	0.8	150	0.3	8
Th-232 (Th-228)	<0.5	3	<0.2	1

Two modern coal-fired plants were selected, one using pit coal, the other brown coal:

(1) A 320-MW(e) pit-coal-fired plant (built in 1970); burning temperature 1600–1800°C; two serial electrostatic filters; 970 000 $m^3 \cdot h^{-1}$ off-gas; residual dust load 60 $mg \cdot m^{-3}$; stack 195 m; thermal lift to 300 m.

(2) A 600-MW(e) brown-coal-fired plant (built in 1974); burning temperature 1100°C; three serial electrostatic filters; 2 720 000 $m^3 \cdot h^{-1}$ off-gas; residual dust load 30 $mg \cdot m^{-3}$; stack 160 m; thermal lift to 200 m.

Measurements of the specific activity of the radionuclides included coal samples, ash deposited in the serial electrostatic filters,and fly-ash samples prior to discharge. The details of these investigations and measurements are described in recent reports [2–5]. Table I gives the average specific activity of the natural radionuclides of the uranium and thorium series in coal and fly-ash measured for the power plants fired by pit coal and by brown coal.

2.2. Discussion of the results

In comparison with the specific activity of the individual radionuclides in the coal, an enrichment in the fly-ash is observed, depending on the nuclide considered and the temperature during burning. The specific activity of individual particles of the fly-ash increases with decreasing particle size. Therefore, in the serial electrostatic filters the contribution of particles with higher specific activity increases up to the last filter stages. Finally, the specific activity of the fly-ash discharged through the stack will be about the same as that of the particles precipitated in the last filter stage.

The principal indications of the measurements were:

(a) For the pit-coal-fired plant (burning temperature 1600–1800°C): the specific activity of the discharged fly-ash has increased by a factor of about 10 for ^{226}Ra, ^{232}Th and ^{238}U, by a factor of 100 for ^{210}Pb and of 200 for ^{210}Po, compared to the specific activity of the coal-fired plant.

(b) For the brown-coal-fired plant (burning temperature 1100°C): the specific activity of the discharged fly-ash has increased by a factor of 3 to 5 for ^{226}Ra, ^{232}Th and ^{238}U and by a factor of 10 for ^{210}Pb and ^{210}Po.

It can be seen that the more volatile radionuclides ^{210}Pb and ^{210}Po show the greatest increase, whereas the main difference between the two types of plants is due to the different burning temperature. The investigations of the particle size distribution of the fly-ash indicated a nearly log-normal particle size distribution with an activity median aerodynamic diameter (AMAD) of 3 to 5 μm for uranium, thorium and radium particulates and of about 1 μm for ^{210}Pb and ^{210}Po. From these measurements we have calculated the emission of the individual radionuclides into the atmosphere, normalized for a power generation of 1 GW(e)·a for a plant of this performance. The data are summarized in Table II. The lower emissions from brown-coal-fired plants should be noted. The emission of ^{222}Rn is calculated as 1 to 2 Ci per GW(e)·a.[1]

3. AIRBORNE RADIOACTIVE EFFLUENTS FROM NUCLEAR POWER PLANTS

3.1. Radionuclide composition and discharge rates

Radionuclide composition and discharge rates of airborne effluents from nuclear power plants are measured as a matter of routine. The results have been published, e.g. in the annual report on Environmental Radioactivity and Radiation Exposure of the Federal Ministry of the Interior. All the data are listed in detail in the emission register of the Federal Health Office.

The primary composition of the radionuclides from nuclear power plants is, of course, that of the fission process. But activation products also arise, and the influence of the plant design and operation conditions has to be taken into account.

[1] 1 Ci = 3.7×10^{10} Bq.

TABLE II. EMISSION OF INDIVIDUAL RADIONUCLIDES INTO THE ATMOSPHERE FROM A COAL-FIRED POWER PLANT WITH POWER GENERATION OF 1 GW(e)·a

Radionuclide	Pit-coal plant (mCi per GW(e)·a)	Brown-coal plant (mCi per GW(e)·a)
U-238	10	3
U-234	10	3
Th-230	10	2
Ra-226	10	2
Pb-210	100	5
Po-210	200	10
Th-232	5	1
Th-228	5	1

There are other factors, such as technical improvements or more stringent licensing conditions for newer plants, which lead to a great variety of observed or expected annual discharges of radioactive airborne effluents.

The working group therefore recommended basing the comparison of coal-fired plants with the nuclear power plants on a large modern pressurized-water reactor (PWR) and a boiling-water reactor (BWR) built during the last decade. It would then be possible to evaluate the operating experience of both plants for some years and thereby to develop average discharge rates and nuclide compositions for modern PWRs and BWRs now operating in the FRG.

Some difficulties are inherent in such a procedure: the operating history in the first years of newly designed plants may be erratic; prolonged operation could give rise to different discharge rates or radionuclide composition. Therefore, only the newer power plants Biblis and Neckarwestheim were chosen for determining an average discharge rate and nuclide composition of airborne radioactive effluents from PWRs, and the Brunsbüttel and Isar plants were chosen for the BWRs. Finally, the annual emissions of these power plants were standardized to a power generation of 1 GW(e)·a.

A total of ten years of reactor operation were evaluated (Biblis Block A and Block B: 1976–1979; Neckarwestheim: 1977–1979; Brunsbüttel: 1977;

TABLE III. AVERAGE EMISSION RATE OF NOBLE GAS RADIONUCLIDES FROM NUCLEAR POWER PLANTS

Nuclide	Average PWR (Ci per GW(e)·a)	Average BWR (Ci per GW(e)·a)
Ar-41	30	–
Kr-85m	68	50
Kr-85	–	50
Kr-87	12	25
Kr-88	16	25
Kr-89	20	175
Xe-131m	2	50
Xe-133m	58	4
Xe-133	1100	4000
Xe-135m	69	68
Xe-135	120	350
Xe-137	10	200
Xe-138	4	125

Isar: 1978 and 1979). Table III shows the average values for noble gases in the effluent air of each of the nuclear power plants. If annual emission rates were not reported for one or more of the noble gas nuclides for any of the operational years in question (e.g. as ^{85}Kr with all PWRs), the emission rates were calculated as 0. Because of the low limit of detection for the measurement of individual nuclides of the noble gases of approximately 10^{-8} $Ci \cdot m^{-3}$, and because of the low annual emission rates of radioactive gases during the evaluated years, this procedure seemed justified. In determining average emission rates for noble gases from nuclear power plants with BWRs, the emissions from the Brunsbüttel power plant for 1978 were omitted since the plant discontinued operation in the middle of that year owing to a malfunction incident and remained out of operation for all of 1979 [6].

The annual average emission rates of the remaining radioactive substances were determined for the radioactive aerosols, ^{131}I and ^{133}I, ^{3}H and ^{14}C, and the same years were evaluated as those for the average emission rates for noble gases. The results are given in Table IV.

4. CALCULATION OF RADIATION EXPOSURE

4.1. Methods and parameters for the calculation

Calculation of radiation exposure of the population caused by airborne radioactive effluents starts with selection of the model for atmospheric dispersion, deposition and accumulation on the ground and transfer into food via ecological pathways. Then, the individual parameters applicable to the site, its surroundings and the population have to be determined. Today, recommended guidelines and standard parameters may be used for this purpose.

In our study for the comparison of coal-fired and nuclear power plants, the site data from the Biblis PWRs were used and radiation exposure was calculated following our regulatory guide for a reference person assumed to stay the whole year at the place of maximum exposure outside the fence and obtaining all food from the same place. One important deviation, however, was that,for the coal-fired power plants,we considered that only 20% of the radionuclides incorporated in the glassy fly-ash particulates would be soluble and thus be available for exposure via ingestion [7].

For calculating the radiation exposure by inhalation and ingestion, the dose factors of our regulatory guide (which is based on ICRP No.2) as well as the new recommended dose factors for effective dose equivalent (which are based on ICRP No.26) have been used. For the coal-fired power plants we also estimated the exposure by a comparison with the natural level of those radionuclides in the environment and in human tissues. The decision on the more important parameters to be used was made after discussion in detail at the working group meetings.

4.2. Results of the calculation

4.2.1. Nuclear power plant

Source terms as given in Table III and IV; site data for a 100-m stack at Biblis: atmospheric dispersion factor $3 \times 10^{-7}\ s \cdot m^{-3}$ at 500 m and for γ-submersion $1.3 \times 10^{-3}\ s \cdot m^{-2}$ at 100m.

The calculation included the exposure pathways: submersion, ground deposition, inhalation and ingestion of vegetables, crops, meat and milk. The results are summarized in Table V for whole-body exposure and the effective dose equivalent. The dose to individual organs of reference man may be up to ten times higher, e.g. for the thyroid due to emission of ^{131}I.

TABLE IV. AVERAGE EMISSION RATE OF AEROSOL-BOUND RADIONUCLIDES ^{131}I AND ^{133}I, ^{3}H AND ^{14}C

Nuclide	Average PWR (Ci per GW(e)·a)	Average BWR (Ci per GW(e)·a)
Cr-51	1.3 E-3	3.7 E-2
Mn-54	1.3 E-4	5.5 E-3
Co-57	–	3.3 E-6
Co-58	2.4 E-4	1.0 E-2
Co-60	2.6 E-3	5.5 E-3
Fe-59	5.2 E-4	4.4 E-3
Zn-65	–	1.3 E-2
Sr-89	1.3 E-6	5.3 E-5
Sr-90	2.9 E-7	2.5 E-6
Zr-95	2.6 E-5	2.6 E-4
Nb-95	1.2 E-4	6.8 E-4
Ru-103	1.9 E-5	2.7 E-6
Ru-106	1.7 E-4	–
Ag-110m	2.9 E-5	8.4 E-4
Sb-124	4.3 E-3	8.2 E-4
Sb-125	3.8 E-5	–
Te-123m	5.4 E-4	–
Cs-134	5.3 E-5	–
Cs-137	2.7 E-4	5.0 E-6
Ba-140	1.2 E-4	–
La-140	3.6 E-5	–
Ce-41	8.4 E-6	–
Ce-144	1.3 E-4	1.4 E-5
H-3	37	10
I-131	0.015	0.013
I-133	0.0012	–
C-14	5	10

TABLE V. RADIATION EXPOSURE FROM A NUCLEAR POWER PLANT AT BIBLIS WITH POWER GENERATION OF 1 GW(e)·a

	PWR (mrem)	BWR (mrem)
Whole-body dose (ICRP No. 2)	0.1	0.4
Effective dose equivalent (ICRP No. 26)	0.1	0.4

TABLE VI. RADIATION EXPOSURE FROM A PIT-COAL-FIRED POWER PLANT AT BIBLIS WITH POWER GENERATION OF 1 GW(e)·a

Whole-body dose (mrem) (ICRP No.2)	0.7[a]
Effective dose equivalent (mrem) (ICRP No.26)	0.8

[a] At a distance of 100 m, deposition by washout results in a higher ingestion dose up to twice the value calculated for 2000 m.

4.2.2. *Pit-coal-fired power plant*

Source terms as given in Table II; site data for a 200-m stack at Biblis including thermal lift: atmospheric dispersion factor 3×10^{-8} s·m^{-3} at 2000 m. Based on the particle size distribution of the fly-ash (see Section 2.2), the deposition velocity of the aerosol-bound radionuclides is estimated as 0.015 m·s^{-1} for the uranium, thorium and radium particulates and as 0.006 m·s^{-1} for ^{210}Pb and ^{210}Po [4].

The calculation included the exposure pathways: inhalation and ingestion. The results are summarized in Table VI for the whole-body dose and for the effective dose equivalent. The contribution of those exposure pathways here, of course, is different because of the different radionuclides, and the dose to individual organs of reference man may be up to ten times higher again, for example for the lung or for the bone owing to emission of thorium nuclides and ^{226}Ra.

4.3. Comparison with ambient levels of natural radionuclides

For the coal-fired power plants, it is possible to compare the calculated activity concentration in air, the deposited activity and the resulting specific activity of the soil of the natural radionuclides discharged with their normal values in the environment, and thus to base a statement on radiation exposure caused by the emission from the plant in comparison with natural radiation exposure from those nuclides. The average activity concentration of the radionuclides of the uranium and thorium series in air was taken from the UNSCEAR Report as about 7×10^{-17} $Ci \cdot m^{-3}$; radon is given with 7×10^{-11} $Ci \cdot m^{-3}$; ^{210}Pb and ^{210}Po with 14×10^{-18} $Ci \cdot m^{-3}$ and 3×10^{-18} $Ci \cdot m^{-3}$, respectively.

The results indicate that the activity concentration in air at ground level at the point of maximum exposure is raised by about 10% for uranium nuclides, by about 3% for thorium nuclides, by about 0.2% for ^{210}Pb and by about 2% for ^{210}Po, whereas for ^{222}Rn there is no change at all in the natural level.

From these data, deposition on the ground by fallout may also be calculated and compared with the natural deposition rate. The results give only a rise of about 1–10% of the natural fallout rate at the area of maximum deposition for the uranium, thorium and radium nuclides and much less even for ^{210}Pb and ^{210}Po because of their higher natural deposition rate.

The estimation on the basis of a comparison with the normal natural level of these radionuclides in the environment leads to an effective dose equivalent per GW(e)·a of about 0.2 mrem for the pit-coal-fired plant and about 0.04 mrem for the plant fired with brown coal. This effective dose value is caused almost equally by inhalation, ingestion and external γ-radiation from the ground deposition.

In the same way, calculation of the accumulation of radionuclides in the soil during 30 to 40 years of plant operation indicates a rise of less than 0.1% of the normal natural radioactivity concentration in soil which, in general, will not be measurable [5].

4.4. Discussion

Calculation on the basis of our regulatory guide leads to the following results:

(a) Pit-coal-fired power plant

The ingestion of food after decades of operation proves to be the major exposure pathway for those natural radionuclides emitted. Inhalation contributes only about 10% or less for whole-body dose and bone dose as well as effective dose equivalent.

The main contribution to bone dose by ingestion is by the thorium nuclides and by ^{226}Ra. However, for calculating effective dose equivalent, the nuclide ^{210}Pb is now the main contributor.

TABLE VII. WHOLE-BODY DOSE AND EFFECTIVE DOSE EQUIVALENT FROM POWER PLANTS PER GW(e)·a POWER GENERATION

Methods of calculation	Pit-coal plant (mrem)	PWR plant (mrem)	BWR plant (mrem)
Whole-body dose (ICRP No. 2)	0.7[a]	0.1	0.4
Effective dose equivalent (ICRP No. 26)	0.8	0.1	0.4
Comparison with natural radiation exposure	0.2	Not applicable	

[a] At a distance of 100 m, deposition by washout results in a higher ingestion dose up to twice the value calculated for 2000 m.

The point of maximum exposure here is at a distance of 2000 m. However, at the very short distance of 100 m, deposition by washout results in a higher ingestion dose up to twice as much as the value calculated for 2000 m.

(b) Nuclear power plants

The main exposure pathways are external radiation from noble gases by submersion in the cloud and thyroid exposure from ^{131}I via the milk food chain.

For ingestion, ^{14}C is the main contributor.

Calculation of effective dose equivalent with the dose factors of ICRP No.26 will not result in a significant change compared to the whole-body dose calcluated with dose factors from ICRP No.2.

5. CONCLUSION

Radiation exposure due to airborne radioactive effluents from coal-fired power plants is caused by α-radiation of high linear energy transfer, and, for nuclear power plants, by β- and γ-radiation of low linear energy transfer. Radiation exposure from modern power plants has been calculated on the conservative assumption that the reference person stays throughout the year at the point of maximum exposure outside the fence and obtains food exclusively from the same location.

Considering these assumptions and the other uncertainties inherent in such calculations, it is nevertheless expected that the radiation exposure of the population in the vicinity of power plants in the FRG is well within an order of magnitude the same for modern pit-coal-fired power plants and nuclear power plants and will lie within the range 0.1–1 mrem·a^{-1} for power generation of 1 GW(e)·a at a site. This estimate for coal-fired power plants is low in comparison with other publications, but it is based on measurements at a power plant with efficient electrostatic filters. The individual results are summarized in Table VII.

The additional radiation exposure caused by the operation of coal-fired and nuclear power plants contributes less than 1% of the average natural radiation exposure in the FRG and is also small in comparison with the local variation of the natural radiation exposure.

For a more complete comparison, of course, the emissions of non-radioactive substances from coal-fired power plants would have to be evaluated, and the high demands on the efficiency of the safety installations of nuclear power plants in case of accident would have to be considered, as must finally all the other separate aspects of the total fuel cycle.

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DISCUSSION

R.M. BARKHUDAROV: Comparing the effect on the population of coal-fired and nuclear power plants on the basis of data obtained at the point of maximum concentration, as was done in your paper, does not give an accurate picture since the dispersal of atmospheric releases from thermal and from nuclear power plants does not proceed in the same way even under the same meteorological conditions. The total effects of releases, i.e. the size of the territory and the possible number of people affected, should be taken into account and compared.

J. SCHWIBACH: The same method of calculation should be used to compare radiation exposure due to airborne radioactive effluents. It is a matter of routine to calculate as a first stage the radiation exposure at the 'hottest' point outside the fence. This gives the highest possible exposure to members of the public. In our case, this operation is performed over a distance of 100 m to 2 km. The doses calculated can then be compared, bearing in mind, of course, that the radiation exposure of population groups living further away will be lower. As our results indicate a radiation exposure in the range 0.1–1 mrem per GW(e)·a electric power generated for coal-fired and nuclear plants, this is really an adequate statement from the standpoint of public health.

Our paper quotes five references which contain a description of the dose distribution in the vicinity of plants and the calculated collective doses to the population. As I pointed out in my oral presentation, the effective dose equivalent calculated is due to different radionuclides in each case. In the paper, we therefore also state the dose to the individual organs of a reference man and make our comparison on the basis of the effective dose equivalent at the point of maximum exposure for members of the public.

CONTRIBUTION TO A COMPARATIVE ENVIRONMENTAL IMPACT ASSESSMENT OF THE USE OF COAL AND NUCLEAR ENERGY FOR ELECTRICITY GENERATION FOR SELECTED SITE CONDITIONS IN THE FRG

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Abstract

CONTRIBUTION TO A COMPARATIVE ENVIRONMENTAL IMPACT ASSESSMENT OF THE USE OF COAL AND NUCLEAR ENERGY FOR ELECTRICITY GENERATION FOR SELECTED SITE CONDITIONS IN THE FRG.

Comparative environmental impact assessments for different technologies and strategies of energy use must comprise technical, exposure and detriment assessments. For the case of electricity generation on the basis of coal and nuclear energy, a contribution to such an assessment was made. For the technical assessment the emissions from a model hard-coal unit and from all relevant stages of a model nuclear fuel cycle (PWR) were compiled. Site-specific investigations for typical sites in the Federal Republic of Germany show marked differences. For assessing radiation exposure, a comparison of the fuel cycles on the basis of local and collective exposure values was performed. A risk-specific comparison can only be made on the basis of collective dose commitment calculations. Own estimates of the collective dose equivalents for siting and technical conditions in the FRG were made on the basis of results presented by the UN Scientific Committee on the Effects of Atomic Radiation (UNSCEAR) in 1977. The estimates show a reduction of the collective dose equivalent for the nuclear fuel cycle from about 4000 (gonads) to about 2000 (effective) person-rem/(GW(e)·a); for the coal-fired unit the estimates yield 400 (effective) as compared to 60 (gonads) person-rem/(GW(e)·a) in the UNSCEAR report. The new estimates show that the radiation exposure due to the emissions from coal-fired units is, at least on the local and regional scale, comparable to the non-occupational radiation exposure from all relevant stages of the nuclear fuel cycle at normal operation, each normalized per unit of generated energy. For a comprehensive risk estimate of the nuclear fuel cycle, accidental radiation exposure must also be considered. This was done on the basis of values from the German Risk Study. Normal operation and accidental risk were each aggregated on the detriment levels, showing that the risk contributions from normal operation and from accidents are of the same magnitude. Finally, it is shown that the risk from radiation caused by coal combustion is about an order of magnitude smaller than the total risk of radiation exposure from nuclear power.

1. INTRODUCTION

Comparative environmental impact assessments for different technologies and strategies of energy use are of increasing importance for energy policy decisions. Ideally, such environmental assessments should be part of comprehensive technology assessments. A major problem of such policy-oriented assessments is the gap between the need for clear recommendations on the one side and the lack of sufficiently reliable scientific knowledge on the other. In the following, the different steps of analysis which comparative assessments should comprise will be discussed. It will be shown that technical, exposure and detriment assessments must each be considered. For the case of electricity generation on the basis of coal and nuclear energy, a contribution to such an assessment was made. Within the framework of the technical assessment, the emissions from a model hard-coal unit and from all relevant stages of the nuclear fuel cycle (PWR) were compiled. The exposure assessment had to be limited to radiation exposure, because only for this part are sufficient data and methodological tools available. Comparisons have already been presented on the basis of local dose commitment calculations. A risk-specific comparison, however, can be made only on the basis of collective dose commitment calculations. Such a comparison was presented by the United Nations Scientific Committee on the Effects of Atomic Radiation (UNSCEAR) in 1977 |1|. This report was the basis for our own calculations, taking special account of German technical and site conditions. Moreover, the comparison was made on the basis of regulations for calculating radiation exposure |2, 3|, with only minor modifications in order to reduce the strong conservative bias which characterizes these regulations. Nevertheless, the results presented here are still more on the conservative side. This also applies to the accidental risk values for nuclear power stations which were taken from the German Risk Study |4|. Finally the effective dose commitment values |5| were used, because they are a better index for total risk evaluations than the organ-specific doses.

2. "IDEAL" STRUCTURE OF COMPARATIVE ENVIRONMENTAL IMPACT ASSESSMENTS

Comparative environmental impact assessments for different technologies and strategies of energy use should ideally regard all impact areas, all stages of the fuel cycles and the sequence from emission to detriment. The impact areas should comprise public and occupational health effects, the natural environment and the local, regional and global climate. The explicit consideration of the sequence "emission-dispersion-exposure-detriment" involves the following detailed steps of analysis:

Analysis of the emissions (technical analysis);

Analysis of the transfer of pollutants in the different media and of human exposure (exposure analysis);

Analysis of the detriment to human health and the environment (detriment analysis).

A special problem of the technical analysis is the development of representative emission data sets which describe the average conditions of the use of an energy technology. For the exposure analysis it is necessary to investigate the local, regional and to some extent also the global exposure levels. This requires, among others, the adaptation and use of models for long-range transport, conversion and transfer of pollutants between different environmental media (e.g. atmosphere to soil). Here again the problem arises of finding representative site conditions allowing an estimation for larger geographical regions.

One focus of the methodological work to be carried out within the framework of comparative environmental assessments is to develop approaches and concepts to systematically compare the impacts of different technical options. One possible approach is the aggregation of the different impacts on the detriment level. This procedure requires dose-effect relations for all relevant pollutants, also taking account of the problem of synergism. Until now such comprehensive quantifications are possible only for limited impacts, such as the impact from radiation exposure. As regards impacts such as those from acid rain on the natural environment and the health impacts from chemical pollutants, there may be hope for sufficient quantification in the near future, but for quite a number of impacts, e.g. those from climatic change, the prospects of developing the necessary methodological tools for quantification are at present rather dim. For these cases, qualitative evaluations can be more useful than insufficient quantitative presentations.

3. TECHNICAL ASSESSMENT FOR THE SELECTED FACILITIES

Emission data sets per GWe·a for a bituminous coal-fired power plant and for the relevant facilities of the nuclear fuel cycle were compiled. Apart from the data for the nuclear power plant, this data set has only limited representative character. The calculated specific emissions of sulphurdioxide (SO_2), nitrogenoxides (NO_x), fly-ash, trace and radioactive elements from coal-fired power plants are in compliance with soon-to-be-enacted standards of the official German regulations (table I). As regards

TABLE I. SPECIFIC EMISSIONS FROM POWER PLANTS IN FRG BURNING BITUMINOUS COAL CONFORMING TO EXPECTED STANDARDS, RELATED TO PRIMARY ENERGY (GJ) AND TO ELECTRICAL ENERGY PRODUCED GW(e)·a

	Pollutant concentration in fly-ash	Specific emission	
	(mg/m^3)	g/GJ	10^3 t/GWe-a
Particulate	50	17.5	1.4
Sulphurdioxide (SO_2)	650	228	18
Nitrogenoxides (NO_2) (D)	800	280	22
Nitrogenoxides (NO_2) (L)	1300	455	36
Hydrocarbons		3.3	0.26
Carbonmonoxide		5.9	0.46
Hydrochlorine		12	0.94
Hydrofluorine		2.5	0.20
Carbondioxide		9.7×10^4	7600

D dry ash removal.
L liquid ash removal.

the NO_x emissions, it is necessary to differentiate between the combustion technologies with dry or liquid ash removal. Relevant trace element emissions, which are mostly connected with particulate emissions, were evaluated on the basis of measurements of bituminous coal |6| (table II). Special importance was given to the evaluation of the emission data for the radionuclides of the natural decay chains of uranium (U-238) and thorium (Th-232). A rough estimation can be made, assuming a content of 1 ppm U-238 and 2 ppm Th-232 in the coal, an ash content of 10 %, a heating value of the coal of 30 MJ/kg, a precipitator efficiency of 99 %, and a secular equilibrium of the products of the decay chain. These assumptions, which are similar to those made by McBride |7|, lead to emission rates which are in good accordance with those presented in UNSCEAR |1|, especially for the U-238 chain. Measurements of the fly-ash activity |8, 9| show higher values, especially for the U-238 chain (table III). Marked differences can be seen for the different combustion technologies. There is good agreement

TABLE II. SELECTED TRACE-ELEMENT EMISSIONS BASED ON TRACE-ELEMENT CONCENTRATION OF FLY-ASH ACCORDING TO Ref.[6]

	Trace element concentration of fly-ash	Specific emission	
	(µg/g)	(g/GJ)	(kg/(GWe·a))
Arsenic (As)	500	8.8	690
Cadmium (Cd)	34	0.60	47
Chromium (Cr)	600	11	830
Fluorine (F)	2200	39	3000
Mercury (Hg)$^{+)}$	5	15	1200
Lead (Pb)	2300	40	3200
Selenium (Se)$^{+)}$	120	15	1200
Uranium (U)	30	0.53	41
Vanadium (V)	900	16	1200
Zinc (Zn)	2800	49	3900

$^{+)}$ 90 % of Hg and 20 % of Se in coal may be emitted as gases.

of the two data sets for liquid ash removal. A specific enrichment for the nuclides Pb-210 and Po-210 can be seen. On the basis of these data an emission data set for our own calculations was derived (table IV).

For the nuclear fuel cycle, own emission estimations for the nuclear power plant and the fuel reprocessing plant were made. Present operating experience and licensing claims for both facilities were taken account of. The emission data set for a typical German PWR was developed from the actual emissions of three plants since 1972, each normalized on 1 GWe·a (table V). This data base will be updated each year and will thus provide a good basis for statistical evaluations. Theoretical calculations simulating typical emission-pathways |10| were also made. First results show sufficient agreement between theoretical and empirical values. For the fuel reprocessing and waste management plant, a global estimation on the basis of release factors was made. The operating experience of the Karlsruhe pilot plant |11| and the nuclide retention planned for the commercial facility |12| were also accounted for. The release factors chosen for the model plant are in good

TABLE III. RADIONUCLIDE CONCENTRATIONS IN FLY-ASH FROM COAL-FIRED POWER PLANT (BITUMINOUS COAL) IN Bq/g (pCi/g)

NUCLIDES	UNSCEAR, 1977 [1] (estimation)		McBRIDE et al., 1977 [7] (estimation)		CHATTERJEE et al., 1980+ [8] (L)		KOLB, 1977++ [9] (D)	KOLB, 1977++ [9] (L)	Chosen values for own emission data set (D)	Chosen values for own emission data set (L)
U-238	0.19	(5.0)	0.12	(3.3)	0.36	(9.8)	0.27 (5.9)	0.33 (8.9)	0.22 (5.9)	0.33 (8.9)
Th-234	-		0.12	(3.3)	-		-	-	0.13 (3.6)	0.20 (5.3)
U-234	-		0.12	(3.3)	0.36	(9.8)	-	-	0.22 (5.9)	0.33 (8.9)
Th-230	-		0.12	(3.3)	-		-	-	0.13 (3.6)	0.20 (5.3)
Ra-226	0.04	(1.0)	0.12	(3.3)	0.27	(7.3)	0.18 (4.9)	0.28 (7.6)	0.16 (4.3)	0.24 (6.6)
Pb-210	0.37	(10)	0.12	(3.3)	2.4	(66)	0.17 (4.6)	1.6 (43)	0.17 (4.6)	1.6 (43)
Po-210	-		0.12	(3.3)	4.6	(125)	-	-	-	4.6 (125)
Th-232	0.04	(1.0)	0.08	(2.2)	0.15	(4.0)	-	-	0.09 (2.3)	0.13 (3.5)
Ra-228	0.04	(1.0)	0.08	(2.2)	-		0.08 (2.2)	0.13 (3.5)	0.11 (2.9)	0.16 (4.4)
Th-228	0.04	(1.0)	0.08	(2.2)	0.15	(4.0)	0.09 (2.3)	0.13 (3.5)	0.09 (2.3)	0.13 (3.5)
Ra-224	-		-		-		-	-	0.11 (2.9)	0.16 (4.4)
K-40	0.6	(15)	0.6	(15)	-		-	-	0.6 (15)	0.6 (15)

D dry ash removal.
L liquid ash removal.

+) fly-ash (after precipitation).

++) precipitator ash measurement (last stage).

TABLE IV. EMISSION DATA SETS FOR RADIONUCLIDE EMISSIONS FROM (BITUMINOUS) COAL-FIRED POWER PLANTS

Radionuclides	UNSCEAR, 1977 [1]		Emission data set for model plant: dry ash removal			Emission data set for model plant: liquid ash removal		
	GBq/GWe.a	mCi/GWe.a	kg/GWe.a	GBq/GWe.a	mCi/GWe.a	kg/GWe.a	GBq/GWe.a	mCi/GWe.a
U-238	2	50	25	0.3	8	37	0.5	13
Th-234	-	-	$2 \cdot 10^{-10}$	0.2	5	$3 \cdot 10^{-10}$	0.3	7
U-234	-	-	$1 \cdot 10^{-3}$	0.3	8	$2 \cdot 10^{-3}$	0.5	13
Th-230	-	-	$3 \cdot 10^{-4}$	0.2	5	$4 \cdot 10^{-4}$	0.3	7
Ra-226	0.4	10	$6 \cdot 10^{-6}$	0.2	6	$9 \cdot 10^{-6}$	0.3	9
Rn-222	40	1000	$6 \cdot 10^{-9}$	32	870	$6 \cdot 10^{-9}$	32	870
Pb-210	4	100	$9 \cdot 10^{-8}$	0.2	6	$8 \cdot 10^{-7}$	2	59
Po-210	-	-	$2 \cdot 10^{-9}$	0.3	9	$4 \cdot 10^{-8}$	6	150
Th-232	0.4	10	31	0.1	3	44	0.2	5
Ra-228	0.4	10	$1 \cdot 10^{-8}$	0.2	4	$2 \cdot 10^{-8}$	0.2	6
Th-228	0.4	10	$4 \cdot 10^{-9}$	0.1	3	$6 \cdot 10^{-9}$	0.2	5
Ra-224	0.4	10	$3 \cdot 10^{-11}$	0.2	4	$4 \cdot 10^{-11}$	0.2	6
Rn-220	-	-	$6 \cdot 10^{-13}$	21	580	$6 \cdot 10^{-13}$	21	580
K-40	6	150	3	0.8	21	3	0.8	21

TABLE V. SPECIFIC EMISSIONS FROM THE MODEL NUCLEAR POWER PLANT (PWR)

A) GASEOUS EFFLUENTS

EMITTED NUCLIDE	EMISSION AVERAGE Bq/GWe.a	Ci/GWe.a	RELATIVE DEVIATION (%)	% PER NUCLIDE
H-3	1.66E+12	4.49E+01	100.0	89.1467
C-14	2.02E+11	5.47E+00	100.0	10.8532
NOBLE GASES				
AR-41	3.37E+12	9.11E+01	61.6	0.9728
KR-85	4.99E+13	1.35E+03	58.8	14.3941
KR-85M	1.80E+13	4.85E+02	96.3	5.1851
KR-87	6.55E+10	1.77E+00	57.6	0.0189
KR-88	8.38E+12	2.26E+02	91.2	2.4189
KR-89	6.55E+10	1.77E+00	57.6	0.0189
XE-131M	1.84E+11	4.98E+00		0.0532
XE-133	2.22E+14	6.01E+03	66.4	64.1785
XE-133M	1.30E+13	3.51E+02	62.1	3.7461
XE-135	3.10E+13	8.38E+02	61.7	8.9559
XE-135M	6.73E+10	1.82E+00	58.2	0.0194
XE-137	6.73E+10	1.82E+00	58.2	0.0194
XE-138	6.55E+10	1.77E+00	57.6	0.0189
PARTICULATES				
CR-51	6.40E+07	1.73E-03	60.7	2.1406
MN-54	3.53E+07	9.53E-04	55.0	1.1802
FE-59	1.17E+07	3.16E-04	49.4	0.3911
CO-57	4.57E+05	1.24E-05	49.7	0.0153
CO-58	2.90E+08	7.83E-03	48.0	9.6925
CO-60	4.87E+08	1.32E-02	40.8	16.2850
SR-89	1.57E+06	4.23E-05	100.0	0.0524
SR-90	2.38E+05	6.44E-06	50.8	0.0080
NB-95	2.77E+07	7.48E-04	87.3	0.9258
ZR-95	2.60E+07	7.02E-04	82.5	0.8688
RU-103	3.07E+07	8.30E-04	88.1	1.0272
RU-106	1.22E+06	3.30E-05	42.4	0.0408
AG-110M	8.12E+07	2.19E-03	90.4	2.7167
SB-124	4.95E+07	1.34E-03	68.2	1.6573
TE-123M	2.85E+07	7.70E-04	100.0	0.9538
CS-134	2.09E+07	5.65E-04	53.3	0.7000
CS-137	6.14E+07	1.66E-03	47.3	2.0550
LA-140	6.83E+06	1.85E-04	85.4	0.2286
CE-141	1.24E+06	3.36E-05	86.6	0.0415
CE-144	1.66E+07	4.47E-04	66.0	0.5539
I-131	1.75E+09	4.72E-02	45.8	58.4669

B) LIQUID EFFLUENTS

H-3	1.49E+13	4.04E+02	100.0	100.0000
CR-51	3.80E+09	1.03E-01	74.0	2.3831
MN-54	2.53E+09	6.85E-02	48.3	1.5871
FE-59	3.22E+07	8.70E-04	52.3	0.0202
CO-57	2.31E+07	6.24E-04	52.9	0.0145
CO-58	2.66E+10	7.18E-01	51.4	16.6383
CO-60	2.28E+10	6.15E-01	44.2	14.2586
ZN-65	1.41E+07	3.80E-04		0.0088
SR-89	3.59E+08	9.70E-03	45.9	0.2248
SR-90	1.71E+08	4.61E-03	70.7	0.1070
ZR-95	2.20E+08	5.95E-03	47.9	0.1379
NB-95	2.65E+08	7.16E-03	41.9	0.1660
MO-99	1.64E+08	4.43E-03		0.1028
RU-103	2.63E+08	7.11E-03	54.7	0.1649
RU-106	1.28E+08	3.45E-03		0.0800
AG-110M	1.14E+09	3.09E-02	48.5	0.7163
SB-124	3.45E+09	9.32E-02	38.6	2.1617
SB-125	6.79E+07	1.83E-03	74.9	0.0425
TE-123M	7.99E+07	2.16E-03	100.0	0.0501
I-131	5.21E+09	1.41E-01	53.5	3.2642
CS-134	2.17E+10	5.87E-01	38.7	13.6090
CS-137	3.43E+10	9.26E-01	42.0	21.4747
BA-140	2.69E+10	7.28E-01		16.8710
LA-140	6.39E+09	1.73E-01	91.3	4.0057
CE-141	2.11E+09	5.70E-02	74.2	1.3205
CE-144	9.42E+08	2.55E-02	67.5	0.5904

TABLE VI. SPECIFIC EMISSIONS FROM THE MODEL NUCLEAR FUEL REPROCESSING PLANT

EMITTED NUCLIDES	INVENTORY[+]		RELEASE FACTOR	EMISSIONS	
	g/t HM	Ci/t HM		Ci/GWe·a	Bq/GWe·a
GASEOUS EFFLUENTS *					
H-3 **	1.49E-02	1.44E+02	2.50E-01	1.11E+03	4.08E+13
C-14 **	7.15E-02	3.19E-01	1.00E+00	9.82E+00	3.63E+11
MN-54	3.19E-03	2.47E+01	1.00E-08	7.60E-06	2.81E+05
FE-55	3.77E+00	9.43E+03	1.00E-08	2.90E-03	1.07E+08
CO-58	1.28E-11	4.07E-07	1.00E-08	1.25E-13	4.63E-03
CO-60	1.35E+01	1.53E+04	1.00E-08	4.71E-03	1.74E+08
NI-63	3.40E+01	2.10E+03	1.00E-08	6.46E-04	2.39E+07
KR-85	1.70E+01	6.68E+03	5.00E-02	1.03E+04	3.80E+14
SR-89	1.75E-14	5.10E-10	1.00E-08	1.57E-16	5.81E-06
SR-90	5.16E+02	7.04E+04	1.00E-08	2.17E-02	8.01E+08
Y-91	3.34E-12	8.19E-08	1.00E-08	2.52E-14	9.32E-04
ZR-95	7.47E-11	1.61E-06	1.00E-08	4.95E-13	1.83E-02
NB-95	9.11E-11	3.57E-06	1.00E-08	1.10E-12	4.06E-02
TC-99	8.54E+02	1.45E+01	1.00E-08	4.46E-06	1.65E+05
RU-103	1.31E-18	4.23E-14	1.00E-07	1.30E-19	4.82E-09
RU-106	1.34E+00	4.47E+03	1.00E-07	1.38E-02	5.09E+08
AG-110M	5.39E-04	2.56E+00	1.00E-08	7.88E-07	2.91E+04
SB-125	1.65E+00	1.71E+03	1.00E-08	5.26E-04	1.95E+07
TE-125M	2.33E-02	4.19E+02	1.00E-08	1.29E-04	4.77E+06
TE-127M	1.25E-07	1.18E-03	1.00E-08	3.63E-10	1.34E+01
TE-127	4.39E-10	1.16E-03	1.00E-08	3.57E-10	1.32E+01
I-129	1.91E+02	3.37E-02	1.00E-02	1.04E-02	3.84E+08
CS-134	1.23E+01	1.59E+04	1.00E-08	4.89E-03	1.81E+08
CS-137	1.14E+03	9.93E+04	1.00E-08	3.06E-02	1.13E+09
CE-141	1.26E-22	3.58E-18	1.00E-08	1.10E-24	4.08E-14
CE-144	7.80E-01	2.49E+03	1.00E-08	7.66E-04	2.83E+07
PU-238	1.49E+02	2.55E+03	1.00E-08	7.85E-04	2.90E+07
PU-239	5.03E+03	3.13E+02	1.00E-08	9.63E-05	3.56E+06
PU-240	2.38E+03	5.43E+02	1.00E-08	1.67E-04	6.18E+06
PU-241	9.06E+02	9.33E+04	1.00E-08	2.87E-02	1.06E+09
NP-239	8.42E-05	1.95E+01	1.00E-08	6.00E-06	2.22E+05
AM-241	3.94E+02	1.35E+03	1.00E-08	4.15E-04	1.54E+07
AM-242	5.09E-06	4.12E+00	1.00E-08	1.27E-06	4.69E+04
CM-242	1.29E-03	4.28E+00	1.00E-08	1.32E-06	4.87E+04
CM-243	2.87E-01	1.48E+01	1.00E-08	4.55E-06	1.68E+05
CM-244	1.74E+01	1.41E+03	1.00E-08	4.34E-04	1.61E+07
LIQUID EFFLUENTS:					
H-3				9.40E+03	3.48E+14
SR-89/SR-90				5.00E-02	1.85E+09
RU-106				8.40E-02	3.11E+09
CS-134/CS-137				4.30E-01	1.59E+10
PU-238/PU-239				3.50E-03	1.30E+08

[+] HM = heavy metal.

* Estimates on the basis of the assumptions:
Burnup 36 000 MW·d/t HM
Enrichment 3.5%
Fuel decay time 7 years.

** Only the fuel inventory has been taken into account.

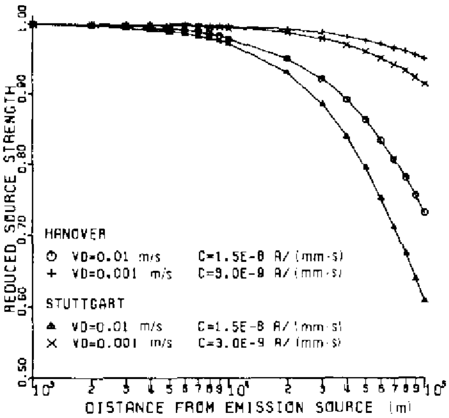

FIG.1. Annual values of the reduced source strength for two sites in the FRG (Hanover – Northern; Stuttgart – Southern). Stack height: 200 m. VD = deposition velocity. C = washout.

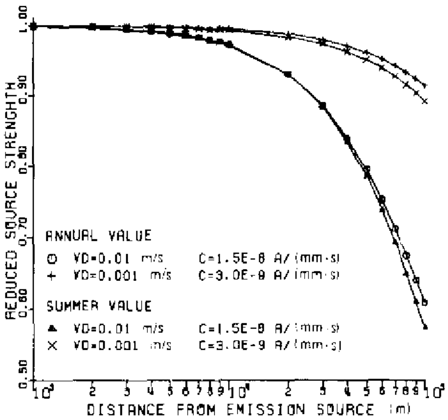

FIG.2. Annual and summer values of the reduced source strength for the southern site in the FRG. Stack height: 200 m. VD = deposition velocity. C = washout.

agreement with those from other publications |13|. Table VI gives an overview of the gaseous and liquid effluents from the model fuel reprocessing plant.

4. EXPOSURE ASSESSMENTS FOR THE SELECTED FACILITIES AND SELECTED SITES

As an example for the exposure assessment, the radiation exposure was calculated because for this impact area sufficient data and methodological tools are available. The influence of different sites can be studied on plots of the reduced source strength, defined as that fraction of the original source strength which has not been depleted within the regarded distance. Moreover, local and collective dose commitments from a coal-fired power plant and from the facilities of the total nuclear fuel cycle were presented. Our own calculations were based on effective dose values |5| with dose conversion factors from |14|. In addition, the regulations for calculating the radiation exposure were accounted for |2, 3|.

The reduced source strength for a typical northern and a typical southern site in the FRG (Fig. 1), and, for the case of the southern site, for a typical full year and a typical summer (Fig. 2),were calculated up to a distance of 100 km. The calculations were made using a diffusion model of the Gaussian type which

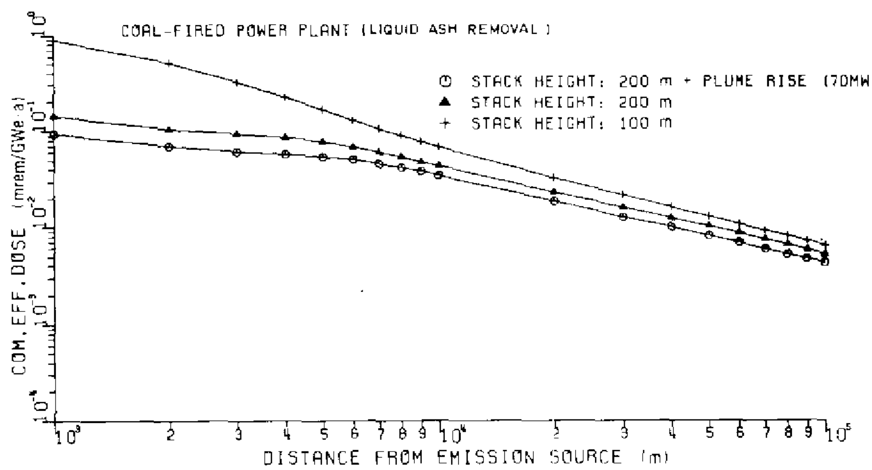

FIG.3. Local committed effective dose equivalent for a model coal-fired power plant with different stack heights and northern FRG site conditions.

also includes the influence of an upper inversion. The depletion processes of dry and wet disposal were also taken into account. For the fallout process two deposition velocities, 0.01 and 0.001 m/s, were considered; for the washout coefficient also, two different assumptions were made. High fallout and washout factors were combined to describe the behaviour of reactive gases like sulphurdioxide or iodine, and low fallout and washout factors were taken for aerosols of a medium diameter of 1 µm. In each case, the washout contribution is only about 10 % of that from the fallout. The aerosol depletion within the 100-km distance is negligible; only for reactive gases a marked depletion can be shown. This points out that for aerosols there is a long-range transport problem. A site-specific difference for the reduced source strength can be seen of about 5 % for aerosols and of about 15 % for reactive gases each in 100 km distance from the source, which means that, on the whole, the northern site conditions will yield lower exposure values in the local range (Fig. 1). Moreover, there is no marked difference between the full year and summer periods. Thus, for calculating the exposure via direct deposition on external plant surfaces the growth season has not to be regarded in order to get conservative results (Fig. 2).

The calculations of the local radiation exposure, made for the northern site conditions, show values far below the current dose limits. The different structure of the plots points out that a comparison of exposure values in one selected point is quite insufficient. The results for the coal-fired power plant (liquid

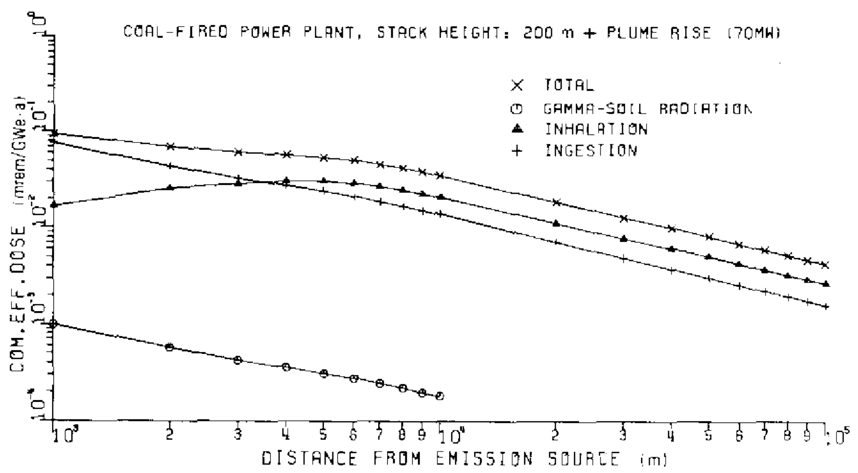

FIG.4. Contribution of different exposure pathways to the total local committed dose equivalent for a model coal-fired power plant; northern FRG site conditions.

ash removal) show the influence of different stack heights and of plume rise for the case of 70-MW thermal emission and 200-m geometrical emission height (Fig. 3). The very high dose values for coal-fired power plants which have been published |9| considered only the case of 100-m stack height without plume rise. For large coal-fired power plants, 200-m stack height with plume rise can be considered as realistic, resulting in a maximal local committed dose of about 0.1 mrem/(GWe·a). The contribution of different pathways to the total exposure shows major contributions of inhalation and ingestion, inhalation being slightly dominant (Fig. 4). The relevant facilities of the nuclear fuel cycle, the nuclear power plant (PWR) and the fuel reprocessing plant, show values of the same magnitude (Fig. 5). The exposure values for the power plant decline much faster than those for the reprocessing plant, due to the contribution via the different exposure pathways. For the power plant, γ-submersion is the major contributor to total exposure (Fig. 6); for the reprocessing plant it is the ingestion pathway (Fig. 7).

A risk-specific comparison can only be made on the basis of collective dose commitment calculations. Such a comparison of the collective dose equivalents, especially the 500-year committed dose equivalents for gonads, was presented by the United Nations Scientific Committee on the Effects of Atomic Radiation (UNSCEAR) in 1977 |1|. For reasons of comparability the dose values from |1| were converted into dose-equivalent values according to the RBE factors from |5|. New estimates on the basis of effective dose

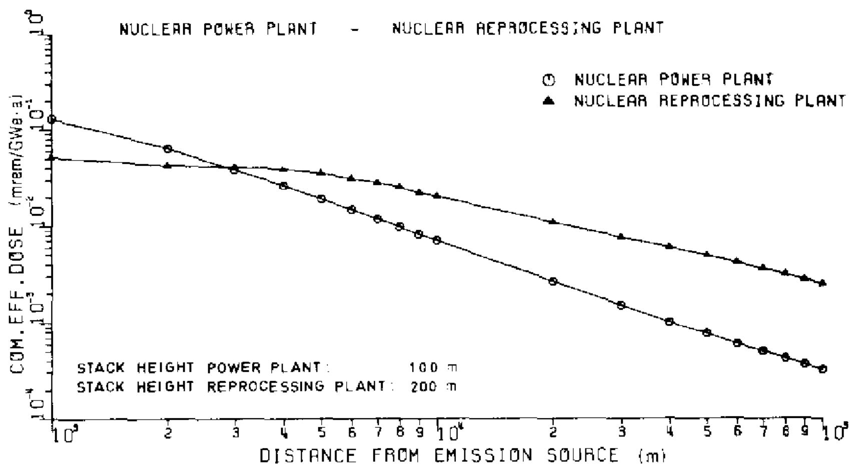

FIG.5. Local committed effective dose equivalent for a model nuclear power plant (PWR) and a model fuel-reprocessing plant for northern FRG site conditions.

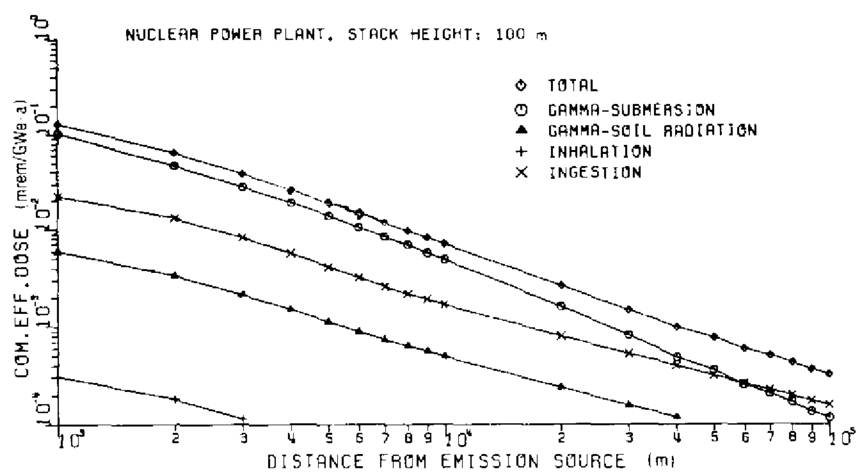

FIG.6. Contribution of different exposure pathways to the total local committed dose equivalent for the model nuclear power plant; northern FRG site conditions.

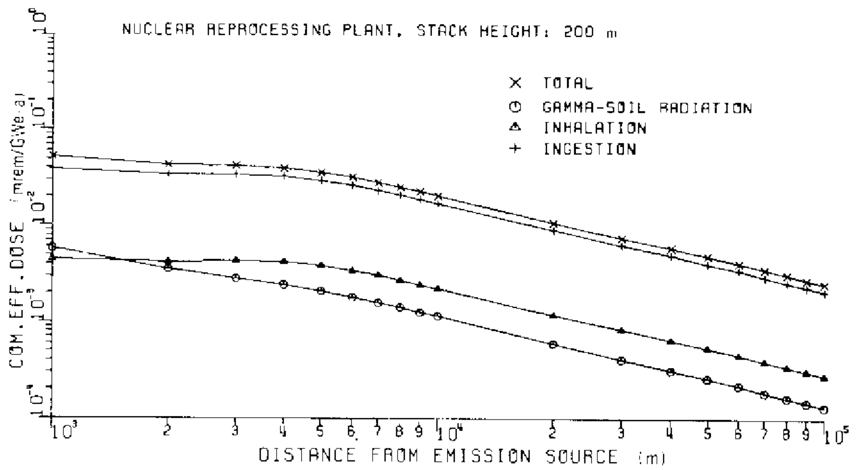

FIG.7. Contribution of different exposure pathways to the total local committed dose equivalent for the model fuel-reprocessing plant; northern FRG site conditions.

commitments take special account of German technical and siting conditions. Apart from the new emission data sets discussed above, the greater population density of the FRG (250 cap/km^2 versus 100 cap/km^2 in |1|) is of particular importance. For the coal-fired power plant these calculations also take into account additional exposure pathways which were not regarded in |1|, e.g. inhalation and ingestion after deposition of the nuclides on plant surfaces. The new calculations were based on the same model assumptions as in |1|, meaning for the ingestion pathway that the nuclides are deposited within a radius of 500 km and accumulate on the upper 20 cm of soil. The new estimates yield 400 (effective) as compared to 60 (gonads) person-rem/(GWe·a) (table VII). The exposure is mainly due to the ingestion and, to a lower extent, to the inhalation pathways; γ-soil radiation from potassium-40 is of only minor importance. The dominant contribution of the ingestion pathway in this case, contrary to the local exposure result, is caused by the strong conservative assumptions which were made when calculating the ingestion pathway. Recent publications point out that γ-soil radiation will increase, taking also into account the short-lived daughter nuclides from radium-224 |14|.

For the nuclear fuel cycle the main facilities, which were already considered within the framework of the technical assessment, the power plant (PWR) and the fuel reprocessing plant were investigated. Again, the same model assumptions were made as in |1| with the modification that typical German technical and siting conditions were assumed. In addition, the global distribution of

TABLE VII. RADIONUCLIDE EMISSIONS AND COLLECTIVE DOSE-EQUIVALENT COMMITMENT FROM COAL-FIRED POWER PLANTS

NUCLIDE	EMISSIONS* (mCi/GWe·a) UNSCEAR 1977+	EMISSIONS* (mCi/GWe·a) OWN CALCULATIONS	COLLECTIVE DOSE EQUIVALENT COMMITMENT (PERSON-REM/GWe·a) UNSCEAR, 1977+ GONADS Ingestion	UNSCEAR, 1977+ GONADS Inhalation	OWN CALCULATIONS EFFECTIVE DOSE Ingestion	OWN CALCULATIONS EFFECTIVE DOSE Inhalation	OWN CALCULATIONS EFFECTIVE DOSE γ-soil radiation
U-238	50	13	10	no values given	0.1	8.5	
Th-234		7.3			-	-	
U-234		13			0.2	17.0	
Th-230		7.3			2.3	13.7	
Ra-226	10	9.4	1.4		8.3	0.5	
Pb-210	100	59	40		154	7.8	
Po-210		150			90.8	9.5	
Th-232	10	4.8	0.2		3.2	31.5	
Ra-228	10	6.1	0.3		5.5	-	
Th-228	10	4.8			0.5	10.5	
Ra-224		6.1			0.3	0.3	
K-40	150	21.0	-		-	-	8
TOTAL			54.6		265	99.6	8
					373		

* 1 Ci = 3.7×10^{10} Bq.
+ Ref.[1].

TABLE VIII. COLLECTIVE DOSE-EQUIVALENT COMMITMENT DUE TO RADIOACTIVE EMISSIONS FROM NUCLEAR POWER PLANTS

NUCLIDE	EXPOSURE PATHWAY	RADIOACTIVE EMISSIONS* (Ci/GWe·a) UNSCEAR, 1977+	RADIOACTIVE EMISSIONS* (Ci/GWe·a) OWN CALCULATIONS	COLLECTIVE DOSE EQUIVALENT COMMITMENT (PERSON-REM/GWe·a) UNSCEAR, 1977 (GONADS)+	OWN CALCULATIONS (GONADS)	OWN CALCULATIONS (EFFECTIVE DOSE)
Kr, Xe, Ar	atmosphere (local and regional)	$1 \cdot 10^6$	$9.4 \cdot 10^3$	200	9.0	9.0
H - 3		$1.8 \cdot 10^4$	$4.5 \cdot 10^1$	4	0.07	0.07
C - 14		6.0	6.0	0.6	1.5	1.5
Cs, Co, Sr, Ru		$7.0 \cdot 10^{-2}$	$7.0 \cdot 10^{-2}$	3.3	3.0	3.1
H - 3	water (local and regional)	$4 \cdot 10^3$	400	30	2.8	2.8
Cs, Co, Mn, I		5.8	2.2	6.0	3.2	3.3
H - 3	atmosphere and water (global)	$1.6 \cdot 10^4$	500	50	1.5	1.5
C - 14		6.0	6.0	270	135	135
TOTAL				564	156	156

* 1 Ci = 3.7×10^{10} Bq.
+ Ref.[1].

TABLE IX. COLLECTIVE DOSE-EQUIVALENT COMMITMENT DUE TO RADIOACTIVE EMISSIONS FROM REPROCESSING PLANTS

NUCLIDE	EXPOSURE PATHWAY	RADIOACTIVE EMISSIONS + (Ci/GWe·a)		COLLECTIVE DOSE-EQUIVALENT COMMITMENT (PERSON-REM/GWe·a)		
		UNSCEAR 1977 *	OWN CALCULATIONS	UNSCEAR, 1977 (GONADS) *	OWN CALCULATIONS (GONADS)	OWN CALCULATIONS (EFFECTIVE DOSE)
Kr - 85	atmosphere (local and regional)	375 000	10 000	0.7	0.05	0.05
H - 3		1 000	1 000	0.2	0.5	0.5
C - 14		14	10	1	1.8	1.8
Cs, Ru, Sr		0.1	0.4	0.2	56	57
H - 3	water (local and regional)	6 000	10 000	40	67	67
Cs, Ru, Sr, I		100	1.0	40	1.7	1.6
H - 3	atmosphere and water (global)	7 000	11 000	22	34	34
Kr - 85		375 000	10 000	90	2.5	2.5
C - 14		14	10	630	225	225
TOTAL				824	389	390

+ 1 Ci = 3.7×10^{10} Bq.

* Ref.[1].

carbon-14 is calculated according to a model from |15|, which treats the worldwide distribution of carbon dioxide in a more differentiated manner, thus yielding lower exposure values. The main exposure contributions come from the global exposure due to tritium and especially to carbon-14 emissions (tables VIII and IX). The much lower noble gas and tritium emissions to be expected from the model power plant will also yield lower exposure values than calculated in |1|, in total about 200 (effective) as compared to roughly 600 (gonads) person-rem/(GWe·a) (table VIII). Lower exposure values also result from the emissions of the model reprocessing plant, i.e. 400 (effective) as compared to 900 (gonads) person-rem/(GWe·a) in |1|. Lower krypton-85 emissions due to the krypton retention technique to be installed, and lower water effluents, are the reasons for this improvement (table IX).

In table X a global overview of all exposure parts from the different stations of the nuclear fuel cycle can be seen. The lower ocupational radiation exposure from the model nuclear power plant of 400 person-rem/(GWe·a) is only valid for the pressurized-water reactor (PWR); summarizing the values for all other reactor types yields about 1000 person-rem/(GWe·a), similar to that in |1|. The low occupational radiation exposure from the model fuel reprocessing plant is based on extrapolated experience with the

TABLE X. COLLECTIVE DOSE-EQUIVALENT COMMITMENTS AND NUMBER OF FATALITIES (SOMATIC) FOR THE NUCLEAR FUEL CYCLE AND FOR COAL-FIRED POWER PLANTS

	COLLECTIVE DOSE EQUIVALENT COMMITMENT (PERSON-REM/GWe·a)			NUMBER OF FATALITIES (SOMATIC) (1/GWe·a)
	UNSCEAR, 1977*	OWN CALCULATIONS		
	GONADS	GONADS	EFFECTIVE DOSE	
nuclear power plant				
public	600	160+	160+	0.02
occupational	1000	400+	400+	0.05
reprocessing plant				
public	800	400	400	0.05
occupational	1200	100	100	0.01
mining, milling, fuel fabrication, transportation				
public	20	20	500	0.06
occupational	200	200	200	0.02
total for nuclear fuel cycle (normal operation)	3820	1280	1760	0.22
nuclear accidents				
non-stochastic effects				$4\cdot10^{-5}$
stochastic effects			3000	0.4
coal-fired power plants	60	-	373	0.05

* Ref.[1].
+ This value is valid for PWR plants.

pilot plant at Karlsruhe. The radiation exposure values for mining, milling, fuel fabrication and transportation have been taken from |1|. The high value of the effective dose compared to the dose for gonads is due to the large lung exposure contribution from the radon-isotopes. On the whole, the new estimates show a reduction of the collective exposure values for the nuclear fuel cycle from about 4000 (gonads) to about 2000 (effective) person-rem/(GWe·a). For the coal-fired unit the new estimates yield 400 (effective) as compared to 60 (gonads) person-rem/(GWe·a) in |1|. The new estimates show that the radiation exposure due to the emissions from coal-fired units is, at least at the local and regional levels, comparable to the non-occupational radiation exposure from all relevant stages of the nuclear fuel cycle at normal operation, each normalized per energy generated. Taking into account occupational and non-occupational exposure at normal operation of the facilities, the contribution from the nuclear fuel cycle is about a factor 4 higher than that of energy conversion by coal.

5. DETRIMENT ASSESSMENT

A major risk of the use of nuclear energy is the effect of potential accidents. A comparison of this part of the overall pact with that from normal operating conditions of the facilities is difficult not only because accidents are events happening by chance, the occurrence of which is described by probability distributions, but also because the possible exposure values are so different that totally different main impact areas have to be considered (stochastic and non-stochastic health impacts). For a comparison of the risk from normal operation with the risk due to an accident, an aggregation of stochastic and non-stochastic impacts would be necessary; this is only possible on the detriment level. Both stochastic and non-stochastic effects are specified in the German Risk Study for nuclear power plants |4|. The stochastic risk, which is proportional to the dose commitment values (expected value), is given with about 3000 person-rem/(GWe·a) (table X). A rough estimate of the total health impact from the radiation exposure from the two alternatives, coal and nuclear power, can thus be made on the basis of the collective dose commitment values multiplied by the dose-effect-relationship for stochastic effects |5| while taking into account, in addition, the non-stochastic accidental risk from nuclear power plants as given in |4|. This estimation on the detriment level indicates that:

The risk contributions from normal operation and from accidents have the same magnitude in the case of nuclear energy, and

The risk from radiation caused by coal combustion is about an order of magnitude smaller than the total radiation exposure risk of nuclear power.

Finally, it should be pointed out that an overall statement presupposes an overall comparison comprising, besides the radiation exposure inside and outside the facility dealt with above, further occupational and other risks, especially of coal-fired units and of coal mining, which do not constitute radiation exposures. For the nuclear fuel cycle, the radiation exposure is the principal risk; for coal-fired units it is only a part thereof.

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DISCUSSION

K. SUNDARAM: In Table X you have calculated the number of fatalities (somatic) per GW(e)·a for coal-fired power plants, arriving at a value of 0.05. Does this figure include the stochastic effects estimated from radioactivity releases and other carcinogenic and mutagenic chemicals released? If this figure is only for radioactive releases then it should be so stated, as the figure might otherwise be taken as a comprehensive estimate.

G. HALBRITTER: The value concerns only the radiation exposure risk. Risk assessment for chemical pollutants was not performed in this paper, and only the emissions of these pollutants were examined.

A COAL INDUSTRY'S VIEW OF RISK COMPARISON OF ENERGY SYSTEMS

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Abstract

A COAL INDUSTRY'S VIEW OF RISK COMPARISON OF ENERGY SYSTEMS.

The benefit of risk comparison of different energy systems is questioned and the methods applied to collect data and quantify hazards for comparative purposes are discussed. The different quality of the risk of a major nuclear accident and the scientific uncertainties about the impacts of the use of coal are shown. Improvements are suggested, particularly concerning a true assessment of the relationship between air pollution and human health.

1. THE BENEFITS OF RISK COMPARISON OF ENERGY SYSTEMS

Late in April 1981 a Council of twelve independent environmental experts, appointed by the Federal German Government, gave their opinion on Energy and Environment in a 435-page report [1]. The judgement of the experts is a strong confirmation of what the coal industry predicted to be the outcome of all endeavours in this fast-growing branch of science: No acquittal of nuclear energy; instead, both nuclear energy and coal are now in the dock. Alongside very critical statements on nuclear energy, the experts fear, in respect of coal, "adverse health effects, which are still little explored"[1]. According to the Council, both energy sources can cause environmental burdens or hazards, so that neither is suited for massive expansion. The experts' advice: *Cut down on demand.*

Some lessons can be learned from this. First, the obvious: There is no such thing as risk-free energy. Secondly, risk comparisons must not necessarily verify what is the objective of most risk comparison assessments [2], that using coal is far more risky than using nuclear energy. This confirms world-wide experience that comparison of nuclear risks with those of coal-burning has not brought about public acceptance of nuclear energy. Thirdly, if our existing social and economic

[1] From paragraphs 615 and 618 of the draft of Ref. [1].

structure is taken into account, the margin for practical recommendations drawn from risk comparison is amazingly small. To recommend energy saving will hardly change the course of events. Indeed, in order to secure the future energy supply and at the same time avoid serious loss of political support, governments seem to depend on constant bargaining with the different social groups about sites, capacity and licensing conditions. In this bargaining process, risk analysts can only procure tactical arguments instead of substantially influencing the policy of 'decision-makers'.

On the other hand, comparative risk assessment has contributed considerably to the difficulties in licensing coal-fired power stations. Under the German Federal Immission Control Law of 1974, these installations can only be licensed under the condition that it is ensured that no harmful effect on the environment can be caused by the operation of the plant. This question has to be discussed with all objectors to the project in an open hearing. All the arguments drawn from comparative risk analyses must, of course, be brought up in these hearings, starting with the death rate allegedly caused by SO_2 emissions and ending with the CO_2 issue. Comparative risk assessment studies, their publication and their sensational coverage in the media are to a large extent responsible for a radical change in the public's perception of risk connected with coal-burning, and have thus added to the public's fears, frustration and distrust in respect of the power industry as a whole.

Risk comparison in this field has definitely done more harm than good.

2. CRITICAL OBSERVATIONS ON METHODS OF RISK COMPARISON

In general, comparative risk analyses on nuclear energy and coal are based on the assumption that public acceptance of nuclear energy can be secured if only people are shown that the risks of already accepted energy systems are very much higher. This assumption has proved erroneous, since people have not, until recently, been aware of the alleged risks pointed out to them by risk analysts and are not inclined to accept them.

In the Federal Republic of Germany, it is mainly the foreign studies of Hamilton [3, 4] and Inhaber [5] and the German reports of the Battelle Institute [6] and those of the German Reactor Safety Commission [7] that have received public notice. Examining other published studies [8], one is struck by the fact that most of them do not differ much in the input data, in the methods of collecting these data, and in their calculations to find a common denominator for comparative reasons, such as death per GW(e)·a or man-days lost per GW(e)·a.

However, the results of these studies display a wide spectrum of different figures that cannot be explained only by scientific uncertainties (particularly dose-response relationships) but seem to reflect the subjective quality of judgement in this field. The risks connected with coal are described in numbers ranging from two to 253 deaths per GW(e)·a. Depending on which numbers one prefers to compare, in an overall view coal-connected risks are either equal to nuclear risks or they orders of magnitudes higher.

The following three sections of the coal fuel cycle are said to make the main contribution to these figures:

Public fatal diseases: up to 240 deaths/GW(e)·a

Occupational deaths and fatal diseases:
In coal-mining up to 9 deaths/GW(e)·a
In transportation up to 3 deaths/GW(e)·a

2.1. Coal-mining

Here the main risk in question is coal extraction and preparation. The Battelle Study [6], for instance, calculated the risk in this area to be more than six deaths per GW(e)·a. Meanwhile the statistical data of 1970 used by Battelle (i.e. frequency of accidents) decreased by 30%. This shows that even actual data become obsolete within a few years. Therefore, Farmer proposed to extrapolate data to the period 2000–2050 instead of using data from the past [2].

In coal-mining, empirical data have existed for a couple of decades, while data for uranium-mining are partly analytical, owing to the long latent period of long-term effects of radon. This long-term safety record has considerably influenced the efforts of the Federal German coal industry to decrease the number of fatal accidents. Whereas between 1960 and 1979 the output per shift in hard-coal mining increased from 2057 to 4024 kg/man and shift, the frequency of fatal accidents decreased from 0.61 to 0.40 accidents per million performed working hours, i.e. by some 35%. Severe accidents that were formerly typical for coal-mining, such as underground explosions, are very rare today and, moreover, have caused no injuries in recent years. The same is true of underground fires [9]. It is a great achievement that life expectancy of miners suffering from silicosis is today equal to average life expectancy [10].

Relating the actual occupational deaths and fatal diseases to GW(e)·a is only convincing at first. On second thoughts, this method is highly questionable as coal-miners will have to find work somewhere else if not in coal mines. None of the alternatives is without risk. Consequently only the greater than normal rate of coal-miners' risk, measured against the mean occupational risk of workers, can be ascribed to the coal fuel cycle. Furthermore, those risks must be related to working hours. The coal-miners' risk in this sense is exceeded by the occupational risks of inland navigators, butchers, road workers, civil engineers, carpenters, roof-tilers and even ore miners, who, regarding accidents, can be compared with uranium miners.

2.2. Transportation

It is true that the coal supply, with its relatively large masses, adds more to the general transportation risks then the nuclear power cycle. On the other hand, it cannot be claimed that railway accidents occurring during the transport of coal is a typical risk connected with the coal fuel cycle. This kind of risk definitely has its own significance, the more so as the coal industry has no opportunity to reduce it. Therefore, in risk comparison, transportation risks should at least not have the same weighting as occupational risks or risks caused by emissions. It would be more justifiable to ascribe transportation risks to those who are responsible for railways and crossings.

Great efforts have been made in the Federal Republic of Germany in recent years to eliminate railway crossings (the main source of risk) by constructing flyovers and installing highly efficient signals. Hence we cannot transfer US data on this matter without caution. Moreover, the average haul is much shorter in Europe than in the USA. Finally, all estimates concerning transportation risks are very sensitive to the assumed mode of transport [11].

2.3. Public fatal diseases

The main contributions to risks associated with the use of coal are said to arise from environmental impacts of air pollution caused by coal-burning. Here the four risk studies described above rely partly on their predecessors, partly on three other reports: the CHESS study [12] and the reports of Lave and Seskin [13] and Winkelstein et al.[14].

It is well known that the CHESS study can no longer be considered a reliable source of data after having been examined by the US House of Representatives [15], when CHESS was found – among other points – to have "too many inconsistencies in the data" as well as "large data uncertainties or errors". Apparently the study was designed "to provide quantitative support for policy decisions" that EPA had previously made. Nevertheless, these data survive in some important risk-comparison studies, where they represent the upper limits of the range of estimates.

Later risk-comparison studies, e.g. L.D. Hamilton [16], Lave and Seskin [13] and Winkelstein et al. [14] used newer sources. None of these sources examined or found correlations between human health and pollutants other than SO_2, sulphates and total suspended particles. To these pollutants the above figures for public fatal diseases must be ascribed. Yet many authors [17- 22] have criticized the conclusions of these studies on the grounds that other important factors, such as smoking habits, exposure at home or at work, and other behavioural and social factors, have not been collected or have been poorly measured. In the FRG, the Federal Government re-examined its ground-level standards on air pollution in 1978 [23] on the basis of expert opinions (see Section 3.1. below). The experts had stated that up to concentrations of 0.14 mg $SO_2 \cdot m^{-3}$ and 0.08 mg $NO_2 \cdot m^{-3}$ and 0.15 mg total suspended particles per m^3 (average per year) it is ensured that no harmful effects on human health can occur, synergistic effects included. Since these standards (as well as the corresponding short-term standards) are met practically everywhere in the FRG today, it is established that in our country these pollutants can no longer cause any adverse health impacts.

In addition, the findings of Lave and Seskin appear puzzling from the pulmonological point of view as no statistical correlation between asthma and air pollution could be found, whereas such correlation with all kinds of cancer and heart diseases are reported to exist. According to experience, adverse health effects should have been observed, above all, as broncho-pulmonial diseases.

The presumption that sulphate in air will contribute to observed health effects [16] does not seem convincing. It is highly improbable that sulphates may affect human health as an acid, because the lung possesses a highly efficient hydrocarbonate-carbonate buffer system. It is also suspected that the metal ions might be harmful too; however, airborne sulphates consist mainly of ammonia-sulphates or ammonia-hydrogen-sulphates [24]. These compounds can be regarded as harmless so far as concentrations in urban areas are concerned.

Many authors refer to other emissions from coal-fired power stations, e.g. trace metals, hydrocarbons and radioactivity; and, of course, there is the question of CO_2. It is true that these elements or compounds can show great toxicity or produce other impacts in high concentrations. Yet no valid epidemiological study exists which supports the assertion that actual ground-level concentrations of these substances can cause significant harm. It is widely accepted that if these air pollutants have any effect at all it will be extremely small and not quantifiable. A recent German study demonstrates that lung cancer incidence in urban and industrialized regions is not higher than in clean areas [25].

This does not mean that the Federal German coal industry is not concerned about the health impact of these substances. Together with medical science, we feel the need for further research. Acting on our own responsibility we are supporting and financing research programmes in order to clear up the uncertainties in this field. This is also true of the CO_2 question, which in essential points remains open.

3. SUGGESTIONS

Considering the pros and cons, one feels lost in this vast field of scientific disagreement. Yet there appears to be an urgent need for a new method of assessing the relationship between air pollution and human health. Obviously it is not enough to gather all opinions offered and sum up the existing uncertainty in a wide range of estimates.

3.1. Assessing health impacts of air pollution

We propose a method that was applied in 1978 during the preparation of the intended amendment of the Federal Immission Control Law. In this connection it was necessary to determine under which conditions it is ensured that the main pollutants, such as SO_2, NO_x and particulate matter, can reasonably have no harmful effects. These conditions were to be expressed in terms of legally binding ground-level standards. For this purpose the Federal Minister of the Interior (who intended to tighten the clean-air rules) conducted a hearing of some 35 medical experts. For a week the experts made their statements, while Federal and State authorities and even industry had the opportunity to ask questions. The hearing had been prepared by a questionnaire and a detailed study giving a survey of world-wide literature. The whole proceedings were taken down in a literal protocol [26]. Agreement was reached, with the result that the ground-level standards hitherto in force were to be considered "on the safe side" – synergistic effects included.

We think that the only way to achieve what is of the utmost importance in assessing health impacts of air pollutions is to evaluate thoroughly all relevant material for scientific reliability and to sort out those findings that cannot be regarded as well founded.

3.2. Continuing scientific uncertainty

The procedure described above fails when uncertainty or ignorance prevail in a certain area. This is true, for instance, of the CO_2 question. Here, in our opinion, any attempt at quantifying a risk is inappropriate. Even though no valid basis is available, an open scientific question is in no way comparable to the risk of a nuclear accident. The chance that science, after ten more years of research, will incline to a certain school of thought is not a quantifiable risk. It may turn out in the future that, unlike the nuclear risk, there is no (CO_2) problem at all. Scientific uncertainty cannot be disposed of by probabilistic calculations.

3.3. Value factor

Meanwhile it has been widely recognized that in risk analysis it is necessary to consider not only the likelihood and the consequences of an event but also the perception of these factors by those who are exposed to the risk. Deaths from a number of small everyday accidents or fatal diseases are not perceived to be equivalent to the same number of deaths caused by a major nuclear accident [2, 27, 28]; there are other important subjective factors which determine perception of risks. Therefore, if quantification of risk comparison is to continue, a value factor must be induced to make allowance for perception, the more so as Slovic thinks that perceived risk is quantifiable (Ref.[28], p.9).

3.4. Risk comparison applied within an energy system

Comparative risk assessment might prove more rewarding if confined to the problems within an energy system. Here the analysts should concentrate on identifying priorities of risks to be reduced. To a certain extent, comparative risk assessment of different energy systems has already produced effects in this direction. In view of the huge costs of emission control techniques, as well as of research work, a measure for more reasonable assignment of means to the various environmental issues would be welcome.

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DISCUSSION

A.A. MOGHISSI: You consider the comparative risks associated with various energy sources in your paper. The regulatory agencies in the United States of America and in certain other countries have been accused of proceeding in an arbitrary manner on the grounds that they have not based their standards and regulations on scientific facts. I find it somewhat contradictory to oppose risk assessment activities while basing regulations on them. Could you comment on this contradiction?

J. SEELIGER: As we have questioned only the comparison of risks of different energy systems, I don't see that there is a contradiction. It is possible, in fact, to try to discover the scientific basis for regulatory restraints to be applied to a certain energy system and at the same time avoid comparing the scientific findings (or what are deemed to be such) with those of another energy system.

I.M. TORRENS: It is your belief, Dr. Seeliger, that the usual sort of comparative risk assessment between energy forms is not constructive and that risk analysis should more usefully limit itself to the risks at various stages of the fuel cycle of one energy form in order to find out where efforts to reduce these risks would yield greatest dividends. Another approach to this problem is to say that the usual comparative risk assessment does not go far enough and should include the risks of not providing the energy as well as the benefits associated with the improved living standards that the energy furnishes.

In fact, the data which a comparative risk assessment yields must undergo a further stage before they can be useful to the policy-maker – namely, an evaluation by suitable experts of their validity and usefulness in the light of the above observations. This is comparable to the 1978 German Policy Review mentioned in your paper. Problems occur when this step is omitted or when the risk comparisons are used by special interest groups to further their objectives. In summary, comparisons are not odious, but need to be handled with kid gloves!

ОБЛУЧЕНИЕ НАСЕЛЕНИЯ ЗА СЧЕТ ЕСТЕСТВЕННЫХ РАДИОНУКЛИДОВ, СОДЕРЖАЩИХСЯ В АТМОСФЕРНЫХ ВЫБРОСАХ УГОЛЬНЫХ ЭЛЕКТРОСТАНЦИЙ

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Abstract – Аннотация

EXPOSURE OF THE PUBLIC TO RADIATION FROM NATURAL RADIONUCLIDES IN THE ATMOSPHERIC RELEASES FROM COAL-FIRED ELECTRICAL POWER STATIONS.

The occurrence of radiation doses to the public living within a radius of 50 km of a coal-fired power station due to the release of natural radionuclides into the atmosphere is discussed. The following ways in which doses occur are considered: inhalation of radioactive gases and aerosols, intake of radionuclides with foodstuffs and external irradiation by radioactive materials settling on the ground. Calculations are performed for a 1000-MW power station. Annual doses and the expected doses over 50 years from annual releases and from continuous release over this period are obtained. The effective dose equivalents are 7.1×10^{-2} mrem, 0.65 mrem and 19.9 mrem, respectively. A comparison is made with irradiation from the atmospheric releases of a nuclear power station of the same capacity.

ОБЛУЧЕНИЕ НАСЕЛЕНИЯ ЗА СЧЕТ ЕСТЕСТВЕННЫХ РАДИОНУКЛИДОВ, СОДЕРЖАЩИХСЯ В АТМОСФЕРНЫХ ВЫБРОСАХ УГОЛЬНЫХ ЭЛЕКТРОСТАНЦИЙ.

Рассмотрено формирование дозовых нагрузок на население, проживающее на территории радиусом 50 км вокруг электростанции, работающей на угле, за счет выбросов в атмосферу естественных радионуклидов. Учтены следующие пути формирования доз: ингаляция радиоактивных газов и аэрозольных частиц, поступление радионуклидов с пищевыми продуктами и внешнее облучение за счет радиоактивных веществ, осевших на поверхность почвы. Расчеты выполнены для электростанции мощностью 1000 МВт. Получены значения доз за год, ожидаемой дозы за 50 лет от годового выброса и за 50 лет от непрерывного выброса в течение этого периода. Эффективные эквивалентные дозы соответственно равны $7{,}1 \cdot 10^{-2}$ мбэр, 0,65 мбэр и 19,9 мбэр. Дано сравнение с облучением за счет атмосферных выбросов атомных электростанций такой же мощности.

Энергетика, являющаяся основой развития всех отраслей хозяйства, характеризуется наиболее высокими темпами роста производства. Тепловые электростанции (ТЭС), потребляющие различные виды органического топлива (каменный уголь, горючий сланец, нефть, газ), являются в настоящее время главными производителями электрической энергии. В частности, в СССР их доля в общем производстве энергии составляла в 1980 г. около 80% [1].

ТАБЛИЦА 1. СОДЕРЖАНИЕ НЕКОТОРЫХ ЕСТЕСТВЕННЫХ РАДИОНУКЛИДОВ В ЗОЛЕ КАМЕННОГО УГЛЯ, пКи/г

Страна, месторождение угля	^{226}Ra	^{210}Pb	^{232}Th	^{40}K	^{210}Po	^{228}Ra	^{238}U	Ссылка на литературу
СССР								
Донбасс	3,5	4,4	2,6	31	4,4*	3,1*	15*	Собственные данные
Экибастуз	1,5	2,4	0,5	5,7	–	–	–	Собственные данные
Кузбасс	3,1	3,1	0,8	18,5	–	–	–	Собственные данные
Караганда	1,6	1,2	1,1	13,3	–	–		Собственные данные
Подмосковье	4,8	2,0	1,6	8,7	–	–	–	Собственные данные
США	0,4-3,1	17,3	2,6	11,0	–	–	–	[5]
Австралия	14,0	–	–	–	–	–	–	[5]
Польша	0,6-6,4	4,4-6,7	0,18-0,22	22,5	–		–	[5]
Венгрия	0,6-15,0	–		–	–	–	–	[5]

* Расчетные данные.

Вместе с тем, энергетические предприятия являются одним из основных источников загрязнения окружающей среды и, по-видимому, будут оставаться таковыми в ближайшем будущем. При этом вредное воздействие энергетических объектов на окружающую среду и здоровье человека перестало носить локальный характер и превратилось в фактор глобального значения. Отсюда понятна необходимость иметь реальное представление об уровнях возможного вредного воздействия на окружающую среду и человека, которое не ограничивается традиционно рассматриваемыми химическими факторами (CO_x, NO_x, SO_2, пыль, токсические металлы, полициклические ароматические углеводы и т.д.), а включают в себя также и воздействие естественных радионуклидов (ЕРН), содержащихся в атмосферных выбросах ТЭС. Реальная и, по возможности, полная оценка такого воздействия тем более необходима, т.к. чисто психологические факторы зачастую приводят к тому, что неблагоприятное влияние выбросов ТЭС признается общественным мнением менее опасным по сравнению с выбросами атомных электростанций (АЭС). Такая позиция, не подкрепленная достаточно точными количественными оценками, тормозит развитие атомной энергетики, компенсирующее истощающиеся запасы органического топлива. Задача ученых, на наш взгляд, заключается не только в том, чтобы убедить общественное мнение в необходимости и неизбежности развития атомной энергетики, но и в том, чтобы дать объективную оценку возможного неблагоприятного воздействия на окружающую среду и человека отходов энергетических объектов, в частности атмосферных выбросов.

В настоящей работе рассматривается радиационное воздействие на население ЕРН, содержащихся в атмосферных выбросах ТЭС, работающих на угле и являющихся основным источником электрической энергии. Количество ЕРН, поступающих в атмосферу с летящей золой, зависит от их содержания в исходном топливе, условий его сгорания, эффективности золоулавливающих систем и от других параметров. Все приводимые в работе расчеты выполнены для электростанции современного технологического уровня мощностью 1000 МВт, потребляющей в сутки 12000 т угля, зольность которого составляет 25%. Следует отметить, что зольность угля в СССР колеблется в весьма широком диапазоне – от 7-10% до 40-45%. Количество шлака, образующегося при сжигании угля, составляет 20%, эффективность золоулавливания принята равной 98,5% [2]. Все использованные в работе характеристики топлива относятся к углю, добываемому в Донбассе, и представлены в табл.1.

Для оценки облучения населения за счет атмосферных выбросов ТЭС были приняты определенные допущения. Следует предварительно отметить, что целью расчетов было определение средних доз облучения населения, проживающего вокруг станции, а не фактическое распределение доз. Это связано с тем, что в основе оценки возможных стохастических эффектов лежит концепция беспорогового действия ионизирующей радиации и линейной зависимости между дозой и эффектом. В этом случае частота эффекта определяется как "произведение полученной средней дозы на частоту интересующего биологического эффекта на единицу дозы" [3], что позволяет избежать необходимости определения фактической дозы и численности групп лиц, подвергшихся воздействию конкретных уровней доз. Отсюда вытекает основное принятое в расчетах допущение – атмосферные выбросы распределяются равномерно в пределах определенной территории. Размер территории определен из условия, что выход радиоактивных веществ за ее пределы не превышает 2-3% от общего выброса. Используя закон убывания концентраций с расстоянием $(R/R_M)^{-1,5}$ [4], было определено, что при высоте трубы 200 м радиус рассматриваемой территории должен составлять примерно 50 км.

Рассмотрены три варианта формирования доз:

– годовой выброс радионуклидов, доза формируется за этот же год;

– годовой выброс радионуклидов, доза формируется за 50 лет (ожидаемая доза);

– непрерывный выброс радионуклидов в течение 50 лет, доза – за этот же интервал времени.

При определении уровней облучения учитывались ингаляция радиоактивных газов (радон и торон) и аэрозольных частиц, поступление радионуклидов с пищевыми продуктами, выращенными на загрязненной территории, а также внешнее облучение за счет оседания радиоактивной золы на поверхность почвы.

Внешнее облучение. Принималось, что вся выпавшая на поверхность почвы зола равномерно распределена в пределах глубины пахотного слоя, равного 20 см. Расчет мощности дозы на высоте 1 м от поверхности выполнялся по формуле [5]:

$$D = 1{,}58\,F_U + 1{,}6\,F_{Th} + 0{,}16\,F_K \qquad (1)$$

где D – мощность дозы внешнего облучения, мкрад/ч, F_U, F_{Th}, F_K – содержание соответствующих радионуклидов в почве, пКи/г.

Использование приведенной формулы предполагает, что все продукты распада ^{238}U и ^{232}Th находятся в радиоактивном равновесии. Однако это условие справедливо для горных пород и вносит определенную неточность при расчете доз от почвы. Кроме того, соотношение концентраций радионуклидов, в частности ^{238}U и ^{226}Ra, в летящей золе отличается от соотношения их в почве, установившегося в результате векового равновесия. Так, средняя концентрация ^{226}Ra в почве колеблется в диапазоне 0,1-1,1 пКи/г, а ^{238}U – 0,3-1,4 пКи/г [4, 6], т.е. соотношение близко к 1. А в летящей золе, как следует из табл.1, концентрация урана примерно в 4 раза выше концентрации радия. Если учесть, что около 99% вклада в мощность дозы за счет ЕРН семейства урана дают ^{214}Pb и ^{214}Bi [4, 5], т.е. продукты распада ^{226}Ra, то использование формулы (1) привело бы к завышению мощности дозы. Поэтому в дальнейших расчетах в формулу (1) вместо концентрации ^{238}U подставлялась концентрация ^{226}Ra. Аналогичный сдвиг соотношений концентраций следовало бы ожидать и для ^{232}Th и ^{228}Ra. Однако, ввиду отсутствия необходимых данных, этот возможный сдвиг концентраций не учтен.

Внутреннее поступление. Содержание ЕРН в рационе оценивалось по линейной модели, общий вид которой получен по натурным наблюдениям за загрязнением рациона в результате глобальных выпадений ^{90}Sr и ^{137}Cs [7]:

$$C_p = p_п F_п + p_в F_в \tag{2}$$

где C_p – содержание радионуклида в суточном рационе, пКи/рацион, $F_п$ и $F_в$ – содержание радионуклидов в почве мКи/км^2 и в выпадениях мКи/(км$^2 \cdot$ год), $p_п$ и $p_в$ – коэффициенты пропорциональности.

Значение коэффициента пропорциональности $p_п$ для радионуклидов, выбрасываемых с летящей золой, получены по литературным данным на основе их содержания в рационе и средней концентрации в почве [6, 8-10]. Что касается коэффициента воздушного загрязнения $p_в$, то определение его реального значения не представлялось возможным. Поэтому, по аналогии с данными, полученными по глобальным выпадениям ^{90}Sr и ^{137}Cs [11], было принято, что этот коэффициент в 10 раз выше почвенного. По-видимому, такой подход завышает значения $p_в$, поскольку биологическая доступность радионуклидов, содержащихся в летящей золе, скорее всего, ниже, чем доступность глобальных радионуклидов. Это же замечание относится и к $p_п$, т.к. биологическая доступность осевшей на почву золы, по-видимому, ниже, чем кларковых ЕРН.

Накопление ЕРН в организме оценивалось по экспоненциальной модели:

$$Q(t) = \frac{C_p \cdot f}{\lambda_{эфф.}}(1 - e^{-\lambda_{эфф.} t}) \tag{3}$$

где $Q(t)$ – содержание радионуклида в органе, пКи/орган, f – коэффициент перехода радионуклида из рациона в орган, $\lambda_{эфф.}$ – постоянная эффективного полувыведения из органа, t^{-1}. Формула (3) справедлива при постоянном C_p. Однако содержание ЕРН в почве меняется по закону:

$$F_п(t) = \frac{F_в}{\lambda_п}(1 - e^{-\lambda_п t}) \tag{4}$$

где $F_п(t)$ – содержание радионуклида в почве, мКи/км^2, $F_в$ – уровни годовых выпадений, мКи/(км^2 · год), $\lambda_п$ – постоянная полуочищения почвы, равная 0,01 год$^{-1}$ [12]. В соответствии с этим меняется и загрязненность рациона. Поэтому, окончательно содержание ЕРН в органе выражается следующим соотношением:

$$Q(t) = F_в f(1 - e^{-\lambda_{эфф.}t})\left(\frac{p_в}{\lambda_{эфф.}} + \frac{p_п}{\lambda_п \lambda_{эфф.}}\right) - \frac{F_в p_п}{\lambda_п(\lambda_п - \lambda_{эфф.})}(e^{-\lambda_{эфф.}t} - e^{-\lambda_п t}) \qquad (5)$$

Обозначения те же, что и в предыдущих формулах. Численные значения коэффициентов f и $\lambda_п$ заимствованы из работ [4, 6]. При рассмотрении перорального поступления не учитывался ^{40}K, т.к. он сопутствует стабильному калию, накопление которого в организме находится под контролем гомеостатических механизмов. Следовательно, и накопление ^{40}K будет ограничено физиологическими потребностями организма.

При оценке ингаляционного поступления аэрозолей рассматривались все радионуклиды, представленные в табл. I. Принималось, что наиболее вероятный активный масс-аэродинамический диаметр составляет 5 мкм [2]. Концентрация аэрозолей в воздухе определялась из предположения, что оседание летящей золы происходит со скоростью 0,01 м/с [2, 4, 13]. Тогда концентрация будет равна:

$$a = \frac{A}{V} \qquad (6)$$

где a – концентрация аэрозолей в воздухе, пКи/м^3, A – выброс активности, Ки/с, V – рост объема, в котором разбавлялся выброс, м^3/с.

Для расчета дозовой нагрузки на легкие и другие органы при ингаляционном поступлении ЕРН использовалась модель легочного обмена, предложенная рабочей группой МКРЗ [14]. Возможное обогащение мелких фракций летящей золы не учитывалось. Как отмечается в докладе НКДАР ООН [4], максимальный коэффициент обогащения наблюдается для фракций с размером частиц, равным 1-2 мкм, и для различных радионуклидов лежит в пределах 1,2-5 по сравнению с фракцией с размером частиц, равным 15 мкм. Поэтому оценка доз, приходящихся на различные органы за счет ингаляционного поступления без учета фактора обогащения примерно в 2 раза занижает величину дозы, поскольку результаты определения ЕРН в золе, представленные в табл. I и использованные в расчетах, получены по золе, отобранной с электрофильтров, а фактическая дисперсность летящей золы меньше. С другой стороны, предполагалось, что поступающая в легкие зола полностью растворима, что в свою очередь приводит к несомненному завышению дозы, оценить которое не представлялось возможным. Можно лишь предположить, что с учетом всех допущений окончательные значения доз лишь не намного превышают реальные.

Расчет доз облучения, обусловленных ингаляцией радона и торона выполнены с учетом непосредственно выбрасываемых в атмосферу радиоактивных веществ и эксхаляции из почвы в результате оседания ^{226}Ra и ^{232}Th. Предполагалось, что радон и торон полностью поступают в атмосферу при сжигании угля. Поскольку период полураспада торона составляет лишь 55 с, он практически не достигает приземного слоя воздуха, и основной вклад в дозу дает торон, эксхалирующий из почвы. Выброс в

ТАБЛИЦА II. ОБЛУЧЕНИЕ НАСЕЛЕНИЯ ЗА СЧЕТ АТМОСФЕРНЫХ ВЫБРОСОВ ТЭС В РАДИУСЕ 50 км, МОЩНОСТЬ СТАНЦИИ 1000 МВт

Условия облучения	Пути воздействия	Легкие	Гонады	Костный мозг	Эндостальные клетки	Все тело (эфф. экв. доза)
Выброс за год, доза за год, мбэр	Внешнее	$0,6 \cdot 10^{-3}$	$0,6 \cdot 10^{-3}$	$0,6 \cdot 10^{-3}$	$0,6 \cdot 10^{-3}$	$0,8 \cdot 10^{-3}$
	Ингаляция	$210,0 \cdot 10^{-3}$	$120,0 \cdot 10^{-3}$	$85,0 \cdot 10^{-3}$	$20,0 \cdot 10^{-3}$	$66,0 \cdot 10^{-3}$
	Пищевой	$6,3 \cdot 10^{-3}$	$6,3 \cdot 10^{-3}$	$12,7 \cdot 10^{-3}$	$21,7 \cdot 10^{-3}$	$4,5 \cdot 10^{-3}$
	Всего	$216,9 \cdot 10^{-3}$	$126,9 \cdot 10^{-3}$	$98,3 \cdot 10^{-3}$	$42,3 \cdot 10^{-3}$	$71,3 \cdot 10^{-3}$
Выброс за год, доза за 50 лет, мбэр	Внешнее	$5,0 \cdot 10^{-2}$	$5,0 \cdot 10^{-2}$	$5,0 \cdot 10^{-2}$	$5,0 \cdot 10^{-2}$	$6,3 \cdot 10^{-2}$
	Ингаляция	$140,0 \cdot 10^{-2}$	$25,2 \cdot 10^{-2}$	$20,9 \cdot 10^{-2}$	$7,3 \cdot 10^{-2}$	$25,7 \cdot 10^{-2}$
	Пищевой	$25,6 \cdot 10^{-2}$	$25,1 \cdot 10^{-2}$	$73,8 \cdot 10^{-2}$	$503,0 \cdot 10^{-2}$	$33,3 \cdot 10^{-2}$
	Всего	$170,6 \cdot 10^{-2}$	$55,3 \cdot 10^{-2}$	$99,7 \cdot 10^{-2}$	$515,3 \cdot 10^{-2}$	$65,3 \cdot 10^{-2}$
Выброс за 50 лет, доза за 50 лет, мбэр	Внешнее	1,3	1,3	1,3	1,3	1,6
	Ингаляция	25,2	10,6	7,6	1,8	6,7
	Пищевой	8,8	8,8	25,6	176,0	11,6
	Всего	35,3	20,7	34,5	179,1	19,9

атмосферу радона оценивался из предположения о равновесии его с ^{226}Ra в угле, концентрация которого в угле по нашим данным составляет 2 пКи/г. Во всех расчетах доз, по всем путям формирования учитывалось не только количество ЕРН, непосредственно выбрасываемого в атмосферу, но и образующегося в результате радиоактивного распада предшественников, что особенно важно при расчетах ожидаемых доз. Дозовые коэффициенты для различных органов, использованные в работе, заимствованы из литературных источников [4-6, 15, 16]. Эффективная эквивалентная доза определялась по рекомендованной МКРЗ методике с учетом взвешивающих факторов [17]. Результаты расчетов представлены в табл. II.

Величина средней эффективной эквивалентной дозы за год от годового выброса ТЭС мощностью 1000 МВт составляет 0,07 мбэр и почти полностью (на 93%) обусловлена ингаляционным поступлением ЕРН. Ожидаемая за 50 лет от годового выброса активности доза равна 0,65 мбэр и на 50% определяется поступлением ЕРН с пищевыми продуктами. Эта доля возрастает при непрерывном выбросе в течение 50 лет до 60%. В данном случае величина дозы равна 20 мбэр.

Интересным представляется сравнение средних величин доз, формирующихся вокруг ТЭС и АЭС одинаковой мощности. По данным работы [18] средняя ожидаемая эффективная эквивалентная доза от годового выброса АЭС мощностью 1000 МВт (эл.) составляет в радиусе до 50 км 0,01 мбэр для реактора типа ВВЭР и 0,07 мбэр для реактора типа РБМК. Сравнение этих величин с аналогичными показателями для ТЭС свидетельствует о том, что радиационная нагрузка за счет выбросов ЕРН тепловыми электростанциями, работающими на угле, в 10-70 раз выше, чем за счет выбросов от АЭС. Следует отметить, что сравнение не совсем точное: ожидаемые дозы от выбросов АЭС рассчитаны для бесконечного периода времени, т.е. до полного распада радионуклидов, а доза от ТЭС -- лишь за 50 лет. Корректное сравнение дозовых нагрузок свидетельствовало бы об еще более сильном радиационном воздействии ТЭС на население, проживающее в районе расположения станции.

Таким образом, можно с очевидностью утверждать, что в условиях нормальной работы выбросы тепловых электростанций, работающих на угле, обуславливают значительно большее вредное воздействие на проживающее вокруг них население, чем выбросы атомных электростанций.

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Session IV

ENVIRONMENTAL TRANSPORT AND TRANSFORMATION OF DISCHARGES FROM ENERGY SOURCES INFLUENCING THE ESTIMATION OF DOSE TO PERSONS

Chairman
J. SCHWIBACH
Federal Republic of Germany

A REVIEW OF MAJOR LONG-RANGE TRANSPORT AIR QUALITY SIMULATION MODELS

*Summary**

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Abstract

A REVIEW OF MAJOR LONG-RANGE TRANSPORT AIR QUALITY SIMULATION MODELS: SUMMARY.

Regional analysis of air quality and precipitation chemistry data in North America and Western Europe has provided strong evidence for the long-range transport and wet removal of sulphur pollution from high emission density areas. The inadequacy of local scale models to simulate sulphate concentrations and wet sulphur depositions in relatively pristine areas has led to the development of 'long-range transport (LRT) models'. The Clean Air Act Amendments and United States-Canada Memorandum of Intent on Transboundary Air Pollution have given considerable impetus to the development, evaluation and application of LRT models to national and international control strategies. While there are currently no 'guideline or regulatory' LRT models, they have been and are being used in preliminary assessments. The author, in his presentation, described the major LRT models, the development of so-called transfer matrices relating source and receptor areas, and the status of air quality and precipitation chemistry data bases for establishing baseline conditions and evaluating the models. The presentation also included an analysis of transfer matrices based on eastern USA, State and Canadian Province SO_2 emissions and their application to the issues of interstate and international transport of SO_2 and SO_4.

DISCUSSION

I.M. TORRENS: What was the magnitude of the variance in the numbers of the transfer matrix as predicted by the different transport models?

B.L. NIEMANN: The standard deviations over the five Phase-I model transfer matrices ranged from 0.01 to 6.83 $kg \cdot s \cdot ha^{-1} \cdot a^{-1}$ per unit emission $Tg \cdot s \cdot a^{-1}$, while the average transfer matrix coefficients ranged from 0.00 to 6.55 in the same units.

* The complete text was not submitted for publication.

G. HALBRITTER: What kind of meteorological data base is used for the trajectory models you presented? Is it simply a one-year data base involving the synoptic data of one selected year, or do you use the data from several years to obtain a representative year?

B.L. NIEMANN: The trajectory models will use both 1978 and 1979 data for the model evaluation work, and then up to five years (1975–1979) of data to produce a representative transfer matrix. The annual average-type models use statistics based on ten years of meteorological data. The representativeness or variability of the meteorological periods used in both the trajectory and annual average models have been analysed using stagnation episodes, precipitation and temperature data as well as air quality and precipitation chemistry episodes.

K.G. VOHRA: In applying models such as yours, it is important to consider variations in the transformation rate of SO_2 to sulphate which depends on other trace constituents in stack releases and ambient air. For example, SO_2 transformation can occur at a very rapid rate when stack air mixes with urban air containing pollutants released by automobiles. Photochemical transformation of SO_2 in such conditions can be very much greater than when stack air mixes with clean air.

B.L. NIEMANN: Your comment is justified. However, the models described in the paper were applied to the simulation of regional SO_2 and SO_4 and not individual plumes or urban sources. Some of the regional models used by the United States-Canada Work Group 2 take into account diurnal and seasonal variations in the SO_2 to SO_4 transformation rate. However, the SO_2 to SO_4 chemistry is highly generalized in current regional models by using rate constants averaged over all sources and the relevant time and space scales.

DEVELOPMENT AND APPLICATION OF TERRESTRIAL FOOD-CHAIN MODELS TO ASSESS HEALTH RISKS TO MAN FROM RELEASES OF POLLUTANTS TO THE ENVIRONMENT*

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Abstract

DEVELOPMENT AND APPLICATION OF TERRESTRIAL FOOD-CHAIN MODELS TO ASSESS HEALTH RISKS TO MAN FROM RELEASES OF POLLUTANTS TO THE ENVIRONMENT.

The paper reviews development and application of mathematical models used to predict the terrestrial food-chain transport of pollutants of potential importance to human health. A distinction is made between models developed specifically for assessment applications and models which may function as research tools. Differentiation is also made between models whose structure is based on steady-state relationships among food-chain compartments and dynamic models developed to simulate food-chain and pollutant kinetics. The strengths and weaknesses of these models are related to the needs of the model-user, the availability of relevant data for parameter quantification, and the feasibility for model validation. For assessment purposes, an optimum level of structural complexity will be achieved when all parameters are readily measurable and predictive error due to unforeseen correlations among parameters is small. The optimum level of simplification, however, will be determined by model validation results and the ease of model implementation. Most examples are derived from models used to assess the terrestrial food-chain transport of radionuclides because assessment methodologies for other types of pollutants are only at an early stage of development. It is concluded that current limitations in parameter quantification and model validation will probably restrict most assessment applications of terrestrial food-chain models to a type of screening calculation. However, once pollutant releases actually occur, environmental monitoring will be necessary to ensure that potential model misprediction does not result in unacceptable consequences.

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1. INTRODUCTION

Terrestrial food chains, because of their dietary inputs to human populations, are of particular importance in the assessment of health impact from pollutant releases to the environment. Transport through food chains may result in pollutant concentrations which exceed background or natural levels and thus increase the risk to human health. Although transport and bioaccumulation of pollutants occurs in aquatic as well as terrestrial systems, our attention is focused on the terrestrial environment because 1) the atmosphere is a major recipient and disperser of pollutant releases, 2) the application of toxic chemicals directly to terrestrial food products for pest control is wide spread, and 3) a major portion of the human diet is derived from agriculture.

Terrestrial food-chain transport starts primarily with reactive gases or particulates that are readily deposited on and retained by vegetation. The deposition of potentially soluble pollutants onto soil may also result in food-chain contamination through incorporation of these pollutants into edible plant tissue through root uptake. Even for relatively insoluble pollutants, the soil may function as a secondary source of pollutants in terrestrial food chains long after direct releases to the atmosphere or water have ceased [1,2,3,4]. When vegetation contamination has occurred, pollutants may then be available for direct human consumption or transported to food products derived from grazing animals.

For the assessment of environmental or human health impacts, prediction of food chain transport is essential. Food-chain transport will not only affect the estimate of health risk to man, but it will also influence the assessment of compliance with regulatory standards and the determination of the relative importance of various pollutants. The determination of the relative importance of pollutants is especially critical to the evaluation of pollution abatement strategies, including site selection and decisions concerning alternative technologies.

One very useful technique for assessing the consequences of food-chain transport of pollutants is through the development and application of mathematical models. Figure 1 represents a conceptual terrestrial food-chain model[5]. A mathematical model quantifies the components of the food chain and their interactions. In such a model, food-chain components are referred to as "compartments" or "state variables" and the interactions as "transfer coefficients." The transfer coefficients and any scaling factors which relate them to the state variables are referred to as model parameters. In a mathematical model the amount of a pollutant in one compartment may be predicted by defining the relationships between this compartment and all

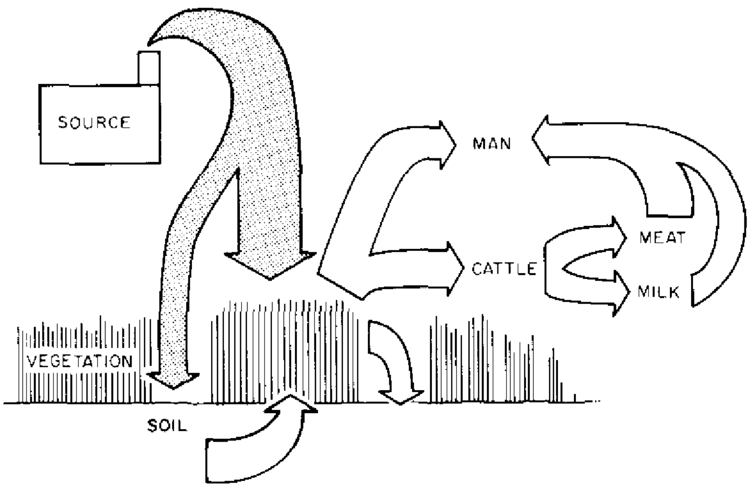


FIG.1. A conceptual terrestrial food-chain model for the assessment of health risks to man and releases of pollutants to the environment (from Garten [5]).

other model compartments and by defining the amount of pollutant input to the system. This predictive capability makes mathematical models ideal tools for decision making in the assessment of pollutant releases to the environment.

At Oak Ridge National Laboratory we have obtained a considerable depth of experience with both the development and application of terrestrial food-chain models. The purpose of this paper, therefore, will be to review, within the framework of our experience, the types of models used in the assessment of nuclear and chemical pollutants. We will also discuss the strengths and weaknesses of these models and will recommend directions for future research to improve application of the models and to increase confidence in model predictions. A special emphasis will be placed on recommendations for the application and refinement of current models in situations, such as in developing countries, where available data are severely limited.

2. TYPES OF MODELS

2.1. Model Categories

Although various classifications of models are possible, we recognize two basic categories of models developed to study the fate of pollutants in terrestrial systems: assessment models and research models.

2.1.1. Assessment Models

The primary purpose for assessment models is to predict environmental concentrations of pollutants and associated health risks and thus serve as tools for decision making. In the absence of detailed data, assessment models may focus on only a few basic food types such as milk, beef and leafy vegetables, which comprise a major portion of the human diet and are considered major pathways of transport. In these models, underlying processes and mechanisms may be aggregated into general transfer coefficients that relate the concentration of a pollutant in one compartment of a food chain to another.

The predictions made by assessment models must be defensible. The model should accurately describe the system and be based on the most current knowledge of the system that it represents. Parameter values and model predictions should be capable of being validated through field testing. In the absence of results obtained through field validation, assessment models are often given an intentional degree of conservative bias to reduce the probability of underestimation. As will be discussed later, such a conservative bias is most useful for calculations designed to screen for pathways and pollutants that may be of potential importance to human health, regulatory standards or effluent design objectives.

2.1.2. Research Models

Research models often differ from assessment models in their objectives. Emphasis is often placed on the ability of the research model to simulate the processes affecting the behavior of the pollutant in the food chain rather than on the prediction of concentrations in foods of importance to humans. Thus, research models are usually developed at the process level, making them far more complex than is typical of assessment models.

The ability to field-test process-level parameters and model predictions is not as important a constraint for the development of research models. For the purposes of research, no efforts are made to include an intentionally conservative bias in model predictions. The development of research models may also not be constrained by the limitations of potential users. Therefore, adaptation of research models to assessments will usually be restricted to specific situations where assessment models do not exist or are not applicable, or when data are unavailable for quantification of key assessment model parameters.

2.2. Model Approaches

A variety of approaches can be taken in the development of terrestrial food-chain models. Basically, these approaches can be classified into two general types of models: quasi-equilibrium models and dynamic models. The quasi-equilibrium model is derived from the approach defined by the International Commission on Radiological Protection [6] as the "Concentration Factor (CF) Method" for the assessment of radionuclides, or the "environmental commitment concept" for the assessment of environmental mercury [7]. These approaches are based on steady-state relationships between food-chain compartments. Dynamic models, on the other hand, are based on system kinetics. These models can be designed to explicitly address the time-dependent behavior of pollutants in terrestrial food chains. The approach represented by dynamic models is referred to by the ICRP as the "Systems Analysis (SA) Method."

2.2.1. Quasi-Equilibrium Models

Quasi-equilibrium models assume that the concentration of pollutants in some compartments of terrestrial food chains are at steady state and concentrations in other compartments are changing with time. Maximum values occur either at the time of harvest or upon some pre-determined length of exposure of the terrestrial system to a continuously uniform or average rate of pollutant deposition. In an equilibrium situation, the sum of the rates of pollutants entering a compartment are equal to the sum of the rates of pollutants leaving the compartment, i.e. the concentration of the pollutant in the compartment of reference does not change with time (Fig. 2). Thus, the concentration in a given food product at equilibrium is determined through simple multiplication, whereby:

$$C_B^* = C_A^* \times P_{AB} \tag{1}$$

where

C_B^* = the equilibrium concentration of a pollutant in a receptor food-chain compartment B,

C_A^* = the steady-state concentration in a donor food chain compartment A, and

P_{AB} = the equilibrium concentration factor for the flow of pollutants between the donor A and receptor B food-chain compartments.

The quasi-equilibrium model assumes that some compartments of the model may not be at equilibrium; therefore, the concentrations in these compartments are corrected for the differences between the value obtained at equilibrium and the value attained

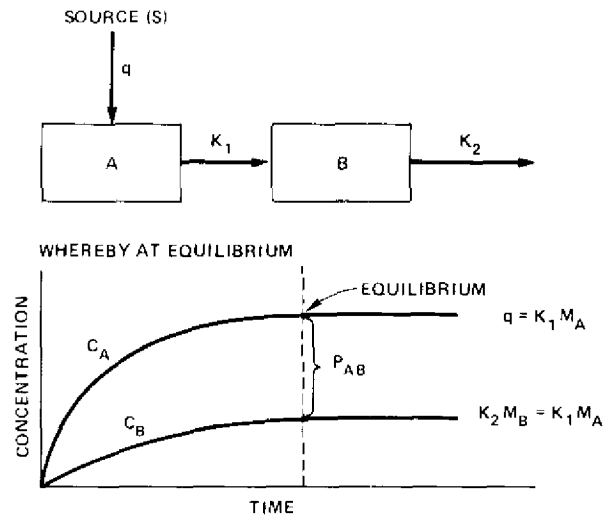


FIG. 2. A simplified food chain model at equilibrium where C_A and C_B are pollutant concentrations in food chain compartments A and B, P_{AB} is an equilibrium concentration factor, K_1 and K_2 are transfer rate constants, M_A and M_B are compartment masses, and the system is exposed to a pollutant from a given source at a continuous flux q.

at the time of harvest or at the end of a specified release period. Thus:

$$C_{B(t)} = C_A^* \times P_{AB} (1 - e^{-K_2 t}) \qquad (2)$$

where

$C_{B(t)}$ = the concentration of a pollutant in food-chain compartment B at time t subsequent to a continuous exposure to a pollutant,

K_2 = the effective first-order rate constant for the loss of pollutants from food-chain compartment B (time^{-1}), and

t = the time between initial exposure to the pollutant and harvest or the end of the release period.

When the reciprocal of the effective rate constant K_2^{-1} is very large with respect to t, equation (2) is simplified to

$$C_{B(t)} = C_A^* \times P_{AB} \times K_2 \times t \qquad (3)$$

When K_2^{-1} is very small with respect to t, equation (2) equals equation (1) and CB(t) equals C_B^*. Quasi-equilibrium models can be reduced to a series of multiplicative chains, and calculations with these models can be readily performed by hand, or at least with the use of a hand calculator.

2.2.1.1. Derivation of Concentration Factors

Various methods are employed to derive equilibrium concentration factors [7]. The simplest method is to ratio pollutant concentrations in food-chain compartments when these compartment concentrations are in equilibrium with each other. For example,

$$P_{AB} = C_B^*/C_A^* \tag{4}$$

Other methods for determining concentration factors include the use of ratios of pollutant concentrations in food-chain compartments averaged over sufficiently long time periods,

$$P_{AB} = \overline{C}_B/\overline{C}_A \tag{5}$$

and ratios of infinite time-integrated concentrations obtained from acute exposure situations,

$$P_{AB} = IC_B/IC_A \tag{6}$$

The infinite time-integrated concentrations IC_A and IC_B can be estimated using kinetic information on the rate constants for the transfer of a pollutant from a donor to a receptor food-chain compartment and on the masses of these compartments [7]. Thus,

$$IC_A = \int_{-\infty}^{\infty} C_A(t)dt = \frac{Q_\circ}{(K_1 + K_3)M_A} \tag{7}$$

$$IC_B = \int_{-\infty}^{\infty} C_B(t)dt = \frac{K_1 Q_\circ}{K_2 M_B (K_1 + K_3)} \quad \text{and} \tag{8}$$

$$P_{AB} = \frac{K_1 M_A}{K_2 M_B} \tag{9}$$

where the structure of the food chain is

$$\begin{array}{ccccccc} S & \xrightarrow{Q_\circ} & A & \xrightarrow{K_1} & B & \xrightarrow{K_2} & C \\ & & \downarrow{\scriptstyle K_3} & & & & \\ & & D & & & & \end{array}$$

K_1, K_2, and K_3 are first-order rate constants, M_A and M_B are compartment masses, and the system is exposed to a single pulsed emission from source S of mass Q_o.

Experimentally, concentration factors are obtained most practically through either time-averaging or time-integrating measured concentrations in food-chain compartments. Equilibrium or steady-state relationships are observed mainly in controlled laboratory studies, but are difficult to confirm under field conditions. Kinetic data, such as rate constants, are typically derived only from situations following either an acute release of a pollutant or an abrupt termination of a prolonged release. Examples of the derivation of concentration factors using various methods can be found in reports by Ng et al. [8] and Saha [9].

Uncertainties associated with the derivation of concentration factors from system kinetics are related to the potentially large temporal and spatial variability of rate constants obtained from acute contamination experiments. In situations where input and loss-rate constants and compartment masses have been determined independently, there is also uncertainty introduced because of possible correlations among these parameters. In addition, equilibrium concentration factors are often derived from ratios of average concentrations of pollutants in soil, vegetation and animal-food products reported in the literature from unrelated references. For example, reported concentrations for a donor compartment of the food chain may be specific for completely different locations and time periods than are relevant for concentrations reported for receptor compartments of the food chain. Many food-chain transfer factors listed for stable elements in quasi-equilibrium models developed for radiological assessments have been derived in this manner [10,11]. The uncertainty associated with this method of parameter value derivation is also large because of the absence of correlations between donor and receptor compartments. When parameter values are so derived, it is easy to overlook situations that may result in significant bias in model predictions.

In the absence of data on the transfer of specific pollutants, concentration factors are often derived from collateral information on related substances having similar physical and chemical properties. The estimated behavior of long-lived radionuclides, for example, is often based on the known behavior of stable isotopes of the element. In addition to stable isotope analysis, chemical analogs are sometimes used to obtain an approximate estimate of transfer factors for radioactive and non-radioactive pollutants [8,12,13].

Most of the quasi-equilibrium models currently in use are similar in structure to those developed for the HERMES computer code for the assessment of radionuclide releases [14]. At Oak Ridge National Laboratory, a similar model has been incorporated into the AIRDOS-EPA computer code [15] which combines food-chain transport with atmospheric dispersion and dosimetry. Versions of this model have also been developed for the assessment of non-radioactive pollutants [16,17]. In these models, equilibrium concentration factors have been generalized somewhat by implicitly expressing them as a function of numerous subparameters (Fig. 3).

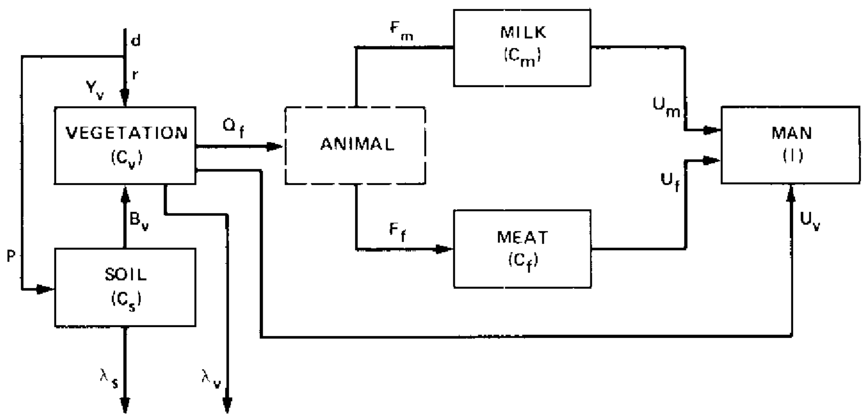
2.2.1.2. Limitations of Quasi-Equilibrium Models

Quasi-equilibrium food-chain models have a limited scope of application. Predicted food-chain concentrations are maximum or steady-state values resulting from a chronic, uniform release of pollutants into the environment. For non-uniform releases, quasi-equilibrium methods can be used if sufficiently long averaging times are selected [7]. For acute releases, the use of quasi-equilibrium models will result in estimates of time-integrated food concentrations, provided that average environmental conditions are prevailing at the time of discharge. Modifications of concentration factors will be necessary, however, to account for conditions that may actually exist at the time of an acute release which may deviate considerably from the expected average [6]. Nevertheless, in those circumstances where changes in the rate of exposure to pollutants is important for decisions, models must be developed to account for the dynamic behavior of pollutants. This requires data on the time-dependent rates of biophysical uptake, transfer, elimination and chemical transformation of pollutants among the various interconnected compartments of terrestrial food chains. Quasi-equilibrium models are not suited for such calculations.

2.2.2. Dynamic Models

In order to improve the ability to predict the time dependency of pollutant transfer through terrestrial food chains, as well as to improve the state of knowledge about the flow of materials through ecological systems, recent emphasis has been on the development of dynamic models [21,22,23]. Dynamic models attempt to conceptually approximate the kinetics of a real system. The advantage of dynamic models is that they may be applied to any type of release (acute, intermittent, continuous or any combination of these).

In their simplest form, dynamic models are based on first-order kinetics with each input or loss term expressed as a rate constant multiplied by the concentration of a pollutant in a donor food-chain compartment. In this form, analytical solutions to the system of linear first-order differential equations are possible. However, as the expressions for individual transfers become more realistic, e.g. when the rate constants are replaced



Equations:

$$C_v = \frac{dr[1 - \exp(-\lambda_v t_v)]}{Y_v \lambda_v} + B_v C_s$$

$$C_s = \frac{d[1 - \exp(-\lambda_s t_s)]}{P \lambda_s}$$

$$C_m = C_v Q_f F_m$$

$$C_f = C_v Q_f F_f$$

$$I = (U_m C_m) + (U_f C_f) + (U_v C_v)$$

where:

C_v = the pollutant concentration in vegetation (μg/kg)

C_s = the pollutant concentration in soil (μg/kg)

C_m = the pollutant concentration in milk (μg/l)

C_f = the pollutant concentration in meat (μg/kg)

I = the intake rate of the pollutant by man via ingestion of food (kg or l/time)

d = the flux of the pollutant from a source (μg/m² · time)

r = the fraction of the flux intercepted by vegetation

Y_v = the vegetation biomass (kg/m²)

λ_v = the removal rate of the pollutant from vegetation (time⁻¹)

t_v = the exposure period for vegetation (time)

B_v = the soil to plant concentration factor

P = the effective soil surface density (kg/m²)

λ_s = the removal rate of the pollutant from the root zone of soil

t_s = the exposure period for soil

Q_f = the rate of vegetation ingestion by an animal (kg/time)

F_m = the milk to vegetation intake rate transfer coefficient (time/l)

F_f = the meat to vegetation intake rate transfer coefficient (time/kg)

U_m = the rate of milk consumption (l/time)

U_f = the rate of meat consumption (kg/time)

U_v = the rate of vegetation consumption (kg/time)

References proposing this model for pollutant assessment:

Fletcher and Dotson [14]

Baker et al. [18]

USNRC [10]

Brenchley et al. [17]

Moore et al. [15]

McDowell-Boyer and Baes [19]

Moghissi et al. [16]

IAEA [20]

FIG.3. A commonly used version of a quasi-equilibrium model developed for the assessment of radionuclides and other pollutants.

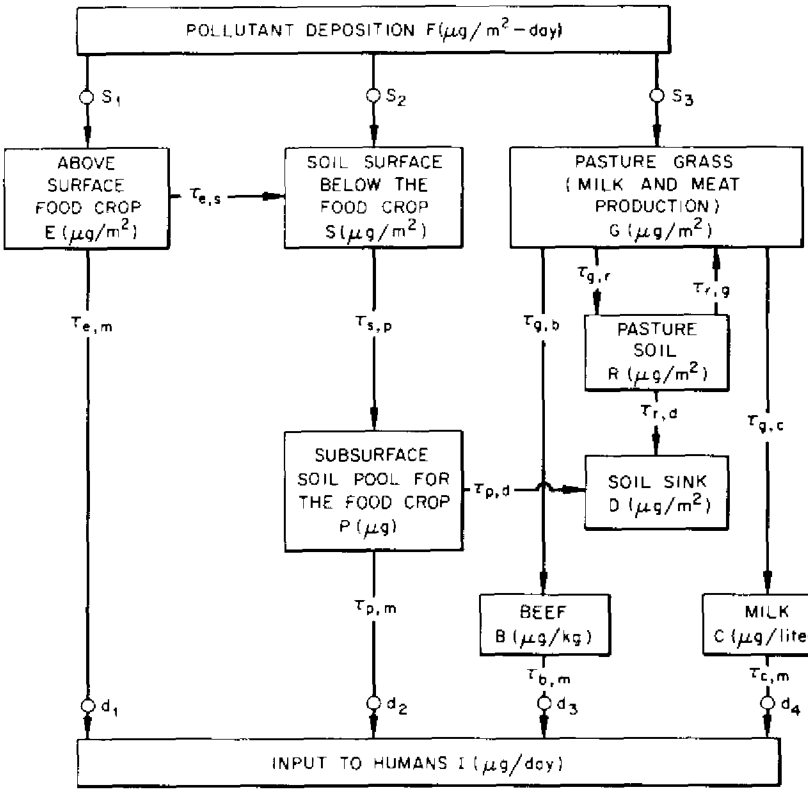


FIG. 4. Block diagram of a donor-controlled linear systems model for terrestrial food chains (Booth et al. [25]).

by time-dependent or concentration-dependent functions, analytical solutions are difficult to obtain. Numerical solution methods must then be called upon to simulate the time behavior of pollutants in food chains. Since these methods require extensive calculations, the model in its most useful form becomes an algorithm coded for a computer.

Dynamic terrestrial food-chain models have been developed as early as the late 1960s at Oak Ridge National Laboratory [24,25]. These were intended for the assessment of radiological impact resulting from either an acute or a chronic release situation, and were basically donor-controlled linear systems models (Fig. 4) composed of coupled first-order differential equations. Similar versions of this model have been developed for the terrestrial food-chain assessment of DDT and DDE [26] and of cadmium [27]. The rate constants developed for the input and elimination of pollutants from each food-chain compartment of these models are time-invariate.

The present state of development of dynamic food-chain models for nuclear assessments emphasizes the time dependence of the entire system to account for radionuclide daughter buildup with time, seasonal effects, and time-dependent rate constants requiring numerical solutions [28]. Terrestrial food chain models having similar objectives are being developed in the United Kingdom [29]. Dynamic food-chain models for non-nuclear assessments are also being developed to account for pollutant dynamics during food-chain transport and to simulate their behavior in the absence of empirical data derived from field observations [30].

Practically all food-chain models developed at the process level for research purposes are dynamic, as their objective is to mathematically simulate the behavior of pollutants in terrestrial systems as realistically as possible [21]. A linear model can be only an abstract conceptual approximation to the kinetics of a real system. In reality, terrestrial system kinetics will vary continuously over time as a function of diurnal and seasonal changes in the physical environment as well as changes in the abundance, distribution and composition of biota. In addition, terrestrial system kinetics may vary as a function of distance and direction from the point of release. The most realistic mathematical model, therefore, is a system of partial differential equations with coefficients varying as joint functions of time and location.

2.2.2.1. Limitations of Dynamic Models

The primary limitation associated with the development of dynamic models is the difficulty in obtaining appropriate data for quantifying model parameters. In most cases measurements have not been performed to determine the rate constants and transfer functions needed to simulate the dynamic behavior of pollutants in terrestrial food chains under temporally and spatially varying conditions. Even for simple linear systems models, values for rate constants must be ultimately determined from a time series of measurements. To obtain parameter values relevant to the assessment of acute releases, this time series of measurements must be repeated temporally and spatially. The experimental requirements to produce relevant dynamic transfer coefficients for even moderate-sized terrestrial food-chain systems may be prohibitively large depending on how the model is to be applied [31].

In the absence of kinetic data, input rates to compartments are sometimes derived from concentration factors and estimated elimination rates [25]. For example, the uptake rate coefficient for milk of a cow, $\tau_{g,c}$ (m^2/ℓ-day), was defined for the terrestrial food-chain model depicted in Fig. 4 as

$$\tau_{g,c} = \frac{C_{eq}}{G_{eq}} \tau_{milk} \tag{10}$$

where

C_{eq}/G_{eq} = equilibrium nuclide concentration ratio of milk to grass, and

τ_{milk} = excretion rate from the udder of a milk-producing animal.

Thus, $\tau_{g,c}$ is constant for a constant elimination rate. This method of parameter derivation is subject to the same sources of uncertainties affecting derivation of the equilibrium concentration factor. In addition, values obtained for the concentration factor and excretion rate may not be representative of comparable conditions.

For dynamic models initially developed as research tools, the experimental determination of individual transfer coefficients for the assessment of a given pollutant may be impractical. This is because dynamic process-level models often are dependent on parameters that cannot be readily measured in the field without undue manipulation of the system. Also, with an increasing number of compartments or state variables within the model, there is an increasing need to measure all parameters simultaneously, otherwise important information on correlations between transfer coefficients will not be included in the model. Without specification of parameter covariance, increasing model complexity will increase the sensitivity of model calculations to error propagation [32,33]. The difficulties encountered in obtaining pollutant-specific data through simultaneous measurements for parameters used in a process-level model of the type illustrated in Figure 5 can be readily appreciated [34].

Because of the limitations associated with data acquisition for detailed dynamic process-level models, some have proposed restricting model development to readily measurable variables even though such variables may include an aggregation of several apparently distinct mechanisms. This proposal is based on the argument that *at best, any model can only hope to be an abstraction of reality*. Therefore, when the formal structure of the model is reduced to include only the flow of pollutants between readily measurable major compartments of the terrestrial system, correlations among the various processes will be included implicitly in parameter measurements. This should improve the

STRUCTURAL COMPLEXITY AND PARAMETER REQUIREMENTS RENDER PROCESS LEVEL MODELS INAPPROPRIATE FOR POLLUTANT IMPACT ASSESSMENTS

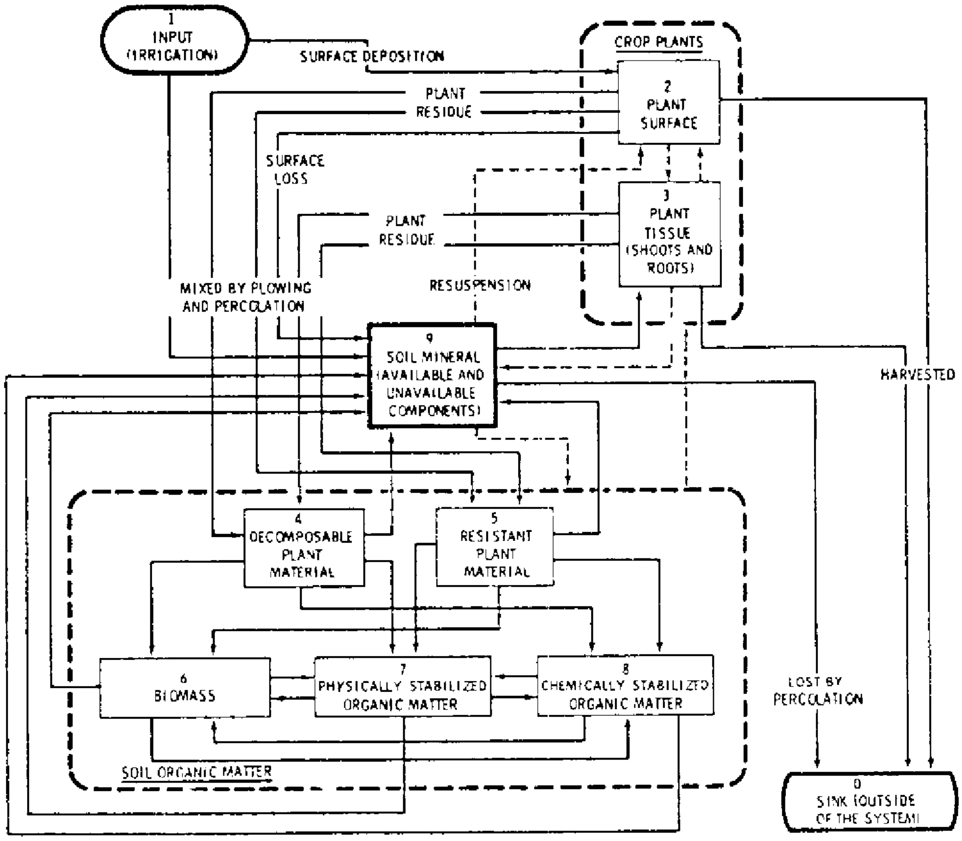


FIG.5. Compartmental model describing the long-term accumulation of radionuclides in crop plants (Schreckheise [34]).

predictive capability of the model for situations similar to those in which parameters have been measured. Enhancing the ability to measure parameters under field conditions should also improve the ability to field test or validate model predictions which is a most desirable step in determining the true utility of assessment models [35,36].

3. APPLICATION OF TERRESTRIAL FOOD-CHAIN MODELS IN ASSESSMENTS

3.1. Screening for Important Pollutants and Pathways

Realizing that quantified predictions are based on simplifying assumptions and generalizations of more complex real-world systems, a useful application of terrestrial food-chain assessment models is to screen for potentially important pollutants and pathways. Screening is usually performed as a part of a preliminary assessment. Typically it involves comparison of conservatively biased model predictions with established acceptable limits of pollutant concentrations in foods or exposures or doses to man.

Conservatism is employed to decrease the probability that model predictions will result in an underestimation of actual food-chain concentrations. Thus, when a predicted concentration is below a certain limit, the pollutant and pathway usually can be considered of negligible importance [37,38]. However, pollutants and pathways predicted to result in food-chain concentrations that encroach upon or exceed limiting values can only be designated as *potentially* important until further analysis of model assumptions and existing data or additional research can be conducted to improve the accuracy of model predictions. This means that in some cases ecologists and other environmental scientists will have to perfect better techniques than are presently available for measuring rate processes in the environment [39].

Conservatism for screening purposes may be assigned to the model structure, the selection of parameter values, and/or the application of a safety factor to either model predictions or established limits [37]. The amount and type of conservatism applied for screening will be dependent on the level of understanding of the pollutant in terrestrial food chains, the quality of available data, and the history of model validation. The degree of conservatism will also be related to the actual or perceived health risk associated with potentially exceeding an established limit. Thus, the greater the level of knowledge about the pollutant in a specific terrestrial system and the lower the health risk associated with exceeding a limit, the lesser is the amount of conservatism required for screening (Fig. 6).

Model validation studies will be useful to determine the extent of the conservative bias in model predictions and to identify possible situations where potential underestimation can be avoided. Once model validation has determined the degree of overestimation associated with conservative bias, efforts can be directed at increasing the accuracy of model predictions. Model validation of screening models will also serve to place an upper

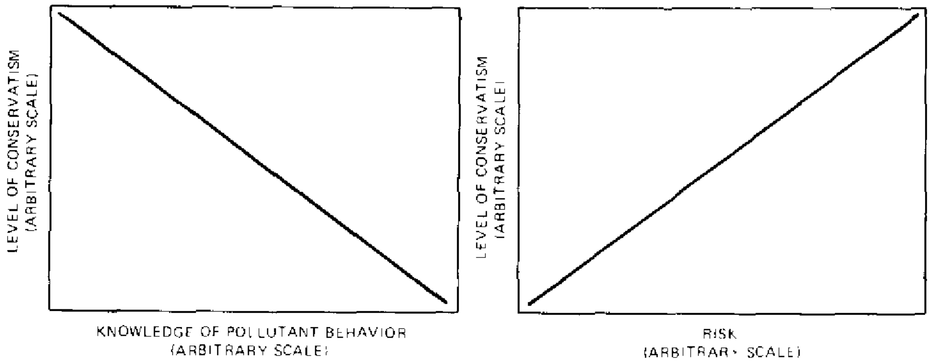


FIG.6. Relationship between the degree of conservatism required for screening calculations, the level of knowledge of the behavior of the pollutant in terrestrial food chains, and the risk of the pollutant to human health.

bound on predictive uncertainty. Thus, if validation indicates that a screening model does not overpredict by more than a certain quantity, predicted food-chain concentrations exceeding limits by more than this amount would be immediately implicated as unacceptable. Without validation, further research and evaluation would be required prior to the final designation of unacceptable food-chain concentrations.

Sometimes process-level models are proposed for screening calculations. These situations most frequently occur when the actual behavior of the pollutant in terrestrial food chains is not well understood and when relevant parameter values and model validation results are lacking. The presence of a complex mathematical structure in such a situation presents an aura of model realism that may appear to produce more defensible results than models having a less complex structure [40]. However, unless available data and validation results can justify the need for increased complexity for screening purposes, the assumption of increased defensibility with increased structural complexity may be unwarranted.

Screening models associated with a sufficient degree of conservatism will be most useful for calculations performed for pollutants, pathways and locations that do not have an extensively developed base of data. The use of such models to evaluate the significance of pollutants released from industrial and agricultural operations in developing countries will enable an efficient identification of situations requiring more detailed analyses and expenditures of resources to improve the accuracy of preliminary predictions. Here in the United States of America, for example, screening models are essential for evaluating potentially hazardous chemicals under the Toxic Substances Control Act of 1976 [41] because data on the toxicity and behavior of each of the thousands of specific chemical species that must be considered are simply not available.

For the assessment of radionuclides, a quasi-equilibrium model using generic default transfer factors is being published by the International Atomic Energy Agency for use as a tool in screening for radioisotopes and pathways resulting in doses to critical groups of the population that approach pre-selected dose limits within one order of magnitude [20]. The development of this model was specifically intended for use by developing countries that may not have access to relevant data for use in assessment calculations. For screening of other pollutants of potential concern to human health, Dacre et al. [37] propose the application of models utilizing a series of equilibrium concentration factors.

Although all types of models used for assessment and research purposes can be adapted for screening calculations, it is our experience that models are most successful for screening when they are easily implemented by individuals not directly related to their development. Usually, calculational procedures are not extensive and the data base includes generic default parameter values for use in the absence of site-specific information. A crude calculation will be entirely acceptable for screening purposes if resultant concentrations are well below established limits.

3.2. The Use of Models for Optimization of Releases

Another application of assessment models is to optimize pollutant release rates to levels considered as low as reasonably achievable. Optimization therefore involves a comparison between the benefit gained in terms of reduced health risk and the societal cost required to achieve this reduced risk [42]. Terrestrial food-chain models developed for optimization calculations would be used to justify the costs encountered to reduce pollutant releases and to compare health risks related to pollutants discharged from alternative types of industrial or agricultural practices.

Screening models have limited applicability to optimization calculations because conservative overestimates of pollutant concentrations can grossly mislead decisions, resulting in potentially unwarranted financial costs or unnecessary dismissal of potentially viable technologies. For optimization, unbiased predictions are preferable to conservative estimates. The achievement of unbiased model predictions, however, requires knowledge of predictive uncertainties. In the absence of such knowledge, models developed for optimization often employ "best estimate" or average values for parameters derived using the

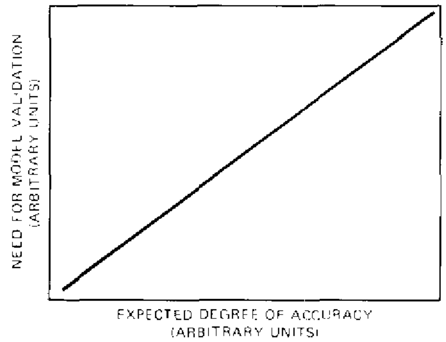


FIG. 7. Relationship between model validation and the degree of accuracy expected of models used for optimization calculations.

judgment of the model developer or the model user. These judgment estimates also may result in misleading conclusions without some indication about the possible uncertainties, because best-estimate values may incorrectly predict actual conditions prevailing in a given assessment situation.

Uncertainties associated with model predictions are best determined through model validation studies [43,44,45]. Unfortunately, only limited validation for a few pathways and radioisotopes has been performed for radiological assessment models [36,46,47], with no known validation having been performed for the assessment of non-nuclear pollutants. Nevertheless, the extent of model validation required before models can be reliably used will be directly related to the degree of accuracy expected of model predictions (Fig. 7). Therefore, present optimization evaluations are limited by uncertainties associated with the judgment used to derive "best estimate" or "expected value" predictions for specific pollutants and food-chain pathways.

Sometimes, in the absence of model validation, process-level models are proposed for optimization calculations because of the apparent realism in model structure. Again, as with the application of such models for screening, the usefulness of process-level models is limited because of their complexity, the magnitude of data requirements, and sensitivity to error propagation. Furthermore, as discussed previously, validation of these models is difficult or impractical. Given the current state of model validation, predictions made with process-level models serve to provide a formalized viewpoint about the possible behavior of pollutants in terrestrial systems that at least can be compared with the results of other types of model formulations.

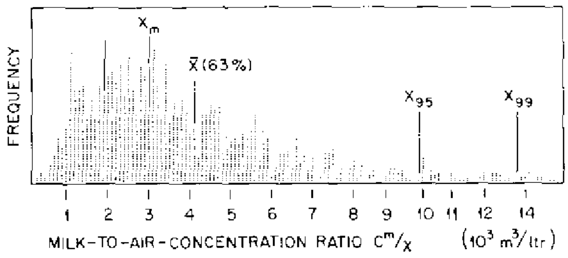


FIG.8. The milk-to-air concentration ratio C^m/χ for $^{131}I_2$ simulated using Monte Carlo computations with estimates of the median (X_m), mean ($\bar{X}$), 95th (X_{95}), and 99th (X_{99}) percentiles included within the distribution of predicted values (Schwarz and Hoffman [52]).

4. DIRECTIONS FOR IMPROVEMENT

4.1. Quantification and Reduction of Predictive Uncertainties

Emphasis on future model improvements for the assessment of pollutant transport in food chains is being placed on the quantification and reduction of predictive uncertainty [48,49,35]. The use of screening models will facilitate identification of pollutants and pathways requiring models with improved predictive accuracy. It is our contention that only through more aggressive efforts to validate terrestrial food-chain models will desired improvements in predictive accuracy be realized.

Model validation represents actual field testing under the range of conditions that a model may be intended for use [43,44]. Model validation will therefore be expensive and time-consuming. The trade-off between an aggressive validation program and no validation is the difference between the cost required to identify model uncertainties and the risks involved with predictions whose uncertainties are either unknown or grossly over-conservative.

In the absence of results from model validation, some investigators have attempted to analyze predictive uncertainties using analytical and numerical techniques for simulating the propagation of error in terrestrial food-chain model predictions due to the variable nature of model parameters [50 – 54]. This approach enables a model to produce a range of predictions with a distinct frequency distribution rather than just a single value. Figure 8 illustrates an example of such an error propagation analysis for the transport of ^{131}I over the pasture-cow-milk pathway. Error propagation techniques by themselves, however, can only provide an estimate of predictive imprecision [44].

The departure of predicted values from real-world concentrations of pollutants in terrestrial foods remains unknown without field testing, although it is conjectured that real-world values should be encompassed by the range of predictions produced through error propagation [6].

For quantifying predictive uncertainty, model validation offers the possibility to identify both bias and imprecision when combined with error propagation techniques. Reduction in predictive uncertainty can be accomplished through model validation by calibrating model predictions with observations. Calibration of predictions with observations is most justified when there is a high correlation between predictions and observations. A poor correlation between predictions and observations indicates the need for improvement of model structure or in the determination of key parameters. However, even when poor correlations predominate, model predictions may still undergo calibration to assure a sufficient margin of conservatism for screening calculations. Nevertheless, such models should only be applied within the realm of environmental and ecological conditions tested through validation studies. Extrapolation of model predictions beyond this realm is extremely dangerous when predictions exhibit poor correlations with observations, especially when the bottom line is a certain degree of health risk.

4.2. Improve Site-Specific Assessment Capability

Further reduction in predictive uncertainty can be expected with improved capability for site-specific assessments. Current terrestrial food-chain models contain parameters whose values may be readily determined for a specific site, such as standing vegetation biomass, the bulk density of surface soil, and some aspects of cattle management practices [10,16,20]. But in general, pollutant-specific transfer factors or rate constants cannot be determined for a specific site without extensive monitoring or experimental investigation. At present these transfer factors or rate constants have been derived from the global literature and aggregated for major food categories such as vegetables, milk and meat. The variation of these parameters with plant and animal species or breeds, agricultural practices, soil characteristics and climatic regimes has not been determined for most types of pollutants.

A considerable amount of site-specific experimentation can be avoided if it is possible to correlate pollutant-specific parameters with easily measurable environmental variables such as soil pH, temperature, biomass, etc. For radionuclide assessments, correlations with a few environmental variables have been developed for isotopes of strontium and cesium [55,36]. However, development of data to permit derivation of such relationships requires a large investment in specifically designed laboratory and field research.

Improving the site-specific capability of assessment models will also require improved knowledge of human dietary habits for specific locations. Currently, health risks are typically assessed by combining predicted food concentrations with rates of food consumption by humans corresponding to reference diets assumed to represent average or maximally exposed individuals [56,10]. The relevance of these assumed diets to specific situations may contribute to assessment uncertainty; however, determination of regional-specific or locational-specific dietary habits may be difficult because of rapidly changing trends in future market practices [57].

4.3. Consideration of Model Simplification

An additional consideration for future improvement of terrestrial food-chain assessment models for assessment purposes is the achievement of an optimum level of model simplification. Usually, the "best" model for a given assessment will be the model that is easiest to use and which produces results within an acceptable degree of accuracy which address the questions asked of the model by the model-user [45]. The degree of "acceptable accuracy" will be determined by the results of validation studies and the risks associated with over- or underestimating the impact to human health.

Simplification should be most achievable for models used for screening calculations because of inherent conservatism. However, if an extensive series of validation tests has been performed, it may also be possible to simplify models developed for optimization calculations through calibration of model predictions to observations. For most situations within the realm of our experience, many complex model formulations appear to be more suited as research rather than assessment tools because of deficiencies in available data and difficulties involved with achieving further improvements in the data base. Nevertheless, these models may still have utility for guiding experimental studies and obtaining valuable insights on pollutant behavior and potential pollutant bioaccumulation.

Additional structural complexity inevitably will be required to increase the flexibility of terrestrial food-chain models for site-specific assessments. For the sake of practicality, however, it may be advisable to maintain structural simplicity with the dependencies of parameters on site-specific and pollutant-specific characteristics being considered external to the model that is used for assessments. Thus, simple transfer factors or rate constants may actually be derived from tables of data and more complex model formulations that are kept separate from the basic assessment model. Site-specific models consisting of few parameters for important pollutants and pathways should be more amenable than more complex formulations to validation studies conducted under conditions of high temporal and spatial variability because of reduced demands on simulataneous parameter measurement [36]. For assessment purposes, an optimum level of structural complexity can be envisioned when all parameters are readily measurable and the possibility of predictive error due to unforeseen correlations among the parameters is small. The optimum level of simplification, however, can only be determined on the basis of results gained through model validation and the ease with which the model can be implemented.

5. CONCLUSIONS

The primary purpose for the development and application of terrestrial food-chain models in the assessment of pollutant releases to the environment is the prediction of human health impact. Most of the examples we have used in this paper have been taken from models applied for the assessment of radionuclides because formal assessment methodologies for other types of pollutants are still at an early stage of development. Models used in assessments have been designed to produce quasi-equilibrium or dynamic predictions; however, both model types suffer from limitations in parameter value determination and validation. Because the determination of parameter values is highly dependent on empirical derivation, extrapolation of model predictions to locations, time periods and chemical species of pollutants for which data are not available must be exercised with due caution. Furthermore, model predictions are inherently limited by the information available for determination of model structure and parameter values. A model cannot be expected to provide better information than was put into it. The limitations in parameter quantification and model validation will therefore probably restrict most uses of terrestrial food-chain models in assessments to a type of screening calculation. Potential for misprediction exists each time a model is applied to a new situation, regardless of the results of past validation experiments. Therefore, there will always be a need to implement environmental monitoring programs to ensure that potential model misprediction does not result in unacceptable consequences once releases of pollutants to terrestrial food chains actually occur.

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DISCUSSION

J. SCHWIBACH (*Chairman*): I fully agree with you about the importance of environmental measurements. One problem here is that in many cases even good results from environmental surveillance programmes cannot be used to improve model parameters because such measurements are not aimed primarily at checking the individual steps of the calculation but are a final check on the calculated environmental effects. However, if the end result is good perhaps we should not worry too much about parameter uncertainties.

F. GIRARDI: I also feel that data uncertainty is one of the major problems in modelling and that experimental validation of model results should be carried out when possible. However, this is rarely possible in long-term consequence analyses. We have found it useful in such cases to treat the data uncertainty in the model in the following way: probability distribution functions are assigned to each variable of interest and the propagation of the uncertainty is studied either by analytical models or by Monte Carlo simulation. Do you have any experience of such methods and what do you think of their use when validation of the results of models proves difficult?

S.V. KAYE: We have in fact performed Monte Carlo simulations to try and determine the uncertainties associated with the predictions, as can be seen from Fig.8 of the paper. Our approach was to examine and evaluate the world's literature for each parameter to ensure that our measurements were taken with the proper instruments, that they represent equilibrium values and so on. We then took the values which were acceptable and used them to generate a distribution. In Fig.8, the milk-to-air concentration ratio for ^{131}I is simulated by Monte Carlo computations and the data in the distribution generated using the methodology described above. One of the main problems which arises when attempting to protect health and calculate risks is deciding on the criterion to be used. The choice of limiting value depends on whether the aim is to protect 99% of the people or simply the average person. The difference between the two values is a factor of 4, which indicates the difficulty of the questions which we must ask ourselves. The answers are just as difficult to find because the modeller often has to decide which values to use in his predictions.

The other type of evaluation performed by us is model validation for which field measurements have to be taken. The main limitation here is knowing the source term, i.e. the release to the environment which was the origin of the pollutant being measured. It is often hard to link the concentration in the receptor compartment with the source term.

Y. NISHIWAKI: I should also like to add a comment. Cost-benefit analysis or risk-benefit analysis may be useful in designing industrial facilities or drawing up certain regulatory measures, but they might well give rise to difficulties if used in the context of public acceptance in Japan. When different groups of people perceive risk and benefit in different ways, the result may well be social conflict, which, if it escalates, increases the overall risk to society. Environmental protection in Japan, therefore, is based largely on the principle of keeping adverse effects within the range of fluctuation of similar effects which occur naturally.

STUDIES ON THE EFFECTS OF ATMOSPHERIC CONTAMINATION DUE TO FOSSIL-FUEL COMBUSTION IN JAPAN

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Abstract

STUDIES ON THE EFFECTS OF ATMOSPHERIC CONTAMINATION DUE TO FOSSIL-FUEL COMBUSTION IN JAPAN.

Epidemiological studies have been conducted since 1961 to investigate health effects of sulphur dioxide in industrial areas of Japan where fossil-fuel power stations are located. The dose-response relationship between prevalence rates of chronic bronchitis and sulphur dioxide was established. Various efforts have been made to reduce the concentrations of sulphur dioxide in the atmosphere. Consequently, the annual value of sulphur dioxide concentrations estimated by the national network of air pollutant measurements decreased from the peak value of 0.059 ppm in 1967 to 0.017 ppm in 1978. However, the atmospheric concentration of nitrogen dioxide estimated by the national network indicated an annual value of 0.022 ppm in 1968, but the annual value in 1978 was slightly increased to 0.027 ppm. The average concentration tended to increase gradually. It was therefore considered important to study the health effects of nitrogen dioxide. In six different areas in Japan with varying atmospheric concentrations of nitrogen dioxide, an extensive epidemiological survey was conducted with 12 717 school-children 6 to 12 years old during the period 1979 to 1981. The prevalence rate of asthma was estimated to be 4.7% for males and 2.1% for females in the high NO_2 concentration

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area, and 1.9% for males and 0.9% for females in the low NO_2 concentration area. For asthma-like symptoms, 12.2% for males and 11.9% for females was observed at the high NO_2 concentration area, and 7.1% for males and 5.9% for females in the low NO_2 concentration area. The basic principle of environmental protection is to keep the adverse effects of pollutants to an insignificant level or within the fluctuation range of the naturally occurring similar effects.

The natural radioactivity from fossil-fuel power plants as well as risk benefit comparisons are also discussed. In decision-making on environmental protection and safety, it should be carefully considered whether a reduction of one type of risk might increase another type of risk or whether a heavy burden on industries to reduce one type of risk might create another type of risk in society, based on a basic principle of 'risk versus risk'. Not only the risk-reduction industries but also the construction and operation of the risk-reduction system may not be completely riskless. The well-balanced overall safety of society may be better achieved by promoting harmony within society rather than by increasing conflict between the risk groups and the benefit groups.

1. INTRODUCTION [1–8]

In the 17th Century, some cases of damage to vegetation by poisonous effluents were first reported in Japan in connection with copper, silver and gold mines. The adverse effects of smoke poison on human health were first reported in about the 18th Century. At the time,these problems were considered to be local ones associated with small-scale primitive local industries. Since large-scale industrialization began about a century ago, the combustion of fossil fuels in various industrial facilities has given rise to a number of serious cases of atmospheric contamination in industrial urban areas which affected a large number of people.

The air pollutant from industrial processing and fossil-fuel power plants may include fly-ash, certain hydrocarbons in the particulate group,and sulphur oxides, nitrogen oxides, carbon oxides, ozone, and various other organic compounds, trace metals and naturally occurring radioactivity. It was also noticed in earlier times that the interactions among pollutants were important and depended on their chemical nature and quantity as well as such physical features as temperature, wind strength and direction, mixing depth, topography, distance from sources, concentration, etc.

After the Second World War, industrialization, urbanization and motorization developed very rapidly in Japan, and the socio-economic situation has greatly changed in recent years. Japan is densely populated and about 60% of the total population is concentrated in the urban areas. Various industrial facilities are also established in urban areas or in their surroundings. The atmospheric contamination in these industrialized areas therefore rapidly deteriorated since the latter half of the 1960s, and the environmental problem has become a serious national social problem. Measurement of atmospheric contamination and epidemiological surveys have been conducted since the mid-1960s in different parts of Japan, for which the standardized questionnaires of the British Medical Research Council were

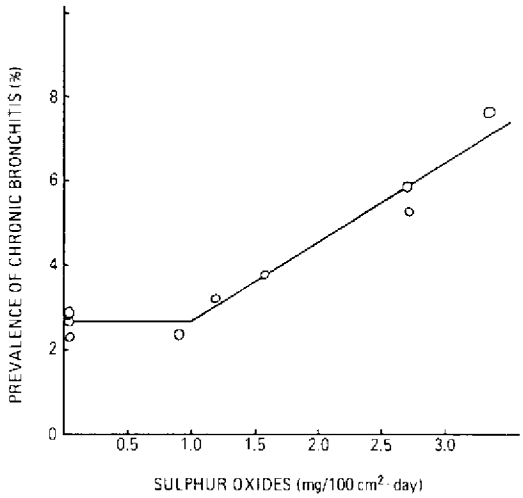

FIG.1. Correlations between the prevalence of chronic bronchitis and sulphur oxide precipitation (measured by the PbO_2 method). The overall air pollution may be represented by the major pollutant sulphur oxides, and 1 mg sulphur oxide per 100 cm^2·day may correspond to about 0.03 to 0.035 ppm in this case.

mainly used. As a result of these studies, a clear dose-response relationship was observed between the prevalence rate of chronic bronchitis and the concentration of atmospheric contamination with sulphur oxides as reference. An example of such a relationship is shown in Fig.1 which was quoted from the report by Tsunetoshi et al. [6] (and see Ref. [8]).

As a measure to prevent environmental contamination, the Basic Law for Environmental Pollution Control was passed in August 1967. The Air Pollution Control Law was established in August 1968 for the prevention of atmospheric contamination. The Environmental Quality Standard for Sulphur Oxides was established in February 1969, followed by standards for carbon monoxide, nitrogen dioxide, suspended particulate matter and oxidants. As a result of these regulatory measures the concentration of sulphur oxides, which were the major contaminants in the 1960s, was greatly reduced in the 1970s. However, the atmospheric contamination level of nitrogen dioxide, suspended particulate matter and other contaminants did not seem to decrease significantly despite various regulatory measures.

Under these conditions, it is extremely important to collect the results of epidemiological surveys, thus providing basic data which may be useful in establishing preventive measures against various atmospheric contaminants.

TABLE I. AMBIENT AIR QUALITY STANDARDS, JAPAN (taken from Ref. [1])

Substance	Sulphur dioxide	Carbon monoxide	Suspended particulate[1] matter	Nitrogen dioxide[3]	Photochemical[2] oxidants
Environmental conditions	Daily average of hourly values shall not exceed 0.04 ppm, and hourly values shall not exceed 0.1 ppm.	Daily average of hourly values shall not exceed 10 ppm, and average of hourly values in eight consecutive hours shall not exceed 20 ppm.	Daily average of hourly values shall not exceed 0.10 mg/m³, and hourly values shall not exceed 0.20 mg/m³.	Daily average of hourly values shall be within the range between 0.04 ppm and 0.06 ppm or below.	Hourly values shall not exceed 0.06 ppm.
Measuring methods	Conductometric method	Nondispersive infrared analyser method	Weight concentration measuring methods based on filtration collection, or light scattering method yielding values having a linear relation with the values of the above method	Colorimetry employing Saltzman reagent (with Saltzman's coefficient being 0.84)	Absorptiometry using neutral potassium iodide solution, or coulometry

Notes: 1. Suspended particulate matter shall mean airborne particles of 10 microns or less in diameter.
2. Photochemical oxidants are oxidizing substances such as ozone and paroxiacetyl nitrate produced by photochemical reactions (only those capable of isolating iodine from neutral potassium iodide, excluding nitrogen dioxide).
3. a) In an area where the daily average of hourly values exceeds 0.06 ppm, efforts should be made to achieve the level of 0.06 ppm by 1985.
b) In an area where the daily average of hourly values is within the range between 0.04 ppm and 0.06 ppm, efforts should be made so that the ambient concentration be maintained around the present level within the range or not significantly exceed the present level.
c) Not only emission control measures against individual sources but also other various countermeasures should be implemented in an integrated, effective and appropriate manner in order to maintain or achieve the ambient air quality standard.

2. AIR QUALITY STANDARDS IN JAPAN [1–5]

The Air Quality Standards of Japan are based on the Basic Law for Environmental Pollution Control. They are the standards which are considered desirable for the health of the people and are administrative targets, i.e. they are to be revised and updated from time to time, on the basis of scientific judgement if necessary. The standards are established for SO_2, CO, suspended particulate matter (SPM), NO_2 and photochemical oxidants, as shown in Table I.

The Air Quality Standards for SO_2, established in February 1969, are as follows: daily average of hourly values shall not exceed 0.05 ppm and hourly values shall not exceed 0.2 ppm. However, scientific findings on health effects of SO_x have gradually accumulated since then and it was considered that the SO_x concentration should be further reduced by intensifying emission control of sulphur oxides. In May 1973 the standards were revised as follows: daily average of hourly values shall not exceed 0.04 ppm and hourly values shall not exceed 0.1 ppm.

The Air Quality Standards for CO were established in February 1970 because of the increase in CO emission due to motorization. They stipulated that daily average of hourly values shall not exceed 10 ppm, and average hourly values in eight consecutive hours shall not exceed 20 ppm (Table I).

The Air Quality Standards for SPM were established in May 1973 as follows: with respect to the weight of an airborne particle of 10 μm or less in diameter, daily average of hourly values shall not exceed 0.1 $mg \cdot m^{-3}$ and hourly values shall not exceed 0.2 $mg \cdot m^{-3}$. The chemical composition of the particles is not considered in the standards.

The Air Quality Standards for NO_2 were established in May 1973 as follows: daily average of hourly values shall not exceed 0.02 ppm. At that time, there were very few scientific findings on health effects of NO_x. For the prevention of hazards, the above strict standards were therefore established, taking into account a large safety factor based on scientific judgement. After some time, a sufficient number of findings on health effects of NO_2 had been obtained, and in July 1978 the standards were revised as follows: daily average of hourly values should range between 0.04 ppm and 0.06 ppm or below.

There is a considerable regional difference in the NO_2 contamination level. It was therefore considered administratively appropriate for each region to establish its own target, according to the respective NO_2 contamination level of each region. To allow for this difference in contamination level between regions, the concept of a range was introduced in the NO_2 concentration limit. Because the effects of NO_2 cannot be specified by a point but may be better expressed as a range, it was considered appropriate to express the concentration limit as a range. Where the background level is high, the higher limit may be used as an administrative target for environmental protection, and where the background level is low, the lower limit may be used.

The Air Quality Standards for photochemical oxidants were established in May 1973 as follows: hourly values shall not exceed 0.06 ppm.

In addition, for the prevention of photochemical air pollution, the following guidelines for hydrocarbons were issued by the Government in August 1976: the average value of concentration of non-methane hydrocarbons for three hours between 6.00 a.m. and 9.00 a.m. should be in the range from 0.20 ppm C to 0.31 ppm C. The three hours from 6.00 a.m. to 9.00 a.m. may correspond to the time of maximum hourly value 0.06 ppm of photochemical oxidants.

3. RECENT STATE OF AIR POLLUTION IN JAPAN AND COUNTERMEASURES [1, 2, 8]

3.1. Sulphur oxides

Measurement of SO_2 concentration began in 1965 at general air pollution monitoring stations. Measurement time of 6000 hours or more per year is called *effective measurement value*. In 1979, effective measurement values were obtained at 1532 stations in 590 cities. The changes in annual average, maximum and minimum concentrations of SO_2 were measured at 15 general pollution monitoring stations, located at heavily contaminated areas, which have been continuously measuring SO_2 concentration since 1965.

The annual average concentration of SO_2 reached the highest value of 0.059 ppm in 1967 and thereafter gradually decreased to a low value of 0.016 ppm in 1979. The difference in concentrations measured at different stations also decreased.

Reduction of SO_2 air concentration may be ascribed to the effective control of various SO_x emissions. The Emission Standards specified for one facility have been in effect since 1968, and the Regional Emission Standards for multiple factories located in heavily polluted areas became effective in 1976. The enforcement of these emission standards led to various measures being taken to reduce SO_2 emissions, and it was estimated in 1978 that more than 75% of the total sulphur would have been removed prior to emission.

3.2. Nitrogen oxides

Nitrogen dioxide concentrations have been measured by general air pollution monitoring stations. Effective measurement values of NO_2 were obtained at 1080 stations in 500 cities in 1979.

Emissions from automobile exhausts were measured by automobile pollution monitoring stations. The effective measurement values of NO_2 were obtained at 205 stations in 112 cities in 1978. The changes in annual average, maximum and minimum concentrations of NO_2 were measured at 15 monitoring stations in continuous operation since 1970. The changes in annual average concentrations of NO_2 were also measured at 26 automobile exhaust monitoring stations in continuous operation since 1971. No remarkable changes in average concentrations were observed in these measurements but there was a tendency to gradual increase.

In the general air pollution monitoring stations, higher concentrations exceeding the upper limit of NO_2 air quality standards (0.06 ppm) were observed at 46 stations (4.6%) in 1979. The stations were located in the Tokyo Bay area and in Osaka prefecture. Of the automobile pollution monitoring stations, higher concentrations of NO_2 exceeding 0.06 ppm were observed at 77 stations (40.5%) in 1978.

Nitrogen monoxide was also measured on a scale almost comparable with NO_2. However, the changes in annual average concentrations were observed to tend to decrease at the general air pollution monitoring stations as well as at the automobile pollution monitoring stations. The sources of nitrogen oxides are various: stationary sources such as fossil-fuel power plants, factories, etc.; mobile sources such as automobiles, aircraft, ships, etc.; domestic sources such as cooking, heating, etc.

Emission standards for NO_x for soot- and smoke-generating facilities were enforced in 1973, and the restrictions on NO_x emitted from automobiles also took effect at the same time. NO_x control is gradually being intensified. It is now considered that regional pollutant standards should be enforced in the near future, in six regions with high concentrations of NO_2: Tokyo, Osaka, Yokohama, Nagoya, Kita-Kyushu and Kobe.

3.3. Carbon monoxide

The major source of carbon monoxide is the exhaust gas of automobiles. The automobile pollution monitoring stations measured CO at 297 stations in 133 cities in 1978. The general air pollution monitoring stations obtained effective measurement values of CO at 200 stations in 167 cities in 1979. CO concentrations are also gradually decreasing. The control of CO concentrations in automobile exhaust gas began in 1966 and was gradually intensified. At present, the CO concentration is reduced to less than 1/10 what it was before the introduction of exhaust control.

3.4. Photochemical oxidants

The effective measurement values of photochemical oxidants (OX) concentrations, excluding NO_2, have been obtained at 899 stations in 460 cities. Oxidants are produced by photochemical reaction and may depend on meteorological conditions to a large extent, but there has been a decrease in recent years.

The main product of photochemical reaction is ozone, but there are others such as peroxyacetyl nitrate (PAN) and aldehydes. Although oxidant concentrations have decreased in recent years, there are some uncertainties about the effects of photochemical reaction products such as PAN and aldehydes, and it was thought necessary to conduct further studies and surveys on their effects.

Control of photochemical oxidants means the control of NO_x and hydrocarbons which may be considered major causes of photochemical oxidants. The control measures of NO_x are described above. The control of hydrocarbons emitted with the exhaust from automobiles began in 1970. The hydrocarbons from stationary sources are not controlled by Government regulation but by the local community through emission controls associated with measures for the prevention of offensive odour.

Hydrocarbons in the atmosphere have been measured (in connection with photochemical air pollution) by the general air pollution monitoring stations and the automobile exhaust monitoring stations. The concentrations of non-methane hydrocarbons were measured at 210 stations, but most of the annual average concentrations measured at these stations exceeded 0.31 ppm C, which is the upper limit in the guidelines on non-methane hydrocarbons given by the Central Council for Environmental Pollution Control in August 1976.

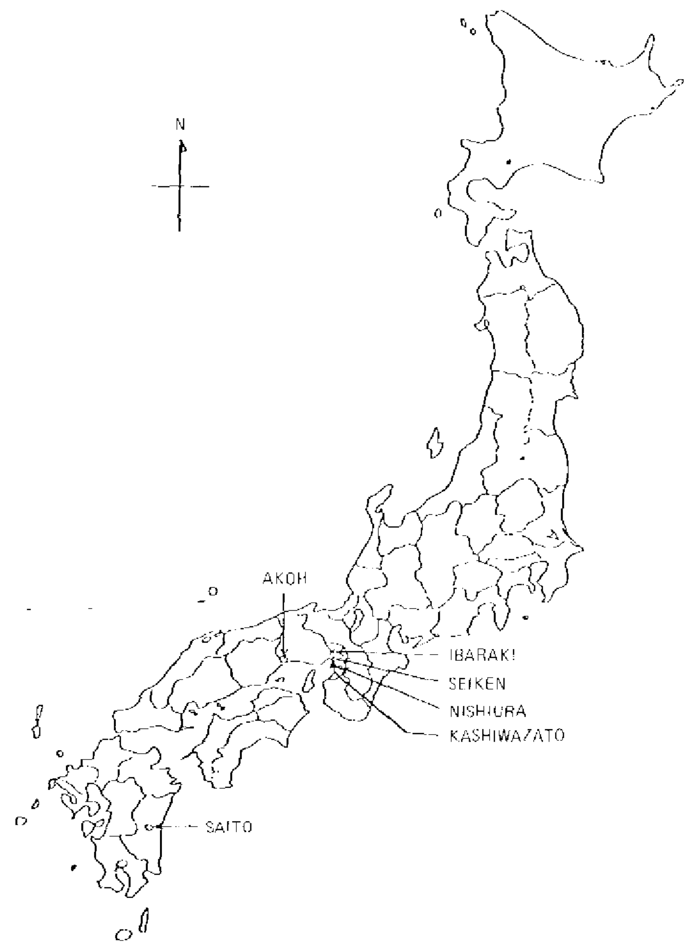

FIG. 2. Location of the six survey areas.

3.5. Suspended particulate matter (SPM) and other substances

Effective measurement values of SPM were obtained at 226 stations in 105 cities in 1979. Of these stations, 180 (79.6%) were observed to have higher concentrations than 0.1 $mg \cdot m^{-3}$ of SPM, which is the value given in the Environmental Air Quality Standards. The amount of dust-fall is gradually decreasing.

Chemical components in airborne particles have recently been measured at major national monitoring stations. They include sulphate and nitrate radicals, heavy metals such as vanadium, benzo(a)pyrene, aluminium, barium, etc. Control of emission of particulate matter from all industrial facilities has been conducted since 1968. Control of black smoke from diesels began in 1972. Control of chemical components included in SPM and the investigation of their health effects are, however, still the subjects of discussion.

4. STUDY ON HEALTH EFFECTS OF AIR POLLUTION IN SIX AREAS [5–13]

This section introduces the results of an epidemiological investigation from June 1979 to March 1981 on the health effects of air pollution on schoolchildren.

4.1. Materials

The epidemiological investigation was based on a total of 12 717 schoolchildren in six areas with varying degrees of atmospheric pollution in different parts of Japan.

Seiken and Kashiwazato are industrial areas of Osaka City where large and small factories are concentrated. Ibaraki is a densely populated commercial area in the north-eastern part of Osaka prefecture. Nishiura is a residential district in the rural area of the south-eastern part of Osaka, where the population is rapidly increasing. Akoh is a local city with a population slightly over 50 000; it is in the southern part of Hyogo prefecture, facing the Inland Sea of Seto, and is a commercial and residential area. Saito is a small city with a population of slightly over 30 000. It is in an agricultural area in the central part of Miyazaki prefecture. The relative positions of the six survey areas are shown in Fig.2.

Figure 3 shows the trends in annual average concentrations of SO_2 and Fig.4 shows the trends in NO_2 concentrations for the six areas. A considerable difference in the degree of air pollution can be seen among these survey areas. The SO_2 concentrations in Seiken and Kashiwazato were quite high until several years ago and the NO_2 concentrations are also high there. The NO_2 concentration in Seiken is always so high that the area is considered one of the five worst in Japan.

As a result of these observations, Seiken and Kashiwazato were considered to be heavily polluted areas. Akoh and Saito, with a low level of SO_2 and NO_2, were considered low-pollution areas, and Ibaraki and Nishiura were considered moderately polluted areas.

4.2. Method

Questionnaires were based on those of the Environment Agency of the Japanese Government, which were based on the standard questionnaires of the Epidemiology Standardization Project of Ferris et al. [9, 10] in the USA, and adapted for the purposes of this study.

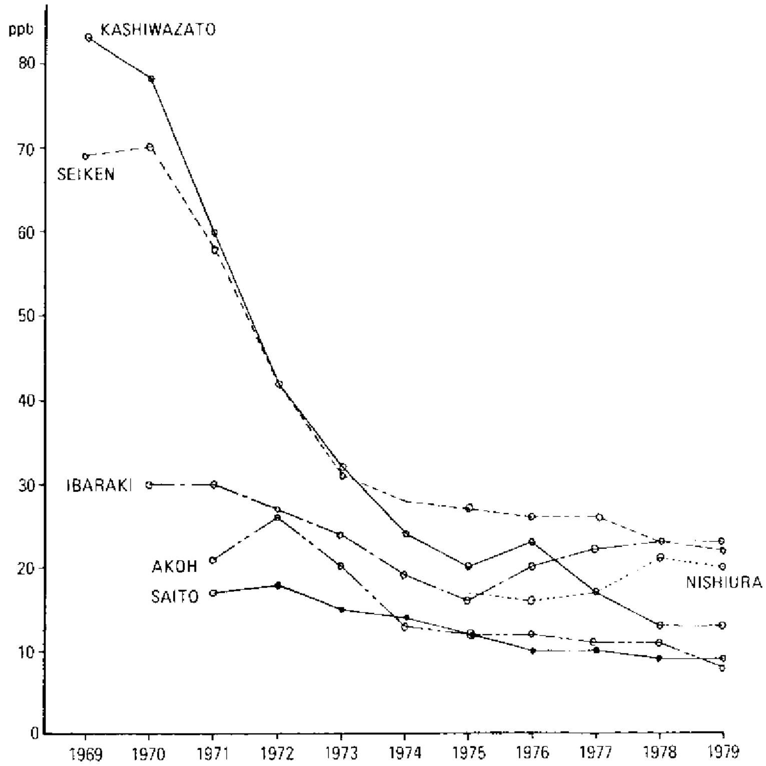

FIG. 3. Trends in annual average concentrations of SO_2 in the six survey areas (ppb = parts per billion, where 1 billion = 10^9).

The questionnaires were distributed via the schoolchildren to their parents for completion and were returned within one week. Omissions and re-entries were checked as far as possible by re-investigation. By a combination of items in the questionnaire the health conditions were classified into three categories: *healthy*, *asthma* and *wheezing*. Those in the *healthy* group were without any pathological findings in heart and/or lungs and without any medical history thereof. The *asthma* group was based on the definition of the American Thoracic Society and was confined to children who had one attack or more during the previous two years. The flow volume curve recorder STM-81-N (Chest Corp., Tokyo, Japan) was used for respiratory function test [11]. This instrument can record simultaneously maximum expiratory flow volume (MEFV) curve and spirogram. The test was repeated until a curve with good reproducibility was obtained. From this curve forced vital capacity (FVC) forced expiratory volume (FEV) in 0.75 seconds, peak expiratory flow-rate (PFR), maximum expiratory

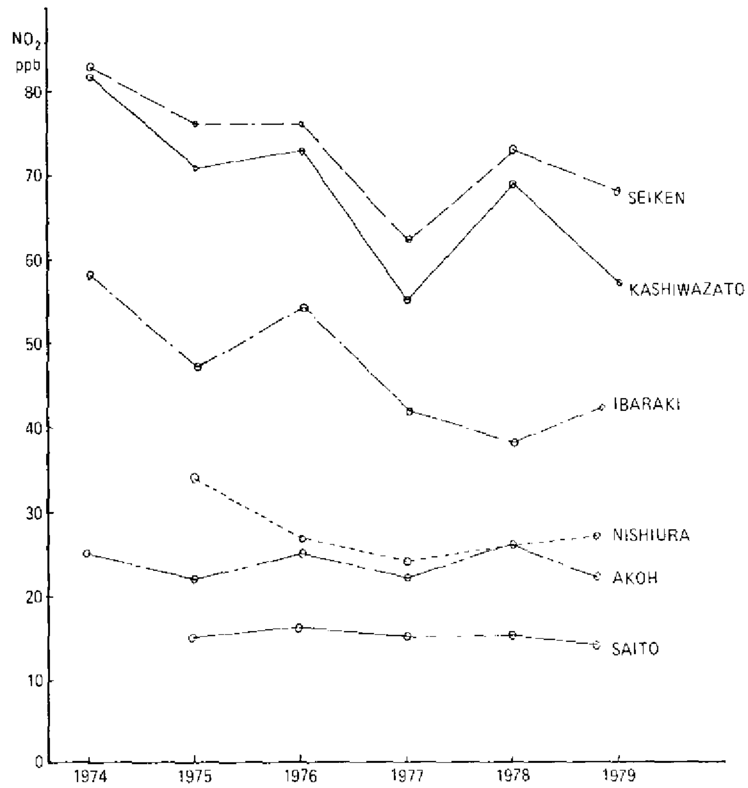

FIG.4. Trends in concentration of NO_2 in the six survey areas; daily average value for 98% of the year (ppb = parts per billion, where 1 billion = 10^9).

flow rate at 50% and 75% FVC ($\dot{V}_{50}$, $\dot{V}_{75}$) were read off. The read-off values were converted to the values at body temperature, atmospheric pressure, vapour-saturated (BTPS) condition. An immunological serum IgE test was conducted, with ultrasensitive sandwich-type enzyme immunoassay, by Ishikawa et al. [12].

The children who received IgE tests were those with respiratory symptoms and one who volunteered for the test. Prior consent of the parents had been obtained.

For the survey of individual exposure to NO_2, a film-badge type NO_2 personal monitor devised by Yanagisawa [13] was used. The monitor is based on the principle that a filter containing 20% triethanolamine absorbs NO_2 which diffuses through layers of hydrophobic fibre filters. The badges were worn by the children or placed in the playground for one week and recovered for measurement of absorbed NO_2.

TABLE II. PREVALENCE RATE OF ASTHMA AND WHEEZING (BY SEX AND REGION)

Region	Male			Female		
	No. of subjects	Asthma	Wheezing	No. of subjects	Asthma	Wheezing
Seiken	491	23(4.7)[a]	59(12.0)	438	9(2.1)	52(11.9)
Kashiwazato	426	19(4.5)	49(11.5)	353	9(2.5)	44(12.5)
Ibaraki	710	15(2.1)	67(9.4)	702	11(1.6)	53(7.5)
Nishiura	479	14(2.9)	33(6.9)	459	5(1.1)	24(5.2)
Akoh	2619	69(2.6)	223(8.5)	2506	38(1.5)	150(6.0)
Saito	1810	35(1.9)	129(7.1)	1645	14(0.9)	96(5.8)
Total	6535	175(2.68)	560(8.57)	6103	86(1.41)	419(6.87)

[a] The figures in parentheses indicate percentage.

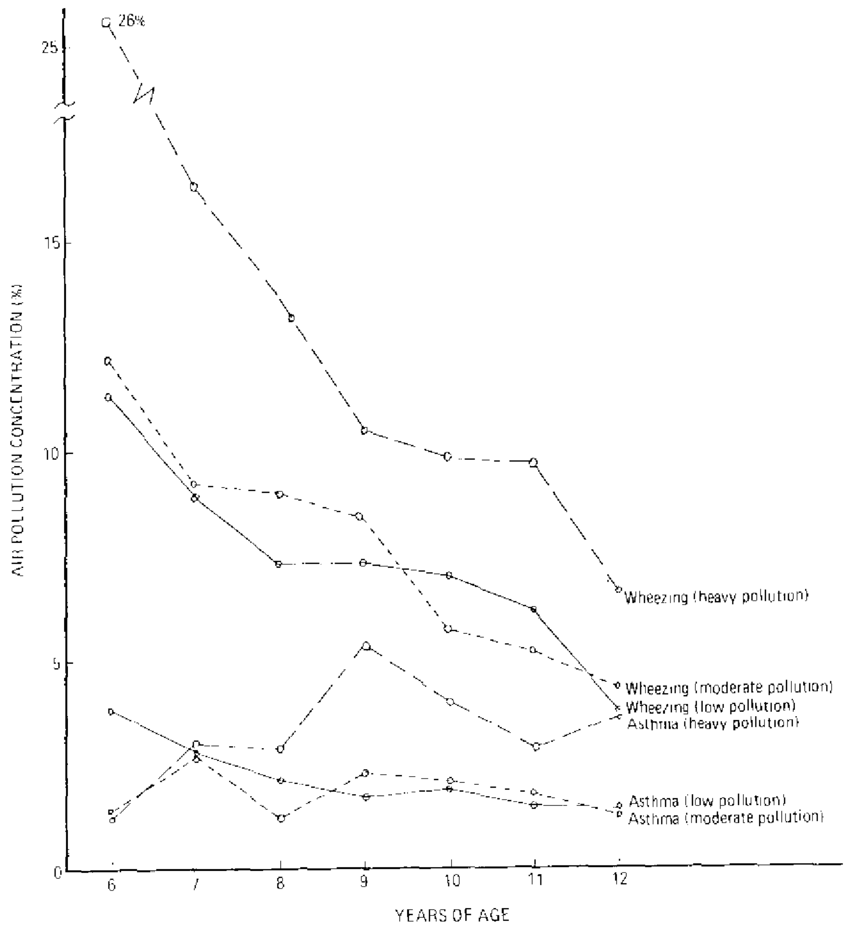

FIG.5. Prevalence rate for asthma and wheezing, as a function of age and air pollution concentration.

4.3. Results

The questionnaire recovery rate was higher than 97.7% and the effective respiratory function measurement rate was higher than 96.5%. In the prevalence rates for asthma and wheezing a difference was observed between males and females, as shown in Table II. The prevalence rates for males was higher than those for females, the ratio of male value to female value being 1.9 for asthma and 1.25 for wheezing. The prevalence rates for asthma were more than 2.4 times higher and the prevalence rates for wheezing more than 1.7 times higher in the heavily polluted areas than those in the low-pollution areas. The prevalence rates for asthma and wheezing as a function of age are shown in Fig.5 for the areas

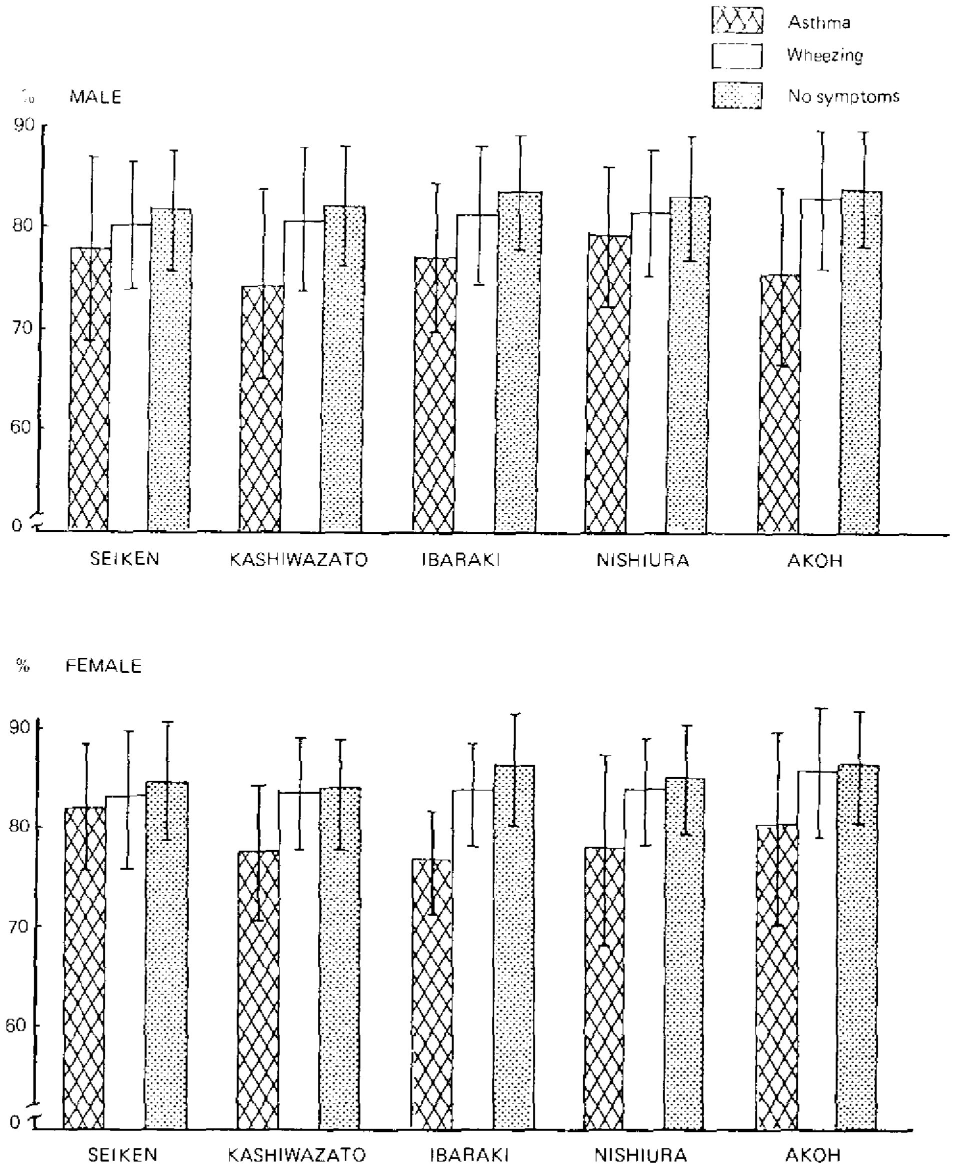

FIG.6. Respiratory function values (% $FEV_{0.75}$) of the three symptom groups in five areas (mean ± 1 SD).

of different levels of pollution. For all age groups, the prevalence rates were higher in the heavily polluted area than in the low and moderately polluted areas. However, no significant differences were observed between the low-pollution and the moderate-pollution areas.

The measurement of immunological serum IgE concentrations were conducted for the *asthma* group and the *wheezing* group. For the *asthma* group, higher values than 300 i.u./mltr were observed in 85% (83.7–100%) which is quite a high percentage, with 59% (56.4–75%) for the *wheezing* group, which is a lower percentage than the *asthma* group. However, no significant differences were observed between the different areas.

According to the results of respiratory function tests for males and females, the *asthma* group showed the lowest value, the *wheezing* group the intermediate value, and the *healthy* group the highest value.

In Fig.6, the respiratory function values ($\%FEV_{0.75} = FEV_{0.75}/FVC \times 100\%$) of the three symptom groups in five areas are shown. Some difference can be seen between the three groups with the lowest values in the *asthma* group. There were some differences between the areas. For the *healthy* and the *wheezing* groups, the lowest value was observed in the high-pollution area and the highest in the low-pollution area. These differences were considered statistically significant. For the *asthma* group, no significant regional difference was observed. It may be considered that the validity of classification into three symptom groups based on the questionnaires is supported by the results of immunological serum IgE tests and respiratory function tests.

Indoor air pollution may be considered a domestic environmental factor. Passive smoking by schoolchildren and the indoor combustion of fossil fuels have been examined. Passive smoking by schoolchildren was considered positive (+) when there was even one smoker among those who live together and negative (–) when there was no smoker in the house. Indoor combustion of fossil fuels was classified as either *clean* or *dirty*, according to the method of heating. *Dirty* applies to a house where a method of heating that causes indoor air pollution is used. *Clean* applies to a house where a clean method of heating which does not cause indoor air pollution (e.g. an electric stove or central heating) is used.

The results of these surveys indicate (see Fig.7) that the prevalence rates for wheezing have a tendency to increase owing to passive smoking and dirty indoor combustion. Since some effects of indoor air pollution on the *wheezing* group were observed, these effects on the wheezing prevalence rates were further studied with respect to the areas of different levels of air pollution. The results are shown in Fig.8, which shows that the wheezing prevalence rate is the lowest (4.5%) in the low-pollution area and when there is no smoker in the family and clean heating, while it is the highest (13.3%) in the heavily polluted area and when there is a smoker in the family and dirty heating. From these results it

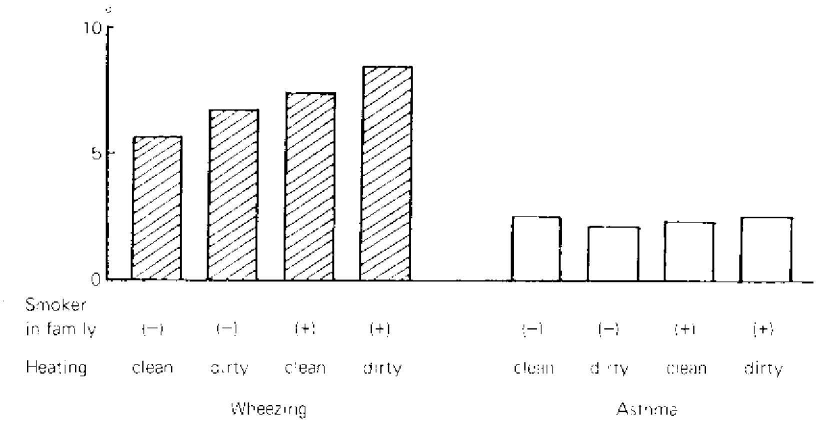

FIG. 7. Effects of family environmental air pollution (passive smoking and type of room heating).

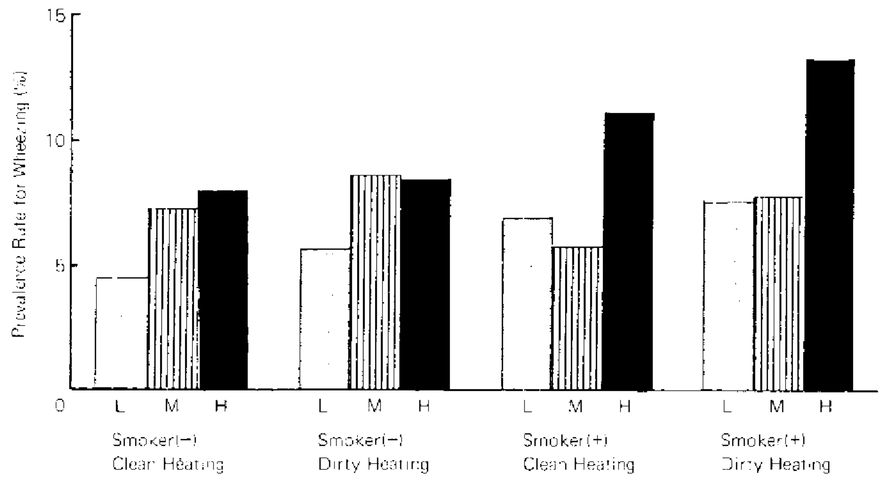

FIG. 8. Effects of family environmental air pollution (passive smoking and type of room heating) for three levels of ambient air pollution.
H = heavy pollution; M = moderate pollution; L = low pollution.

seems possible that the wheezing prevalence rates would be influenced not only by the general atmospheric pollution but also by indoor air pollution.

The individual exposures to NO_2 were measured by the simplified personal NO_2 monitoring badge. The period of investigation was not the same for different areas and the period of room-heating in different areas was not surveyed. Therefore, a comparison of the data between different areas would not be significant, but a comparison of the data on personal exposure to NO_2 in the same area between different types of heating was made. The NO_2 exposures of schoolchildren from a house with dirty heating tend to be higher than those with clean heating. This tendency is especially clear in the low-pollution area of Akoh. The difference due to passive smoking was not clear. No significant difference was observed between the *no-symptoms* group and the *asthma* and *wheezing* groups. The results of NO_2 exposure, as given above, might have been obtained because the survey was made during winter in order to note the effects of indoor heating.

The relative positions of the six survey areas (Fig.2) are quite different, and the period of room-heating may also be different between Saito in the south and Akoh and other areas in central Japan.

5. DISCUSSION AND CONCLUSIONS

The health effects of SO_2 and NO_2 have been emphasized in this paper. However, as explained in the Introduction, a number of other pollutants are also released to the environment by combustion of fossil fuels, one of the most important of which is the natural radioactivity in fossil fuels.

5.1. Natural radioactivity [8, 14–28]

Coal has been known to contain traces of uranium which have been absorbed on the carbonaceous material during remote geological periods. The size of this deposition depends not only on the availability of coal and uranium but also on the presence of a medium by which the uranium can be brought into contact with the coal bed. Uranium concentrations may therefore differ considerably among individual coal deposits. Oil is a carbonaceous material which may also absorb uranium.

Table III show the natural radioactivity in ash of coal and coal-oil mixture obtained from Japanese power plants nearly 20 years ago. It was requested that the data should not be published at that time because of the very sensitive attitude of the Japanese population towards radioactive contamination of the environment. The coal mines in central and southern Japan, from which the samples of coal were obtained, were closed down more than ten years ago. A large fraction of coal consumed in Japan is at present imported. The annual coal production in Japan is about 10×10^6 t, of which 6×10^6 t are coking coal and 12×10^6 t steam coal. The average ash content of coking coal is about 7.5–8%, with the lowest value 6% and the highest 9%. The average ash content of steam coal is about 20%, with the lowest value 9% and the highest 35%.

TABLE III. NATURAL RADIOACTIVITY IN ASH OF COAL AND COAL-OIL MIXTURE OBTAINED FROM JAPANESE POWER PLANTS [8, 15–17]

	Radioactivity ($Bq \cdot kg^{-1}$ ash)			
	Pb-210	Ra-226	Ra-228	Th-228
Coal-ash:[a]				
Central Japan	7.3×10^2	3.7	0.43	5.7×10^2
Southern Japan	3.0×10^2	36	17	87
Northern Japan	3.9×10^3	23	54	2.5×10^2
Coal-oil:[a]				
Mixed ash[b]	24–120	35–40	37–77	44–76

[a] Samples received nearly 20 years ago from Japanese power plants.
[b] Received as ash from fossil-fuel power stations where both coal and oil are used, but the exact proportion of coal and oil was not identified.

Note: In the UNEP report ERS–1–79 the following assumptions are made for release conditions based on the report by McBRIDE, J.P., et al., Nucl. Saf. **19** (1978) 497:

(1) The coal contains 1 ppm U and 2 ppm Th;
(2) Ash release is 1%;
(3) Rn-220 is produced from Th-232 in the combustion gases at the rate of 5.1×10^4 $Bq \cdot s^{-1} \cdot kg^{-1}$ Th;
(4) The annual release of natural U is 23.2 kg and of Th-232 46.4 kg;
(5) 15 seconds are required for the gases to travel from the combustion chamber to the top of the stack;
(6) Release per 1000 $MW(e) \cdot a^{-1}$: U-238 chain (Ra-226, Pb-210, etc.) 3×10^8 $Bq \cdot a^{-1}$; Th-232 chain (Th-228, Ra-228, etc.) 1.9×10^8 $Bq \cdot a^{-1}$; U-235 chain 1.3×10^7 $Bq \cdot a^{-1}$; Rn-220, 1.5×10^{10} $Bq \cdot a^{-1}$; Rn-222, 3×10^{10} $Bq \cdot a^{-1}$.
(Converted to SI units)

Radioactive analysis of Japanese coal has recently been attempted at the Radiation Centre of Osaka prefecture and other institutes in Japan. Some preliminary results are as follows:

(a) U and Th content in coal, estimated by activation analysis:

U: 5.3 ± 0.1 $\mu g \cdot g^{-1}$ (66.6 $Bq \cdot kg^{-1}$)
Th: 16 ± 0.5 $\mu g \cdot g^{-1}$ (66.6 $Bq \cdot kg^{-1}$)

The values in parentheses are the radioactivity in ash, assuming radioactive equilibrium and no loss by combustion.

(b) Results of γ-spectrometry of the daughters of the same sample:

U series: Pb-214 – 66.6 ± 3.7 $Bq \cdot kg^{-1}$ ash
Bi-214 – 66.6 ± 3.7 $Bq \cdot kg^{-1}$ ash

(Some of the gaseous Ra daughters in the U series might have escaped immediately after combustion.)

Th series: Ac-228 – 70.3 ± 7.4 $Bq \cdot kg^{-1}$ ash
Pb-212 – 70.3 ± 7.4 $Bq \cdot kg^{-1}$ ash
Tl-208 – 21.83 ± 1.48 $Bq \cdot kg^{-1}$ ash

Since the coal samples and the time and method of measurement are different, a direct comparison of these results with the results in Table III may be difficult. The radioactivity of Th-228 in coal-ash in central and northern Japan in Table III appears to be quite high. We must,however, take into consideration the possible concentration of radioactivity at the time of ashing. In the above preliminary analysis the average concentration ratio may be assumed to be about 5, but in general it may depend on the content of ash, the mode of combustion and the type of coal.

Table IV lists some of the natural radioactivity in coal and ash reported by different authors as well as the specific radioactivity ratios (ash/coal). It may be noted that the very fine particles which go through the dust collector may adsorb radioactivity and tend towards higher radioactivity. However, the radioactivity released from both fossil-fuel and nuclear power plants is small under normal operating conditions and below the maximum permissible limits of ICRP, as pointed out in the 1977 UNSCEAR report [28]. It should be noted that, on the average, a member of the world population receives a whole-body dose of about 1 $mSv \cdot a^{-1}$ from natural radiation and about 0.5–0.8 $mSv \cdot a^{-1}$ from man-made sources (medicine, nuclear explosions, power production, etc.) at present, of which less than 0.01 $mSv \cdot a^{-1}$ is due to radiation from coal and the nuclear power industry.

5.2. Comparison of risks [8, 14–31]

It is clear from the above that the effects of chemical pollutants such as SO_x and NO_x from fossil-fuel power plants would be much more significant than those of radioactivity. For reference, the hazard indices of SO_x, NO_x and radioactivity are given in Table V.

The differing nature of the hazards make it difficult to compare the possible hazards due to chemical effluents from fossil-fuel plants with those due to radioactive effluents from nuclear power plants. Even though some quantitative

TABLE IV. NATURAL RADIOACTIVITY IN COAL AND ASH FROM POWER PLANTS AND RELATIVE RATIO OF SPECIFIC RADIOACTIVITY (ASH/COAL) [8, 14–27]

Radionuclides	Coal ($Bq \cdot kg^{-1}$)	Ash			
		Dust collector inlet ($Bq \cdot kg^{-1}$)	Radioactivity ratio (ash/coal)	Dust collector outlet ($Bq \cdot kg^{-1}$)	Radioactivity ratio (ash/coal)
Th-228	21	89	4.2	104–122	5.0–5.8[a]
	25.9–92.5[b]	77.7–177.6[b]	1.2–5.3[b]	77.7–177.6[b]	1.2–5.3[b]
Th-232	8.5	81	9.5	107	11
Ra-226	24	107	4.5	122–218	5.1–9.1[a]
	3.7	22	6.0	37	10
	14.8–48.1[b]	51.8–170.2[b]	1.5–7.7[b]	51.8–170.2[b]	1.5–7.7[b]
U-238	32	130	4.1	200–444	6.3–14[a]
	27	370	14		
Pb-210	25	81	3.2	159–629	6.4–25[a]
	23	851	37	1406	61
	30	148	4.9	740	25
Po-210	25	1036	45	1554	68
K-40	55.5–347.2[b]	188.7–284.9[b]	0.87–3.8[b]	188.7–284.9[b]	0.87–3.8[b]

[a] Depend on type of coal, mode of combustion, type of dust collector, and size distribution of the particulate matter, but the very fine particles that go through the dust collector may adsorb radioactivity and tend to higher radioactivity.

[b] Highest and lowest values are given based on Indian data. Ash content is reported to be about 25%. The radioactivity ratio was calculated based on individual corresponding values, and the ranges are indicated. Whether the values of ash are at the inlet or outlet of the dust collector is not mentioned in the original report and therefore the same values are given in both the inlet and outlet columns of the table.

TABLE V. HAZARD INDICES [Q/MPC] OF SO_x AND NO_x FOR 1000 MW(e) FOSSIL-FUEL POWER PLANTS COMPARED WITH THOSE OF RADIOACTIVITY (in 10^9 m^3) [15–18]

Type of power plant	SO_x	NO_x	$(SO_x + NO_x)$	Radio-activity
Coal-fired power plant[a]				
Lower estimate[b]	4.23×10^5	2.39×10^5	6.62×10^5	110
Higher estimate[c]	10.6×10^5	3.59×10^5	14.2×10^5	700
Oil-fired power plant[a]				
Lower estimate[b]	1.42×10^5	2.19×10^5	3.61×10^5	2.8
Higher estimate[c]	3.56×10^5	3.3×10^5	6.86×10^5	8.0
Nuclear power plant[d]				
Lower estimate		–	–	0.3
Higher estimate	–		–	40

[a] Airborne effluents given in UNEP ERS-1-79 [11] are used as the value of Q (in t/1000 MW(e)·a).
SO_x: 1.1×10^5 (coal), 3.7×10^4 (oil);
NO_x: 2.7×10^4 (coal), 2.48×10^4 (oil);
CO: 2×10^3 (coal), 7.1×10^2 (oil);
Hydrocarbons: 4×10^2 (coal), 4.7×10^2 (oil);
Aldehydes: 2.4×10^2 (oil);
Particulates: 3×10^3 (coal), 1.2×10^3 (oil).

[b] Higher ambient air quality standards given in Table I are assumed as MPC values.

[c] Lower ambient air quality standards given in Table I are assumed as MPC values.

[d] The overall hazard index

$$V = \sum_i \frac{Q_i}{(MPC)_i}$$

for radioactivities is based on Ref. [18]. The hazard index is the volume of air necessary to dilute the airborne effluents, Q, down to the MPC. The hazard indices of radioactivity for fossil-fuel power plants are based only on the natural radioactivities released by combustion of fossil fuels. MPC values are based on ICRP reports. The average values of hazard indices for LWRs are estimated to be about 7–8 in this case.

Note: At standard conditions (25°C and 760 torr), ppm by volume may be converted into $\mu g \cdot m^{-3}$ by the relationship:

$$\mu g \cdot m^{-3} = \frac{10^3 \text{ (ppm) (molecular weight of pollutant)}}{24.5}$$

TABLE VI. COMPARISON OF EQUIVALENT RISKS UNDER DIFFERENT ASSUMPTIONS

A. Average risk of 10 person-days loss per year in 1% of the population corresponds to:

(1) Average risk to population:

9.1×10^{-6} equivalent deaths per year

if we assume the average number of years remaining to the population to be 30 years (1.1×10^4 days).

(2) Average risk to individuals:

3.7×10^{-6} equivalent deaths per year

if we assume the average life span of the individual to be 75 years (2.7×10^4 days).

B. Average risk of 10 occupational person-days loss per year in 1% of the occupational population corresponds to:

Average risk to occupational population:

1.7×10^{-5} occupational equivalent deaths per year

if we assume the average number of working years remaining to the population to be 20 with 300 working days per year, then 6000 lost working days would be equivalent to one occupational person-death.

relationships have been established between the effects and the contamination levels for various types of pollutants, it may still be difficult to make direct comparisons between the different types of risks. For instance, one type of hazard might result in a non-fatal immediate public nuisance to a large percentage of population, while another might be fatal in the remote future, though with a very small probability of occurrence. For these reasons, Nishiwaki et al. pointed out in 1970 [8] that it is necessary to establish some basic philosophy of 'risk versus risk', or a basic principle with which to weigh one type of risk against another.

One of the very preliminary examples of comparing stochastic fatal risks such as cancer and leukaemia due to low-level radiation with other types of risks such as bronchitis, asthma or wheezing due to exposure to non-fatal levels of chemical pollutants may be introduced for a hypothetical case. We assume that, on an average, about 10 days sick leave per year would be necessary because of health hazards caused by non-fatal levels of chemical pollutants, and that such cases were observed in about 1% of the exposed population with an average number of remaining effective days of life of about 10^4.

If we further assume that 10^4 person-days lost in this population would correspond to one equivalent life lost, then the average risk of the chemical contamination level which causes a loss of 10 person-days per year in 1% of the population would be 10^{-5} per year. If we use the average lifespan instead of the average remaining days of life, this value of annual risk would be reduced by a factor of 2 to 3. In the case of occupational risk, one death is usually assumed equivalent to 6000 lost person-days of work. If we use this value, the estimated risk would be correspondingly higher. In Table VI,equivalent risks are calculated under different assumptions.

According to the ICRP, the stochastic risk due to ionizing radiation is estimated to be about 10^{-2} per sievert, i.e. the average risk of 10^{-5} per year may be considered to correspond to 1 $mSv \cdot a^{-1}$ of exposure to radiation, which is comparable to the acceptable level of risk and complies with the dose-equivalent limit for individual members of the public.

It seems to be extremely important to establish basic principles with which to compare the risks with high probability and low consequences and the risks with low possibility and high consequences. If we consider the risk of hereditary effects (which may appear in remote descendants quite scattered in space and time) due to prolonged irradiation of a large population with a relatively low dose-rate by fallout and the direct immediate risk of atomic bombing of densely populated industrial urban areas, it may also be necessary to have some principle to weigh diluted risk and concentrated risk with respect to time and space.

5.3. Risk of risk reduction [8, 30, 32, 33]

The basic principle of 'risk versus risk' may also be applicable to the optimization of environmental protection and safety. If the risk associated with a certain industry is larger than that associated with the risk-reduction measures, the application of such risk-reduction methods may be justified. However, if the risk associated with the risk-reduction measures becomes higher than the risk to be reduced, the application of such risk-reduction methods may not be justifiable. For instance, in a specific industry, if one applies additional filters, tanks, piping systems, valves, etc., for environmental protection and safety, the risk of that particular industry may be greatly reduced, but if the risks associated with the industries producing the risk-reducing equipment becomes higher than the risks of that particular industry, the overall risk to society may be increased.

The basic principle of cost-benefit analysis or risk-benefit analysis seems difficult for some people in Japan to accept because the perception of risk and the perception of benefit are often quite different among different groups of people, and thus social conflict may increase. If social conflict increases, the social risk may also increase rather than decrease. The procedures for quantifying health effects in monetary terms have often been questioned in discussions on safety. It may therefore be important to base the optimization of environmental protection and safety on social harmony, taking into consideration not only technical, economic and social factors but also socio-political and socio-psychological factors.

Cost-benefit or risk-benefit analysis is an important basic principle required by specialists in commercial transactions, in engineering design, in drawing up regulatory measures, or in the operation of an industrial facility. Cost-benefit analysis and optimization are often made intuitively by businessmen. However, the idea of weighing the risk of human life against the benefit to industries for justification of a certain level of risk to the public may not be easily accepted by the people. In such cases, if it is explained that the excess investment on safety may increase the overall risk to society based on the principle of 'risk versus risk', people may much more easily understand a situation where a certain level of risk may be unavoidable in society.

Whether the reduction of one type of risk might lead to an increase of another type of risk, or whether a severe restriction of one type of pollution might increase another type of pollution, or whether a heavy burden imposed on industries to reduce one type of risk might create another type of risk in society, are possibilities that must be carefully considered by decision-makers on environmental protection and safety.

In assessing the risk of risk-reduction equipment industries, all the risks would have to be considered, including not only those of the industry directly producing the equipment but also those associated with the raw material industries. For instance, if the equipment is made of metal, the risks associated with mining, transport and production of metal must be included, in addition to the risks to the industry and the transport of the end products.

According to Black and Niehaus [30], the specific risk of producing safety equipment is estimated to be about 3×10^{-2} equivalent deaths or 180 equivalent lost person-days of work per million US \$ of equipment. This suggests that the expenditure on safety at marginal costs of risk reduction higher than US \$33 million per equivalent life saved would actually lead to an increase in risk, and that the total risk cannot be reduced below any given limit. The total risk may first decrease effectively with a small investment on safety, but after reaching a minimum it may be expected to increase again with the excess investment on safety. Not only the risk-reduction industries but also the construction and operation of the risk-reduction system may not be completely riskless.

In the nuclear industry, almost every part may be more or less related to safety, and it is difficult to distinguish exactly the safety from the non-safety parts. The rate of accidents may depend not only on the equipment, but also on the workmanship at all levels of manufacture as well as the skill of the operators in the field, and may vary from one case to another. Therefore, it is difficult to estimate a universally applicable value of the specific risk to industries. However, for the construction of nuclear power stations in Japan, the specific risk may be estimated as less than 10^{-2} equivalent deaths per million US \$ or about one order of magnitude smaller than the above value in some cases.

According to an expert in the Japanese Ministry of Transport, although it may be quite different in different types of engineering, the minimum point of risk may be about the order of 10^{-7} and therefore if the risk is reduced to this level one may be justified in using the term 'absolutely safe' in the ordinary sense of the words used by the general public in daily life. However, these risks associated with various industries may tend to decrease with the development of automation technology and improvement of workmanship, and would therefore be a function of time in general.

When the resources allocated for safety are limited, it would be important to consider optimizing the distribution of financial resources for the most effective way of achieving overall safety. If too much money is spent to reduce only one particular risk, the other risks with much higher probability of occurrence, which could be reduced effectively with much less expenditure, may not be reduced sufficiently owing to shortage of money. In such a situation, the overall risk to society may be increased by undue over-emphasis of only one particular type of risk. It is therefore important to ensure that people have the proper feeling for safety, corresponding to the relative magnitude of the objectively estimated risks, especially for decision-making [32].

Justification of a certain level of risk based on the 'risk versus risk' principle and harmonization of the safety system with social benefit may be important in achieving the well-balanced overall safety of society.

5.4. Risk and benefit [8, 29, 33, 34]

When the term 'risk' is used, the meaning may have to be guessed from the context. There are many different types of risks: natural and artificial, occupational and non-occupational, voluntary and involuntary, enjoyable and unenjoyable, individual and social or public, acceptable and unacceptable, etc. Depending on the types of risks, their perception by the individual would be quite different and their public perception and acceptability would also be different. If these factors are ignored, it may be difficult to obtain public acceptance of a risk [34].

Whatever the meaning of 'risk' may be, it may not be easy to define a level of risk which alone could be accepted unconditionally by the public. The risk may be considered acceptable by a person if a greater benefit is received by the same person. What is called acceptable risk or accepted risk seems to be the level of risk to which people are accustomed and at which they do not worry about the risk, particularly if the level of risk is not too high to tolerate. Even in these cases, it is questionable whether people really 'accepted' the risk without any benefit or without any conscious or subconscious consideration of the 'risk versus risk' principle.

If people very much like doing something, they will do it despite the high risk (voluntary, enjoyable and acceptable risk, for example motorcycling by young people). If people dislike something, they will not accept it despite the low risk (non-voluntary, unenjoyable and unacceptable risk, such as the objectively low but subjectively very high risk of nuclear energy perceived by anti-nuclear people). Judging by these two extreme cases, it seems that the strong positive perception of benefit may play an important role in the public acceptance of new technology, such as motorcycles or nuclear energy. However, the image of fear of the unknown and the confusion of peaceful uses of nuclear energy with the fear of and anxiety about nuclear weapons must first be eliminated by effective public education and unprejudiced information. Public acceptance is an increasingly important constraint to be considered by those responsible for technological policies.

In view of the difficulties in public acceptance of the concept of cost-benefit or risk-benefit analysis when the benefit groups and the risk groups are different, the basic principle of environmental protection in Japan seems to be based on the administrative goal of keeping the adverse effects of environmental contamination or pollution to an insignificant level or within the fluctuation range of the naturally occurring effects of a similar type [33].

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ASSESSMENTS OF LONG-TERM EFFECTS OF CO_2 AND ^{14}C: VARIOUS ENERGY SCENARIOS

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Abstract

ASSESSMENT OF LONG-TERM EFFECTS OF CO_2 AND ^{14}C: VARIOUS ENERGY SCENARIOS.

A non-linear model for the global carbon cycle has been developed and applied for prognostic assessment of concentrations of CO_2 from the combustion of fossil fuel and of radiocarbon released from facilities of the nuclear fuel cycle. The model is built up from two boxes for the atmosphere (stratosphere, troposphere), three boxes for the ocean (mixed surface layer, deep sea and sediments), and two boxes for the biosphere (short- and long-lived biota) with non-linear troposphere-biota and troposphere-ocean surface layer exchange rates and linear fluxes between the other reservoirs. Two different models are used for the man-made reduction of the biomass: (a) no deforestation function, and (b) slightly growing deforestation function. The biota growth factor, the exchange of the atmospheric CO_2 with the ocean and the pre-industrial atmospheric CO_2 content were adapted to agree with the records of the atmospheric CO_2 concentration in Mauna Loa, the Suess effect until 1954, and the dynamic response to the ^{14}C from nuclear weapon tests. The three scenarios considered are: (I) annual energy growth rates of 2% and 4%, no nuclear power; (II) an upper, lower and medium estimate of replacement of fossil fuels by nuclear power. In addition, two assumptions concerning the decontamination of ^{14}C in the nuclear power plant effluents were made: one in which ^{14}C is released completely, and one with a decontamination factor of 4. Assuming logistic source functions for the increase of fossil-fuel combustion and an exponential growth of nuclear power until the year 2020, by around 2100 the CO_2 concentration of the troposphere will reach concentrations twice to five times as high as the pre-industrial level. Various environmental effects of this increasing CO_2 level are briefly discussed. The specific ^{14}C activity of the atmosphere is decreased. Up to the year 2200, the specific activity will be lower than the pre-industrial level. The individual lifetime dose commitments (70 years) are found between 0.85 and 0.45 mSv (pre-industrial value: 0.73 mSv).

1. INTRODUCTION

Environmental impacts and related health risks from atmospheric releases of CO_2 from fossil-fuel combustion and of radiocarbon from nuclear facilities have been of increasing interest in the past decade. Both pollutants are globally distributed on earth and enter the biosphere by the photosynthetic activity of plants. Many authors have emphasized climatological consequences and potential health hazards by an increasing CO_2 content of the atmosphere. On the other hand, radiological effects of ^{14}C (e.g. on DNA by energy deposition in human cells) could result in somatic health effects (malignant neoplasms or leukaemia) or genetic diseases. Although the long-term effects of both carbon isotopes are quite different, the global cycling can often be simulated by the same model. Such a simulation makes it possible to calculate the possible future effects of increasing fossil-fuel combustion as well as the potential radiation exposure to the population of the world by increasing nuclear power capacity.

Here we mainly discuss various energy scenarios and the influence of releases of non-radioactive and radioactive carbon on the atmospheric CO_2 content and on the specific activity of the biosphere.

2. THE MODEL FOR THE GLOBAL CO_2 AND ^{14}C CYCLE

Various models for the distribution of carbon in the environment have been developed in the last few years. CO_2 is by far the most important carbon compound, compared to which all others (e.g. CO, CH_4, C_2H_8) may safely be neglected in most cases. The cycling of CO_2 can be simulated, e.g. by a compartment model. We based our seven-compartment model mainly on that of Bacastow and Keeling [1] (Fig.1). This model is built up from two boxes for the atmosphere (stratosphere, troposphere), three boxes for the ocean (mixed surface layer, deep sea and sediments), and two boxes for the biosphere (short- and long-lived biota). The exchange rates were assumed to be non-linear between the troposphere and the biota, on one hand, and the ocean surface layer on the other, and to be linear between the other reservoirs.

The troposphere is the connecting link to the biosphere as well as to the ocean. The uptake of CO_2 from the troposphere into the mixed surface layer is determined by physical processes with subsequent chemical transformations, i.e. the formation of bicarbonate and carbonate. The uptake depends on several factors, such as pH value, alkalinity, borate and phosphate content, etc., and therefore no longer follows a simple linear mechanism. All the various physico-chemical factors can be represented by introducing a 'buffer-factor' [1]. Its value now lies at about 10; with further increase of CO_2 in the atmosphere it can rise to 30 or even higher.

A second non-linear flux of carbon exists between the troposphere and the biosphere. The time constant for the exchange with the short-lived biota lies in the range of two to three years; that for the long-lived biota at about 60 years. Since the biomass production is higher for an increased CO_2 level in the atmosphere, a logarithmically growing function has been introduced for its simulation. To obtain consistency between this model and experimental data, the biota growth factor β must be between 0.0 and 0.6 [1].

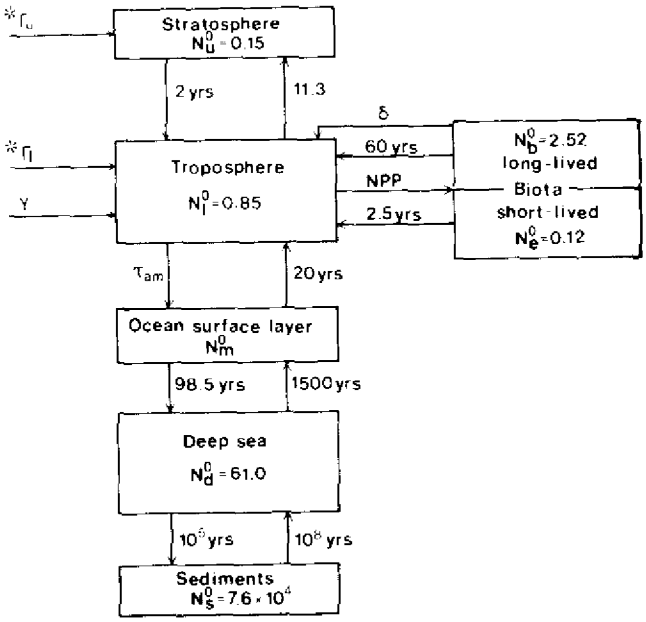

FIG. 1. Compartment model for the global carbon cycle. Values for N_i^0 are given as multiples or fractions of the pre-industrial atmospheric carbon content, N_a^0.

The mobilization of carbon from the long-lived and short-lived biosphere can be represented by an additional exchange rate, the 'deforestation function' δ. At present, this annual input of CO_2 into the atmosphere is estimated at $3.0 \pm 1.5 \times 10^{15}$ g C·a^{-1} [2, 3]. To study the influence of this input function on the response of the system, we examined two different models:

Model A: Without deforestation function

Model B: With a function δ(t) increasing exponentially at an annual rate of 3% until the above value of 3×10^{15} g C·a^{-1} was reached in 1978

For model B we assume an exchange of 5% from the short-lived biota and 95% from the long-lived biota, according to Zimen et al. [4].

Both models have been tested for consistency with experimental data for the increase of the atmospheric carbon content as measured at Mauna Loa [1, 5], the Suess effect [6], and the variation of the specific activity in the atmosphere and the ocean due to atomic bomb ^{14}C [7]. We fit these experimental data by varying the following most sensitive parameter values within their reasonable ranges: the pre-industrial atmospheric carbon content N_{a0}, the exchange rate of the

TABLE I. PARAMETER VALUES USED FOR THE GLOBAL CARBON MODEL

Parameter	Model A	Model B
$\delta(t)$	0.0	Exponential growth rate $\alpha = 3\%$
β	0.2	0.44
N_{m0}	$3 \times N_{a0}$	$4 \times N_{a0}$
τ_{am}	7 years	5 years
N_{a0}	293 ppm	284 ppm
Suess effect 1954	−2.4%	2.9%

atmosphere with the ocean τ_{am}, the carbon content of the ocean surface layer N_{m0} and the biota growth factor β. The other time constants and pre-industrial carbon pools were held fixed (Fig.1). The best estimates for the variable parameters as obtained by this method are given in Table I.

In the global model for ^{14}C, the specific activity, i.e. the ratio $^{14}C/C_{total}$, was calculated instead of the absolute concentrations. An isotope discrimination factor of 0.96 for the transfer of ^{14}C from the atmosphere to the biosphere and to the ocean surface layer, and of 0.83 into the deep sea, had to be assumed [8].

Both sets of differential equations were numerically solved by the 4th-order Runge-Kutta method. Then the specific activity in each compartment i at time t was calculated by

$$A_i(t) = \frac{{}^*n_i(t) + {}^*n_{i0}}{n_i(t) + N_{i0}}$$

3. THE ENERGY SCENARIOS

Since most estimates for future energy supply contain great uncertainties, we shall consider various scenarios in order to obtain some information of the range of future ^{12}C and ^{14}C content in the different compartments. The data proposed by the Workshop on Alternative Energy Strategies (WAES [9]) were mainly used. Similar scenarios have been given by the World Energy Conference [10].

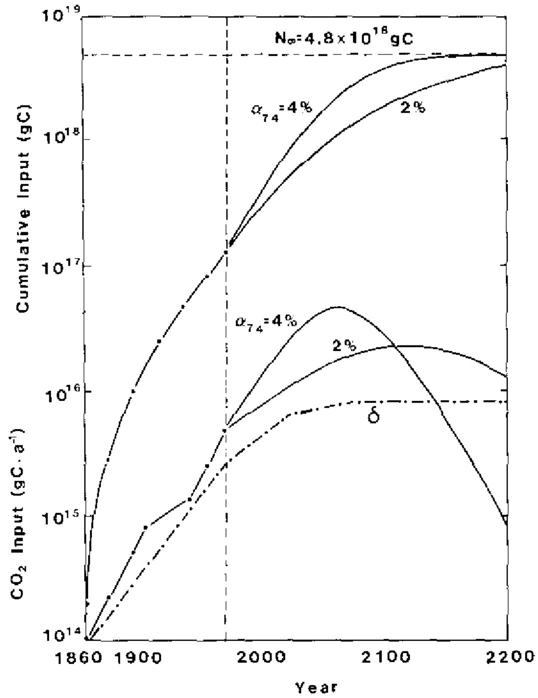

FIG.2. Annual and cumulative CO_2 input into the atmosphere: α_{74} = annual growth rate at 1974; δ = non-fossil carbon input.

Two main scenarios were assumed with the following variants:

Scenario I:

The energy consumption increases and it is provided only by fossil fuels. We used a logistic growth function, as did Zimen et al. [4] and Killough [11]. The annual input is described by

$$\dot{n} = \alpha_{74}\, n \left(1 - \frac{n}{n_\infty}\right)$$

and the cumulative input by

$$n = \frac{n_\infty}{1 + \left(\dfrac{n_\infty}{n_0} - 1\right) e^{-\alpha_{74} t}}$$

The two variants of Scenario I, α_{74} = 2% and 4% (n_∞ = 4800 × 10^9 g C), are given in Fig.2 up to the year 2200.

TABLE II. VALUES ASSUMED FOR THE SCENARIOS CHOSEN

Scenario	α_{74} (%)	GW(e) installed capacity	
		In 2000	In 2020
I(a)	2.0	–	–
I(b)	4.0	–	–
II(u)	2.0	2000	5500
II(m)	3.4	1400	3800
II(ℓ)	4.0	900	2500

Scenario II:

In the second scenario a certain part of the energy consumption is assumed to be supplied by nuclear energy. The following three variants are chosen to estimate upper, medium and lower values for the ^{14}C exposure (Table II).

(u): An upper limit with a low increase of fossil fuel combustion (α_{74} = 2%) and a high world-wide extension of nuclear energy production (2000 GW(e) in the year 2000 and 5500 GW(e) in 2020).

(m): A medium estimate with a moderate increase of fossil-fuel consumption (3.4%) with partial replacement by nuclear energy (1400 GW(e) in the year 2000 and 3800 GW(e) in 2020).

(ℓ): A lower limit with rather fast growing consumption of fossil fuels (4%) and only slowly increasing nuclear capacity (900 GW(e) in the year 2000 and 2500 in 2020).

The input of carbon from fossil fuel combustion into the atmosphere is described by logistic growing functions for all variants. For the non-fossil input it is assumed that δ is proportionally linked to the world population growth [12]. The release of ^{14}C by nuclear installations depends on the installed nuclear capacity, the production rates in the various reactor types, the degree of retention and the fractional contribution of each reactor type to the overall capacity.

An exponential growth of the nuclear capacity was assumed within the limits given above. After the year 2020, the projections are even more uncertain, and

TABLE III. CALCULATED ^{14}C PRODUCTION RATES IN LIGHT-WATER REACTOR TYPES

Reactor type	Coolant ($Bq \cdot W^{-1} \cdot a^{-1}$)	Fuel elements ($Bq \cdot W^{-1} \cdot a^{-1}$)	Sum ($Bq \cdot W^{-1} \cdot a^{-1}$)
BWR	518 (14)[a]	702 (19)	1221 (33)
PWR	222 (6)	555 (15)	777 (21)

[a] $Ci \cdot GW^{-1} \cdot a^{-1}$ are in parentheses. 1 Ci = 3.7 × 10^{10} Bq.

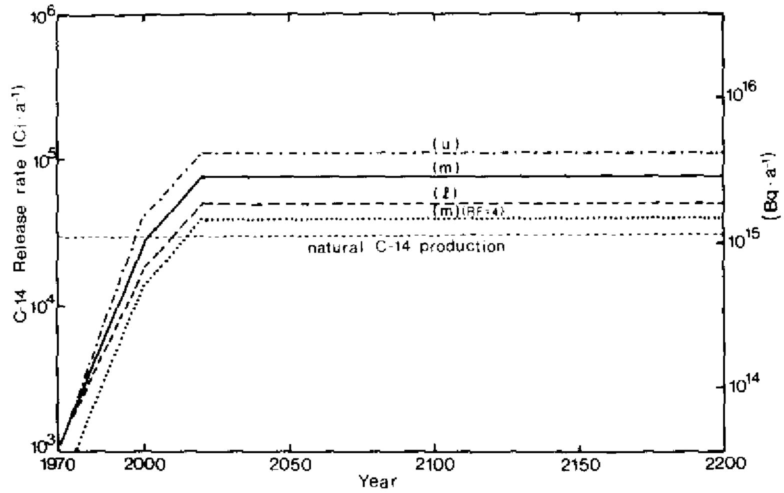

FIG.3. Release rates of C-14 from nuclear facilities for Scenario II for four variants (RF = retention factor at fuel reprocessing installations).

the exposure was therefore calculated only for constant release rates from 2020 until 2200.

In Table III the calculated ^{14}C production rates for the two most common reactor types are given (after Ref.[13]). Since until the year 2020 the predominant part of the capacity might be provided by light-water reactors, the assumption given in Fig.3 might be realistic. If, however, one third of the installed capacity is due to boiling-water reactors and two thirds to pressurized-water reactors, a release rate of 7.4 × 10^{11} Bq per GW·a is calculated (with 80% net

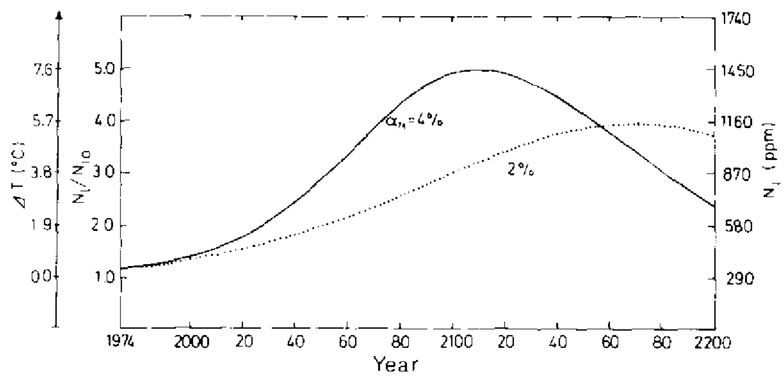

FIG.4. Relative and absolute carbon concentration in the lower atmosphere for annual logistic growth rates in 1974 of 2% and 4%.

production and no ^{14}C retention during reprocessing). To examine the influence of the retention of ^{14}C during reprocessing, the ^{14}C input was also reduced by a retention factor of 4 in Scenario II (m). The assumed release rates of ^{14}C for the various scenarios are given in Fig.3.

4. RESULTS AND DISCUSSION

4.1. Scenario I

For this scenario (only fossil fuels, no nuclear power) we assume that nuclear power will not contribute to the energy supply. Two different annual growth rates of 2% and 4% for the combustion of fossil fuels seem to be an upper and lower estimate.

In Fig.4, the time-dependence of the atmospheric carbon content is given for two different logistic growth rates. For the upper variant ($\alpha_{74} = 4\%$), the CO_2 concentration in the atmosphere will be doubled in the year 2030 and can reach levels up to five times the pre-industrial level. However, even for the low variant ($\alpha_{74} = 2\%$),CO_2 concentrations of three to four times the undisturbed level seem to be possible. A doubling of the CO_2 content in the troposphere could be reached in the year 2060, i.e. only 30 years later than for the high variant. The potential consequences of such high atmospheric CO_2 concentrations are still very uncertain at present. One of the major environmental impacts of an increase of the CO_2 content in the atmosphere could be a global warming up of the earth's surface air temperatures owing to the greenhouse effect (i.e. absorption of infra-red radiation from the earth's surface by CO_2). The estimates found in the literature on

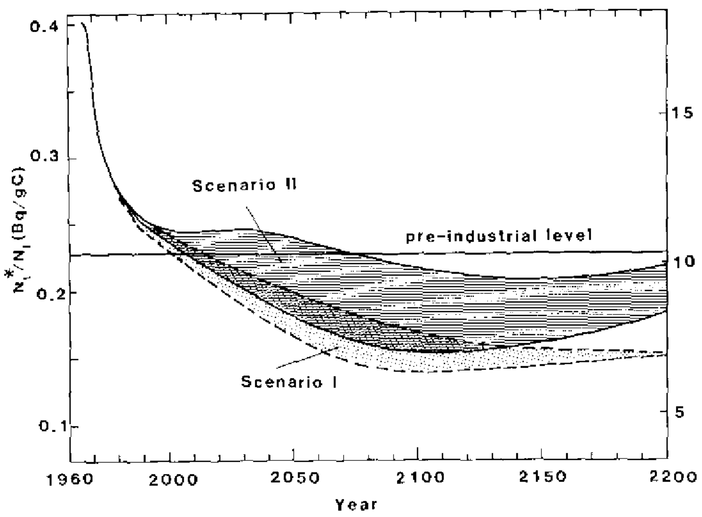

FIG.5. Ranges of the specific activity of the troposphere and effective dose rate.

temperature changes due to a doubling of the CO_2 concentration range from a low value of 0.26 K to a high value of almost 10 K [13–15]. A mean global surface air temperature rise of 4 K could possibly lead to melting of the arctic polar ice and larger climatological changes in most geographical latitudes [13]. Despite the great uncertainties in such assessments, reduction of CO_2 releases into the atmosphere seems today the best way to prevent global climatological changes.

Figure 5 shows the time-dependence of the specific activity of the atmosphere. For comparison, the pre-industrial level is given.

Human exposure to ^{14}C can occur by three principal means: external irradiation (only skin since ^{14}C is a pure β-emitter); internal irradiation after inhalation; and internal irradiation after ingestion.

Killough and Rohwer [16] showed that external β-irradiation contributes only less than 1% to the total body dose. It can therefore be neglected in these calculations. The internal dose can be conveniently calculated by a specific activity model [16, 17]. In this model it is assumed that the specific activity in man is equal to the specific activity in air. On this basis, dose-rate factors for the various organs have been calculated and combined to an effective dose conversion factor (Table IV). Hence, the effective dose rate can be expressed by a simple multiplication of the specific activity in the atmosphere with the effective dose conversion factor and is given on the right-hand side of Fig.5.

The releases of ^{14}C by nuclear weapons in the 1950s and 1960s will only be compensated by ^{12}C input and ^{14}C flux into the ocean after the year 2000.

TABLE IV. ORGAN AND EFFECTIVE DOSE FACTORS FOR ^{14}C ASSUMING THE SPECIFIC ACTIVITY MODEL

Organ	g $\left(\frac{Sv}{a} \Big/ \frac{Bq}{a}\right)$ ($\times 10^{-6}$)	w*	w·g $\left(\frac{Sv}{a} \Big/ \frac{Bq}{a}\right)$ ($\times 10^{-6}$)
Gonads	22.7	0.25	5.4
Breast	59.4	0.15	8.9
Lung	24.3	0.12	2.9
Red bone marrow	99.9	0.12	12.0
Thyroid	25.7	0.03	0.8
Bone surfaces	89.1	0.03	2.7
GI tract	43.2	0.24	10.4
Liver	35.1	0.06	2.1
Effective dose factor g_{eff}			45.2

* w = weighting factor.

A further decrease of the specific activity of the atmosphere will be expected owing to further input of non-radioactive CO_2 from the combustion of fossil fuels. According to the two assumed logistic growth functions, the minimum will be reached at about the years 2100 ($\alpha_{74} = 4\%$) and 2200 ($\alpha_{74} = 2\%$). Simultaneously, the effective dose rate will be reduced by about 30% to 40% as compared to the pre-industrial level.

The individual lifetime doses are calculated from the annual dose rates by integration over a mean lifetime of 70 years. These lifetime doses are given in Fig.6 as a function of the year of birth. The highest doses are caused by ^{14}C from the nuclear weapon tests for those persons born between 1900 and 1990. Further generations can expect dose values which will be below the naturally caused value. The differences between the two variants of the scenario are small compared with this decrease. For a person born in 1974 the calculated lifetime dose will be near to the pre-industrial value. About 16% of this dose will be caused by radiocarbon released from nuclear weapon tests.

4.2. Scenario II

The results of the simulation for the three variants of Scenario II (partial replacement of fossil fuels by nuclear power) are given in Figs 5 and 6. For

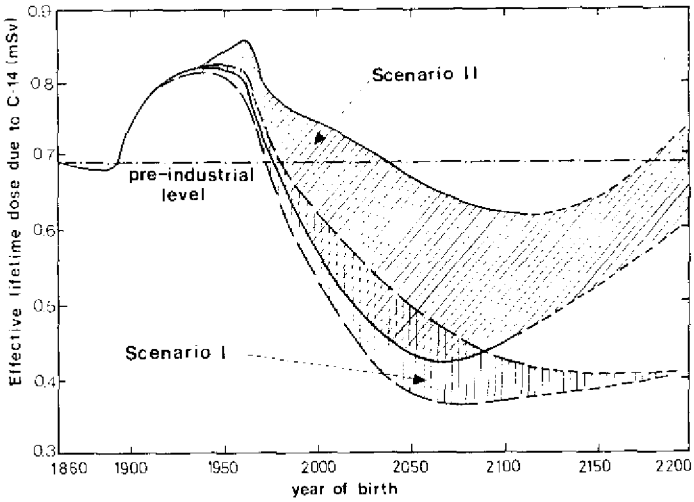

FIG.6. Ranges of the effective lifetime (70 years) doses due to C-14 as a function of the year of birth.

comparison, the pre-industrial levels are also shown in the same figures. Again, the considerable contribution of the ^{14}C from the atomic bomb tests can be seen. For the upper variant, the pre-industrial level will be reached in the middle of the 21st Century, and about 50 years earlier for the other variants. The minimum of the ^{14}C exposure for the medium variant lies at about 8 μSv per annum. The lifetime doses are highest for persons born about 1960 and lowest for those born between 2050 and 2100. After the year 2200 a second maximum can be expected.

In Table V the lifetime doses for a person born in 1974 are compared for all scenarios. Radiocarbon released by nuclear power production contributes between 3% and 6% of the total exposure. The last column contains the values for the medium variant if after the year 2020 no further ^{14}C is released from nuclear installations (i.e. assuming total retention, installation of other reactor types, or strong reduction of nuclear power production). The lifetime dose is then reduced by almost a factor of two.

5. CONCLUSIONS

Discussion of the various scenarios has shown that short- and long-term effects for the future radiation exposure by ^{14}C have to be considered. The long-term exposure is determined mainly by the amount of ^{14}C released and its residence

TABLE V. EFFECTIVE LIFETIME (70 YEARS) DOSE (mSv) FOR A PERSON BORN IN 1974 CALCULATED FOR THE VARIOUS SCENARIOS

	Scenario I		Scenario II				
	(a)	(b)	(u)	(m)		(ℓ)	(m)*
Annual growth rate in 1974 α_{74} (%)	2.0	4.0	2.0	3.4	3.4	4.0	3.4
Retention factor at fuel reprocessing	–		-	-	RF=4	-	–
All sources	0.75	0.72	0.81	0.76	0.74	0.73	0.75
Nuclear weapon tests	0.12	0.12	0.12	0.12	0.12	0.12	0.12
Nuclear power plants			0.02	0.01	0.01	0.01	0.01
Fuel reprocessing plants			0.03	0.02	0.01	0.01	0.01

* No releases after 2020.

time in the atmosphere. Until the year 2050, short-term effects due to radiocarbon released from the nuclear weapon tests are dominant. Until the year 2200 (medium-term), the exposure depends on various factors of different importance:

- The reduction or growth of the long-lived biosphere;
- The combustion of fossil fuels;
- The installation of nuclear power plants and the retention of ^{14}C, especially in reprocessing plants;
- The installation of other reactor types (e.g. fast-breeder, high-temperature reactor with higher exploitation of the nuclear fuel reserves);
- The development of new energy supply techniques (solar, geothermal,etc.).

In the time interval considered, radiation exposure by ^{14}C for the next few generations is estimated to be below the pre-industrial level.

Because of the large dimensions of potential impacts on future climatological conditions by an increasing CO_2 content of the atmosphere, these risks to future generations appear to represent more important aspects of technology assessments than the radiological effects of radiocarbon.

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THE HEALTH ASPECTS OF BIOGAS AS AN ENERGY SOURCE

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Abstract

THE HEALTH ASPECTS OF BIOGAS AS AN ENERGY SOURCE.

Data on the positive health impacts of biogas as fuel for rural household cooking have been collected from three villages near Bombay, one of which used traditional firewood as cooking fuel, one used biogas plants, and the third used biogas plants connected to latrines. The study illustrates the advantages of the use of biogas compared to wood, dung-cakes and crop residues. The biogas plants in the villages selected for study have been in operation for three or four years. Short-time studies show positive advantages of the use of biogas as fuel.

1. INTRODUCTION

Water, food and energy are the essential requirements of human beings. There is an abundance of natural resources of water in many parts of the world. Food requirements are also met by indigenous production or by imports. The sources of energy, however, particularly for the industrial and domestic sectors, are diminishing rapidly and their availability has become anybody's guess. In the context of an increasing demand for energy in the industrial and domestic sectors, several new alternative energy technologies are being practised.

For over twenty years, scientists have been studying alternative and renewable sources of energy. In developing countries in the tropics, renewable sources of energy, such as biogas, solar energy and windmills, are gaining popularity and are more or less accepted as possible sources of alternative energy, to replace conventional and commercial fuels. This paper studies the health aspects of the use of biogas.

Biogas has been in use in India for over twenty years under the name of Gobar gas, propagated by the Khadi and Village Industries Commission, an autonomous and statutory body which is responsible for rural development. Over 80 000 biogas plants have been installed to date and it is proposed to instal a further 500 000 in the next five years.

TABLE I. COUGHING ATTACKS AND EYE TROUBLE: FIREWOOD USERS AND BIOGAS USERS

	Irritation and watering of eyes						Coughing attacks					
	Members of family who cook			Non-cooking members of family			Members of family who cook			Non-cooking members of family		
	TFW[a]	BG[b]	BGL[c]	TFW	BG	BGL	TFW	BG	BGL	TFW	BG	BGL
Yes	8	–	–	11	–		8		–	11	–	–
No		5	13		9	17		5	13		9	17

[a] TFW = traditional firewood users.
[b] BG = biogas users only.
[c] BGL = biogas users with latrines connected to digesters.

TABLE II. COUGHING AND BRONCHITIS WHILE COOKING: BIOGAS USERS AND FIREWOOD USERS

Recurrent cough			Chronic bronchitis		
TFW[a]	BG[b]	BGL[c]	TFW	BG	BGL
13/19	4/14	2/26	8/19	0/14	0/26
68%		15%	44%	0%	0%

[a] TFW = traditional firewood users.
[b] BG = biogas users only.
[c] BGL = biogas users with latrines connected to digesters.

Gobar gas plants constructed throughout India can be viewed from three different points: (i) as a source of energy and fertilizer; (ii) as bringing socio-economic benefits to the family; and (iii) as being less hazardous to health and hygiene and at the same time reducing drudgery for women.

Since India is mainly agriculture-based, with 80% of its population living in rural areas, the main source of domestic fuel comes from firewood, cattle-dung cakes, and crop residues. Biogas plants are accepted by the rural farmers since firewood is becoming increasingly scarce as a result of the denudation of forests. Similarly, the use of cattle-dung as a fertilizer owing to the need for increased agricultural output has meant that cattle-dung is no longer available as fuel. This is also the case with crop residues, which can be better utilized for other purposes. The gradual change in the living habits of rural people has made it imperative to look for new sources of energy, and biogas has therefore become more popular.

Biogas technology is locally available and easily adaptable by rural people in the developing countries. In keeping with the theme of the present Symposium, this paper reports comparative studies on the use of fuels: traditional versus the biogas systems.

2. METHODOLOGY

The methodology adopted in this study deals with three categories: (a) users of firewood in the traditional way, (b) users of biogas for cooking, and (c) users of biogas with toilets connected to biogas digesters.

TABLE III. EFFECTS OF USE OF BIOGAS AND FIREWOOD ON EYESIGHT

	Distant vision			Colour vision			Near vision		
	TFW[a]	BG[b]	BGL[c]	TFW	BG	BGL	TFW	BG	BGL
Normal	15	11	20	19	14	26	17	10	21
Abnormal	4	3	6	0	0	0	2	4	5

[a] TFW = traditional firewood users.
[b] BG = biogas users only.
[c] BGL = biogas users with latrines connected to digesters.

Traditionally, firewood, dung-cakes and crop residues are used in open chullas (ovens) from which a lot of smoke emanates and heat is dissipated, resulting in low thermal-utilization efficiency of the fuel. Studies have been made on respiratory diseases due to inhalation of smoke, irritation and watering of eyes due to smoke and the resultant fatigue, as well as perspiration due to high room temperature.

This study explores the effects on the respiratory system of using biogas and the continuous saving in man-hours in terms of procurement of fuel and reduction of cooking time owing to the high thermal efficiency of biogas fuel.

In most parts of rural India, owing to lack of sanitary facilities, people prefer open-air defecation, which has several disadvantages, in particular, spoiling the superficial layer of the earth with dangerous bacteria. leading to contamination of well-water and pond-water. Consumption of this contaminated water leads to a host of abdominal diseases.

Walking barefoot on the contaminated earth leads to deadly diseases like hookworm and anaemia, in addition to perpetuating the disease known as dracunculosis. The use of latrines and disposal of excreta after proper treatment through biogas digesters has been studied where the biogas plant users have toilets and latrines attached to the biogas plants.

3. COMPARATIVE STUDY

Materials and data for this study came from three villages situated near Bombay: Dhani, Bilalpada and Vatal, representing, respectively, traditional firewood users, biogas plant users, and users of biogas plant connected to latrines.

TABLE IV. HOURS SPENT BY PARENTS ON CHILD-CARE AND HOUSEWORK: FIREWOOD USERS AND BIOGAS USERS

No. of hours spent	Firewood users	Biogas users only	Biogas users with latrines connected to digesters
0	8	0	0
1	0	0	0
1–2	0	0	0
2–3	0	5	9

A total of 59 subjects were throroughly examined as to their state of health, and relevant socio-economic information was collected from the head of each family and charted in a pro forma specially prepared for the study. Table I shows clearly that smoke, which irritates the eyes and makes them water, is found only among traditional firewood users. Biogas users are only theoretically liable to this effect where rooms are ill ventilated and where combustion of methane might lead to increased CO_2 concentration and decreased oxygen, leading to a health hazard which is so far unknown to us. This, of course, needs a long-term, detailed study.

Table II shows occurrences of coughing attacks during cooking hours, recurrent coughs and chronic bronchitis in 100% of traditional firewood users. Coughing attacks during the course of cooking are present. Recurrent respiratory infection and chronic bronchitis are present, respectively, in 68% and 50% of the cases examined. None of the biogas users had coughing attacks during cooking. Only 15% of biogas users had recurrent coughs, which is the usual proportion present in the general population.

We tried to compare distant vision, near vision and colour vision on Snellen's chart. The details are given in Table III. To our surprise, more cases of abnormal distant vision as well as near vision were found among users of biogas which probably reflects the past effect of using firewood, since the use of biogas is of recent origin -- say three or four years. Long-term follow-up studies might prove the point for the next generation.

Table IV shows man-hours spent in procuring firewood, estimated as between 500 and 1000 per year, which shows a distinct advantage to the biogas users in terms of man-hours per annum saved. An average biogas user completes cooking within one hour in the morning and one hour in the evening,

TABLE V. GENERAL STANDARD OF HEALTH OF COOKING-MEMBERS OF THE FAMILY: FIREWOOD USERS AND BIOGAS USERS

	Perspiration and fatigue			Tired at end of day		
	TFW[a]	BG[b]	BGL[c]	TFW	BG	BGL
Yes	7	1	2	8	0	0
No	1	4	11	0	5	13

[a] TFW = traditional firewood users.
[b] BG = biogas users only.
[c] BGL = biogas users with latrines connected to digesters.

while traditional firewood users require two to six hours per day depending on the size of the family.

A rough estimate of accidents that have occurred, according to the history given by heads of families, shows that among eight families using firewood in a traditional manner, minor or major injuries and burns were more commonly sustained than by users of biogas. Another advantage for biogas users is that they can cook on a raised platform, whereas firewood users do their cooking on the floor. No major accidents, such as burns, etc., were reported by either group.

Table V shows that the women who use firewood for cooking were not only exhausted after cooking but also felt tired at the end of the day. More sweating and fatigue was experienced by firewood-users than biogas users. Table VI shows the time saved by biogas users and spent in a useful manner, e.g. in taking care of their children, in general upkeep of their houses and other productive activities.

Table VII shows the recurrent diarrhoea, recurrent abdominal pains and number of motions per day in the three groups studied. Recurrent diarrhoea and abdominal pains were more common among the traditional firewood users and biogas users without latrines than in the third group. Biogas users who have connected their latrines to digesters had on an average one or two motions per day as compared to other groups who had between two and four motions per day. Although there was no apparent ill effect from this physiological process, obviously their health was below the standard of other groups.

TABLE VI. TIME SPENT ON PROCURING COOKING FUEL AND ON COOKING: FIREWOOD USERS AND BIOGAS USERS

Man-hours per year spent in procuring firewood for cooking				Man-hours per day spent in cooking			
	TFW[a]	BG[b]	BGL[c]		TFW	BG	BGL
0	–	5	13	2	0	5	9
500–750	3	–	–	2–3	5	0	0
750–1000	5	–	–	3–4	1	0	0
				4–5	1	0	0
				5–6	1	0	0

[a] TFW = traditional firewood users.
[b] BG = biogas users only.
[c] BGL = biogas users with latrines connected to digesters.

TABLE VII. ABDOMINAL DISEASES OF USERS AND NON-USERS OF TOILET FACILITY: FIREWOOD USERS AND BIOGAS USERS

Recurrent diarrhoea			Recurrent abdominal pains		
TFW[a]	BG[b]	BGL[c]	TFW	BG	BGL
10	3	4	11	5	7
25%		5%	50%		20%

No. of motions per day	TFW	BG	BGL
1–2	4	11	23
2–3	5	2	3
3–4	8	1	0

[a] TFW = traditional firewood users.
[b] BG = biogas users only.
[c] BGL = biogas users with latrines connected to digesters.

4. CONCLUSION

In conclusion, the impact on the health of users compared with non-users of biogas in the three groups studied may be summarized as follows:

(1) The use of biogas by and large has made a positive impact by improving the health of rural people and at the same time maintaining all-round hygienic and good environmental conditions.

(2) The time saved in cooking and in collecting firewood is better utilized and can lead to additional income to the family. It can be used for giving more attention to children and upkeep of the house.

(3) Connection of latrines to biogas digesters can prevent the abdominal diseases which are prevalent in rural areas of developing countries.

(4) The only negative effect of use of biogas in ill-ventilated rooms may lead to breathing disorders due to the high proportion of CO_2. This can be overcome by providing proper cross-ventilation.

(5) To quantify and identify different areas of health aspects on the use of biogas technology, it is necessary to undertake further studies which may bring to light more objective facts. These may help the policy-makers to examine all aspects of new alternative energy sources.

In the field of international co-operation, including exchange of information on the development of biogas technology, the Khadi and Village Industries Commission is engaged in training the experts required for developing countries from the Commonwealth,sponsored by the Commonwealth Science Council. The Commission is also providing technical know-how for construction of biogas plants through UNIDO, UNEP, HABITAT, UNOTC,etc. Since the health effects of the impact of new energy systems are to be further studied and results disseminated to the needy countries, the Commission can take up further work if required, provided support is forthcoming.

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Session V

EXPERIMENTAL VALIDATION OF BIOLOGICAL DAMAGE FROM ENERGY-ASSOCIATED EFFLUENTS

Chairman
H. JAMMET
France

IAEA-SM-254/107

РАДИОБИОЛОГИЧЕСКИЙ ЭКВИВАЛЕНТ 3,4-БЕНЗПИРЕНА ПО ЭФФЕКТУ КАНЦЕРОГЕНЕЗА В ЛЕГКИХ

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Запрошенный доклад

Abstract – Аннотация

THE RADIOBIOLOGICAL EQUIVALENT OF 3,4-BENZOPYRENE IN TERMS OF CARCINOGENESIS IN THE LUNGS.

The results of experimental studies on the incidence of lung cancer induced by 3,4-benzopyrene are presented. An experiment was performed on non-pedigree female white mice. The results obtained show that the dose-effect relationship is non-linear, which makes it more complicated to determine the radiobiological equivalent of benzo(a)pyrene, since the concept of a linear relationship between dose and stochastic effects is now used for estimating the effects of ionizing radiation. A comparison of the effects of benzo(a)pyrene and ionizing radiation has shown that the risk of cancer associated with the concentrations of benzo(a)pyrene in the air in many towns in industrially developed countries may be several times higher than the risk associated with the effects of ionizing radiation in the context of the standards in force.

Аннотация

РАДИОБИОЛОГИЧЕСКИЙ ЭКВИВАЛЕНТ 3,4-БЕНЗПИРЕНА ПО ЭФФЕКТУ КАНЦЕРОГЕНЕЗА В ЛЕГКИХ.

Представлены результаты экспериментальных исследований возникновения рака легких под действием 3,4-бензпирена. Эксперимент проводился на беспородных белых мышах-самках. Полученные результаты свидетельствуют о нелинейном характере зависимости "доза – эффект", что усложняет определение радиобиологического эквивалента бенз(а)пирена, т.к. для оценки последствий воздействия ионизирующей радиации в настоящее время используется концепция линейной зависимости между дозой и стохастическими эффектами. Сопоставление эффектов воздействия бенз(а)пирена и ионизирующей радиации показало, что риск раковых заболеваний, обусловленный существующими во многих городах промышленно развитых стран концентрациями бенз(а)пирена в воздухе, может быть в несколько раз большим, чем этот риск в результате воздействия ионизирующей радиации в рамках установленных нормативов.

В последние годы появились работы, в которых предприняты попытки определить радиобиологический эквивалент воздействия ряда химических загрязнителей окружающей среды, обладающих канцерогенным и мутагенным действием. Потребность в таком показателе связана с необходимостью количественной оценки и сопоставления опасности воздействия различных канцерогенных факторов окружающей среды [1-3], например, обусловленных атмосферными выбросами ядерных и других типов энергетических установок [4]. В качестве эталона канцерогенной активности обычно предлагается относительно хорошо изученное действие единицы дозы ионизирующей радиации (бэр или Зв). Наличие данных, характеризующих канцерогенную эффективность ионизирующей радиации дает возможность определить радиобиологический эквивалент канцерогенности (РЭК) для любого химического канцерогена, для которого известна зависимость "доза - эффект".

Одним из наиболее распространенных в окружающей среде и активных канцерогенов является бенз(а)пирен (БП) [5]. Вопросу о возможной роли БП в этиологии рака легких уделяется много внимания, но он не решен окончательно. Рядом исследователей показано, что при интратрахеальном введении БП вместе с большим объемом некоторых сорбентов в легких экспериментальных животных наряду с деструктивными процессами, вызванными сорбентами, развиваются опухоли, адекватные раку легких у человека [6,7]. Однако, в литературе отсутствуют данные, которые позволили бы количественно характеризовать зависимость "доза - эффект" при естественном ингаляционном поступлении БП в легкие.

В настоящей работе представлены результаты экспериментальных исследований возникновения рака легких под действием БП. Эксперимент проводился на 282-х половозрелых беспородных белых мышах-самках. Животные были разбиты на 4 группы, одна из которых служила биологическим контролем, три другие подвергались затравке сухим аэрозолем чистого БП. Концентрации в камерах поддерживались на уровнях 0,2, 6,3 и 78 мкг/м^3 (для групп животных I, II и III, соответственно). Затравка проводилась в течение 6 ч 5 раз в неделю на протяжении 3 месяцев. Активный масс-аэродинамический диаметр (АМАД) частиц составлял 5,9 мкм, чему соответствовала 9%-ная задержка частиц в легких. Наблюдения за животными велись до их естественной гибели. Условия эксперимента не вызывали грубых токсических или иных побочных эффектов, в частности, деструктивных или воспалительных процессов в легких, и не отразились на внешнем виде и поведении животных. Падеж животных в течение 200 сут после прекращения ингаляции проходил равномерно во всех группах и не превышал 10%. Причиной гибели животных в этот период явились сезонные пневмонии. У животных, павших в более поздние сроки, при патоморфологическом исследовании были обнаружены в легких низкодифференцированные раки, лимфосаркомы и аденокарциномы – злокачественные новообразования, идентичные раку легких у человека. Условия проведения и результаты эксперимента представлены в табл.1. Как и следовало ожидать, наибольший процент злокачественных опухолей легких зарегистрирован в группе III, в которой доза БП в легких составляла 9 мкг/г ткани.

Полученные данные по соотношению "доза – эффект" (под дозой понимается количество БП, задержавшееся в легких за все время затравки) свидетельствуют о том, что зависимость эта нелинейна и описывается степенной функцией:

$$Y = 16{,}4 \cdot X^{0{,}19}$$

ТАБЛИЦА I. ВОЗНИКНОВЕНИЕ ОПУХОЛЕЙ ЛЕГКИХ В ЗАВИСИМОСТИ ОТ ПОСТУПЛЕНИЯ БП С ВОЗДУХОМ

Показатели	Группы животных			
	I	II	III	IV (контрольная)
Концентрация БП в воздухе, мкг/м³	0,2	6,3	78	-
Доза БП, задержавшегося в легких, мкг/г ткани	0,025	0,8	9	
Процент животных с опухолями легких	23	27	40	14,3
Процент животных с опухолями легких за вычетом процента случаев спонтанного возникновения опухолей по данным, полученным из наблюдения контрольной группы животных	9	13	26	-

где Y – процент животных со злокачественными опухолями легких, X – доза БП в легких, мкг/г ткани.

Полученное уравнение достаточно надежно описывает соотношение "доза – эффект" в пределах использованных в работе концентраций и может быть использовано для прогнозирования выхода опухолей. Существование подобной зависимости подтверждается анализом литературных данных по индукции рака при интратрахеальном введении различных доз БП [6-8].

Следует отметить, что выход опухолей на единицу дозы БП при его ингаляционном поступлении мышам оказался несколько выше, чем было найдено при интратрахеальном введении крысам [6]. Очевидно, это является следствием, прежде всего, лучшего контакта БП с легочной тканью в случае его хронического поступления с воздухом, а также, возможно, более высокой чувствительности использованных в опыте мышей к возникновению рака легких. Нелинейность соотношения "доза – эффект" при воздействии БП несколько усложняет определение радиобиологического эквивалента этого вещества, поскольку, как известно, для оценки последствий воздействия ионизирующей радиации в настоящее время используется концепция линейной зависимости между дозой и стохастическими эффектами.

Попробуем проэкстраполировать полученные данные на человека и оценить радиобиологический эквивалент канцерогенного действия БП ($РЭК_{БП}$) на человека. Естественно, что прямая экстраполяция полученных данных с мышей на человека вряд ли правомерна. Вопрос этот сложен и до конца не разработан. Один из возможных подходов заключается в учете того обстоятельства, что в условиях эксперимента мышь живет дольше, чем в естественных. С другой стороны, часть людей умирает, не "успевая" дожить до возникновения рака. Поэтому, человек оказывается в несколько раз менее чувствительным к воздействию БП по сравнению с экспериментальными мышами.

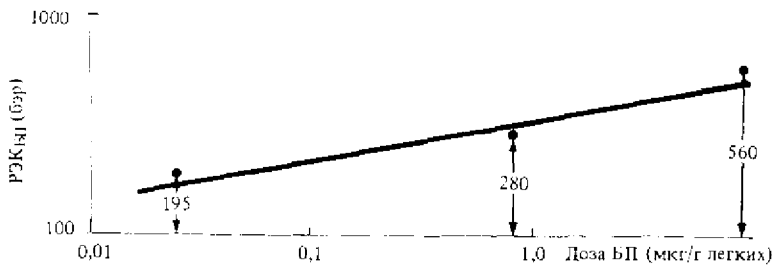

Рис.1. Радиобиологический эквивалент канцерогенности бенз(а)пирена.

Наиболее приемлемым нам представляется метод экстраполяции, учитывающий спонтанную частоту рака данного вида у конкретной группы животных и у человека. По этому показателю чувствительность контрольной группы мышей – 14,3% – оказалась примерно в 5 раз выше вероятности умереть от рака легких, характерной для населения СССР [9]. Можно предположить, что это обстоятельство отражает чувствительность мыши и человека к различным факторам, вызывающим рак легких. В таком случае, используя полученное уравнение и учитывая меньшую чувствительность человека, можно оценить выход опухолей легких у человека, постоянно работающего в условиях действующих в СССР ПДК БП в воздухе рабочих помещений (15 мкг/100 м^3). Количество дополнительных случаев рака легких составит 3,8%. Эта величина определена из условия, что человек работает 40 ч в неделю в течение 30 лет, коэффициент задержки пыли в легких 20%. При этом нагрузка БП на легкие составляет 1,4 мкг/г легкого. Ионизирующая радиация обусловила бы аналогичный риск при облучении эффективной эквивалентной дозой в 380 бэр за 30 лет, т.е. около 13 бэр/год. Эта величина в 2,5 раза выше принятого в настоящее время предела дозы для персонала – 5 бэр/год [10]. Таким образом, РЭК$_{БП}$ при дозе БП в легких 1,4 мкг/г ткани, отражающей производственные условия, равен 380 бэр. Аналогичный расчет для ПДК БП в атмосферном воздухе населенных мест (0,1 мкг/100 м^3) показывает, что за 50 лет доза БП в легочной ткани составит 0,07 мкг/г ткани, а выход опухолей легких у человека - 2,15%. Такой риск соответствует облучению организма дозой в 215 бэр. В настоящее время смертность от рака легких составляет 2,8% от числа всех умерших или 250 случаев на 10^6 человек в год [9]. Отсюда следует, что рассчитанному риску на уровне 2,15% будет соответствовать около 200 случаев на 10^6 человек в год, что эквивалентно воздействию ионизирующей радиации при дозе 2 бэр/год. Таким образом, ПДК БП в атмосферном воздухе населенных мест обусловливает риск, который в 4 раза выше действующего норматива предела дозы для отдельных групп населения 0,5 бэр/год [10]. Соотношение между дозами БП в легких и соответствующим радиобиологическим эквивалентом представлено на рис.1. Это соотношение позволяет определить дозу БП, адекватную по канцерогенному эффекту данной эквивалентной дозе ионизирующей радиации и, наоборот, выразить данную дозу БП в единицах эквивалентной дозы (по крайней мере в рамках изученного диапазона доз БП). Так, доза БП, адекватная 100 бэр (1 Зв), равна примерно 0,01 мкг/г ткани легких, а $РЭК_{БП}$ для 0,073 мкг/г ткани равен примерно 200 бэр (2 Зв).

Проведенное исследование показывает, что существующие во многих городах промышленно развитых стран концентрации БП в воздухе обуславливают в несколько раз больший риск раковых заболеваний, чем нормативы на уровни облучения ионизирующей радиацией.

Дальнейшие исследования должны позволить уточнить связь между дозой и эффектом для БП у экспериментальных животных и человека, дать возможность прогнозировать заболеваемость раком легких в зависимости от концентраций БП в воздухе и уточнить действующие нормативы на БП.

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DISCUSSION

K. SUNDARAM: Your paper is most interesting. I am not entirely sure what is meant by radiation dose, however. Does it refer to the radon dose equivalent to the lung? From the public health point of view, it might be useful to compare the B(a)P with the ambient radon concentration.

R.M. BARKHUDAROV: The ionizing radiation dose with which the effect of the B(a)P is compared here means a uniform whole-body dose or the effective equivalent dose as defined in ICRP Publication 26.

K.G. VOHRA: The concept of rad-equivalence of B(a) P is most useful for the assessment of the comparative hazards of ionizing radiations and chemicals produced by the combustion of fossil fuels. We too considered it a few years ago.

It is very important that the particle size of B(a)P be taken into account in assessing its rad equivalence. A large proportion of B(a)P particles in automobile exhausts are of respirable size, for example, and this is of concern when considering the carcinogenic effects of urban air pollution. What was the particle size used in your studies?

R.M. BARKHUDAROV: The amount of B(a)P retained in the lung does, of course, depend on particle size. However, in the equation we derived to describe the dose-effect relationship for B(a)P the main variable is not the concentration of B(a)P in the air but the amount retained in the lung – in other words, the equation does not depend on particle dispersion. The average activity median aerodynamic diameter (AMAD) of the particles in our experiment was 5.9 μm.

R. WILSON: Have you tried to relate your work to the incidence of lung cancer in humans due to the intake of B(a)P from cigarette smoking?

R.M. BARKHUDAROV: If our extrapolation method is accepted, the results of our research make it possible in principle to assess the potential carcinogenic effect on the lung from any source of B(a)P intake to that organ, including cigarettes. However, we did not perform calculations specifically for the latter.

G. ZIMMERMEYER: Your data on inhalation are very interesting because this is one of the few studies which have examined the inhalation effects of B(a)P on animals. The first study on this subject was carried out in the Federal Republic of Germany, where hamsters were exposed to even higher concentrations of B(a)P up to the mg/m^3 region), but did not show a significant increase in the incidence of lung cancer. However, these findings, which are supported by other work, would give a curve comparing ionizing radiation with chemicals with a much smaller slope than the curve you showed. If your curve is to be considered valid, therefore, the statistical significance of your data has to be given.

R.M. BARKHUDAROV: Hamsters are not a good choice for experiments on lung cancer induction. As is well known, spontaneous lung cancer is virtually non-existent in hamsters and we therefore think that there are no justifiable grounds for extrapolating experimental results to man. The low sensitivity of hamsters to the induction of lung cancer seemed to be the reason why no reliable effect was found in the experiments to which you refer. In our experiment with mice the number of animals in each group was about 70. The difference in the effect as between the experimental and control groups is statistically reliable, the level of significance being 0.05 to 0.01.

M. EL DESOUKY: What was the threshold limit value of B(a)P in the working environment? How frequently is this monitored, and what control measures are used to protect workers?

R.M. BARKHUDAROV: In the Soviet Union, the maximum permissible concentration of B(a)P in the air of industrial facilities is 0.015 $\mu g \cdot m^{-3}$. The concentration is monitored by taking air samples for filters and then measuring the B(a)P content. Protective measures consist mainly of preventing the B(a)P from entering the air. No cases have been recorded where the maximum permissible concentration has been exceeded in industrial facilities.

H. JAMMET *(Chairman)*: On what basis are B(a)P concentration limits established in the USSR?

R.M. BARKHUDAROV: These are established on the basis of experimental results which are extrapolated to man.

SENSITIVE INDICATORS IN THE DETECTION OF COAL-WORKERS' PNEUMOCONIOSIS*

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Abstract

SENSITIVE INDICATORS IN THE DETECTION OF COAL-WORKERS' PNEUMOCONIOSIS.

Twenty retired West Virginia coal-miners, who presented with respiratory symptoms or radiological changes consistent with coal-workers' pneumoconiosis (CWP), participated in a study to identify changes in pulmonary function which occur on exposure to coal dust and to develop improved methods for the early detection of CWP. They underwent a comprehensive series of studies to assess their medical, physiological and biochemical status. All the miners were life-long non-smokers with a mean age of 58.5 years and an average of 34.1 years exposure to coal dust. All had dyspnea on exertion, but physical examination showed only minor abnormalities. Chest X-rays showed varying degrees of impairment, ranging from complicated CWP to no detectable lesions. Conventional pulmonary-function tests were supplemented by measurements of the distributions of inspired gas ($\dot{V}$), pulmonary blood flow ($\dot{Q}$), and lung volume (V_A), using Kr-81m, Tc-99m macroaggregate and Xe-127, respectively, to determine regional abnormalities in lung function. A computer analysis of the regional distribution of $\dot{V}/V_A$ was performed, for which the lungs were divided into 6.5-mm × 6.5-mm pixels. The average difference between the values of $\dot{V}/V_A$ for each pixel and its four nearest neighbours was then determined. The mean overall value of these differences for all pixels in both lungs was used as an index of the degree of $\dot{V}$ non-uniformity. Pulmonary impairment was measured at the level of the small airways of the lung in all miners, even in the presence of *normal* chest X-rays. This impairment was characterized by a non-uniform distribution of ventilation (Kr-81m scans); abnormal gas exchange manifested by increased alveolar-arterial oxygen gradients; and obstruction to air flow in the small airways. In contrast, conventional spirometric measurements were within the predicted normal limits and the steady-state CO-diffusing capacity at rest was reduced in only four miners. A mean ventilation non-uniformity index of 0.179 ± 0.027 demonstrated that $\dot{V}/V_A$ for the miners was more non-uniform than that of six normal volunteers who had an index of 0.149 ± 0.003 ($P < 0.001$).

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1. INTRODUCTION

Occupational disease in coal miners is a major concern in the assessment of health effects from different energy options. With the projected increase in the use of coal in the U.S. over the next few decades, it is clear that an increasing number of miners may be exposed to coal dust and suffer from coal-workers' pneumoconiosis (CWP), unless: (1) working conditions improve markedly, or (2) early evidence of impending complications can be obtained, and preventive action taken in the management of those individuals. Unfortunately, CWP is not manifested by obvious symptoms to the affected miner in the early stages of the disease, and thus there may be no early warning. Even chest x-rays, which are still the principal basis for the diagnosis of CWP, do not show changes in the lungs until the disease is well established, or after complications such as emphysema, chronic bronchitis, or pulmonary fibrosis have occurred. Pulmonary-function screening tests routinely used to detect pulmonary disease are not usually helpful in diagnosing the early stages of CWP [1], probably because they measure functional changes primarily caused by large-airway obstruction or restrictive disease. On the other hand, changes in pulmonary gas exchange have been found in simple CWP, even in the absence of impaired ventilatory function [2]. These results indicate that regional lung abnormalities are present in early CWP which cannot be evaluated properly with studies of the overall pulmonary function. Further evidence for this is provided by a study of simple CWP using radioactive tracers, in which abnormalities in the distributions of pulmonary ventilation and perfusion were found to be the only measured functional impairment [3]. Since at least 10% of the miners ultimately develop CWP [4], the problem is a serious one and improved diagnostic screening tests are badly needed.

A joint Brookhaven National Laboratory and Marshall University investigation is currently underway to identify changes in pulmonary function which occur in miners exposed to coal dust prior to the development of symptoms, and to develop methods for the early detection of the disease.

2. METHODS

All subjects were selected for this study by a physician at Marshall University. They were retired coal miners with more than 25 years of exposure to coal dust. All were lifelong non-smokers and presented with respiratory symptoms or radiological changes consistent with CWP. They underwent a comprehensive series of studies to assess their medical, physiological and biochemical status during a five-day stay at the Hospital of the Medical Research Center at Brookhaven. These studies included a complete physical examination and medical history; laboratory analyses of blood, urine, stools and sputum; electrocardiogram; chest roentgenogram; spirometry, to measure lung volumes (vital capacity (VC), residual volume (RV), total lung capacity (TLC), and functional residual capacity (FRC)), 1-sec forced expiratory volume (FEV_1), and maximum midexpiratory flow rate (MMF); closing volume (CV/VC); peak expiratory flow rate ($\dot{V}_{max}$) and flow rates at 50% and 25% VC ($\dot{V}_{max}$ 50 and $\dot{V}_{max}$ 25) from maximal flow-volume loops; body plethysmography to measure airway resistance (RAW) and thoracic gas volume; lung compliance; gas exchange at rest and during exercise with a bicycle ergometer; steady-state CO diffusion (DLCO) at rest; and nuclear medicine studies with radioactive tracers to measure regional ventilation and perfusion.

Predicted values for lung volumes were obtained from Goldman and Becklake [5]; FRC from Boren et al. [6]; FEV_1 from Kory et al. [7]; $\dot{V}_{max}$ from Leiner et al. [8]; MMF from Morris et al. [9]; $\dot{V}_{max}$ 50% from Cherniack and Raber [10]; $\dot{V}_{max}$ 25% from Bass [11]; and CV from Susskind et al. [12].

2.1. Nuclear medicine studies

Regional pulmonary blood flow was measured with radioactive Tc-99m labeled macroaggregates (MAA) injected intravenously, while regional ventilation was measured by continuous inhalation of the very short-lived radioactive Kr-81m gas. The distribution of lung volume was measured following the inhalation and equilibration of radioactive Xe-127 gas. The 13-sec half-life of Kr-81m makes it ideal for obtaining lung views of regional ventilation during continuous tidal breathing, which can then be compared with the corresponding perfusion views. The short half-life of Kr-81m, relative to the transit time required to reach the alveoli, ensures that the images will reflect primarily the distribution of regional lung ventilation. Since airway obstruction increases the transit time, some of the gas will have decayed before reaching the alveoli distal to airways with moderate to severe obstruction. Those regions will then be represented by darker areas on the images. However, the short half-life precludes equilibration of Kr-81m in the lungs. The 36.4-d half-life of Xe-127, on the other hand, permits its equilibration in the lungs, followed by its subsequent washout. The distribution of Xe-127 at equilibrium is therefore a measure of regional lung volume. Delayed washout indicates obstructive airway disease.

The miners breathed through a mouthpiece attached to a three-way non-rebreathing valve. The nasal passages were closed by an external clamp. A steady flow of ∼25 ml/min of air flowing through the Rb/Kr generator to elute the Kr-81m was mixed with room air inspired through the inlet valve port. The expired air exited through the outlet valve port and was vented through a disposable plastic hose.

Scintiphotos of the Kr-81m ventilation images were obtained in the posterior, anterior, and right and left posterior oblique positions, and were interdigitated with Tc-99m MAA perfusion images. Scintiphotos containing 500 000 counts were acquired for optimum information density and the data stored on magnetic tape for each ventilation and perfusion image.

The Xe-127 ventilation studies were carried out with the miner rebreathing through a rubber mouthpiece attached to the inlet of the recirculating spirometry system. The nasal passages were closed by an external clamp. CO_2 was removed with soda lime and O_2 automatically introduced as required to maintain a constant volume during the equilibration phase. The miner was seated in front of the scintillation camera, with the posterior chest against the collimator face. After a suitable subject-training period, background counts were collected for 1 min, and then the radioxenon was injected as an ∿5-mCi bolus[1] directly into the mouthpiece as the miner was inhaling slowly to TLC. This first deep inspiration was held for 20-30 sec, after which the miner resumed normal breathing for 6 min. to equilibrate the concentration of radioactive xenon in the lungs and spirometer. A second 20-30 sec breath-hold at TLC was obtained just before washout, which was then carried out for 5-10 min. The gas was vented through a charcoal trap to remove the Xe-127.

Scintiphotos were obtained at TLC during the first deep inspiration and again after equilibration. Then continuous 1-min scintiphotos were obtained during tidal breathing sequentially at equilibrium and during the washout of radioxenon. Computer acquisition at 5-sec intervals was carried out during the entire study. The four Tc-99m-MAA and Kr-81m images were then compared with Xe-127 images in the posterior position obtained from the inspiration of the initial tracer bolus during a 30-sec breath-hold, after equilibration, and for the first 5 min. of washout (FIG. 1).

2.1.1. Computer analysis

The Kr-81m and Xe-127 distributions for each miner were subsequently analyzed with a CDC 7600 computer and displayed on a 64 x 64 matrix with a gray scale for comparison with the scintiphotos. The posterior Kr-81m images were normalized to

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[1] 1 curie = 3.7×10^{10} Bq.

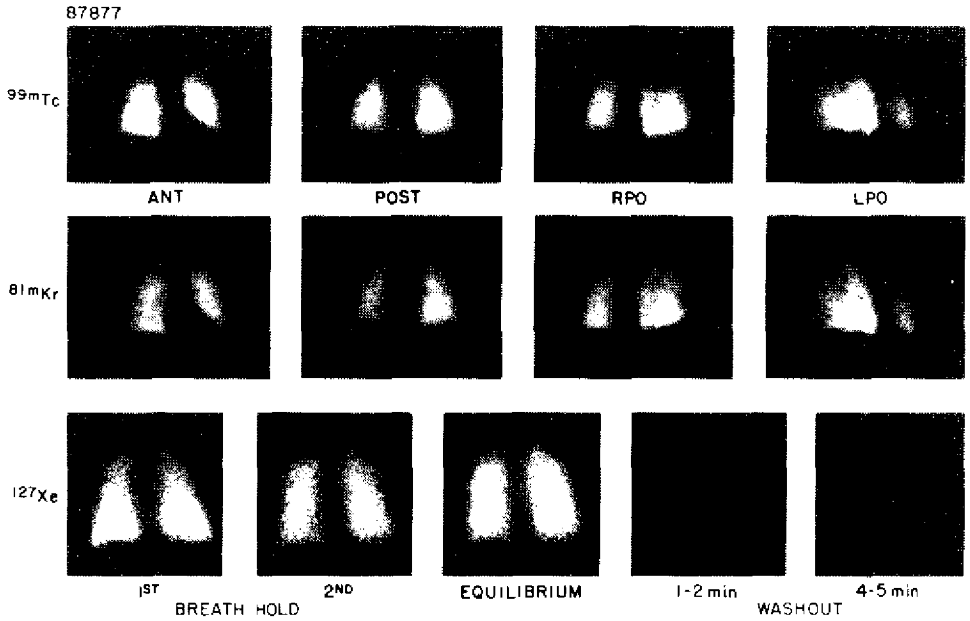

FIG.1. Scintiphotos obtained with Tc-99m, Kr-81m and Xe-127 for a 64-year-old miner with 38 years' exposure to coal dust. The Xe-127 images were obtained in the posterior position. Peripheral irregularities may be seen on the Kr-81m and Tc-99m images. Xe-127 washout was normal.

500 000 counts, as were the 1-min Xe-127 images obtained during tidal breathing at equilibrium. Each lung was divided into 6.5-mm x 6.5-mm-square areas (pixels) (FIG. 2). The point-by-point Kr-81m values were then corrected for lung volume by dividing them by the corresponding Xe-127 activities at equilibrium. The results for each pixel were thus approximately proportional to ventilation per unit lung volume ($\dot{V}/V_A$).

Since CWP is manifested by the relatively small, but widespread, distribution of focal damage, and the distribution of $\dot{V}/V_A$ was found to be irregular - both horizontally and vertically (FIG. 2), the degree of this ventilation nonuniformity was used as an index to indicate the degree of nonuniformity of ventilation. It was determined as follows (FIG. 3):

1. The value of $\dot{V}/V_A$ was determined for each pixel.

2. The average of the differences between the value of $\dot{V}/V_A$ in each pixel and that of its four nearest-neighbor pixels was determined.

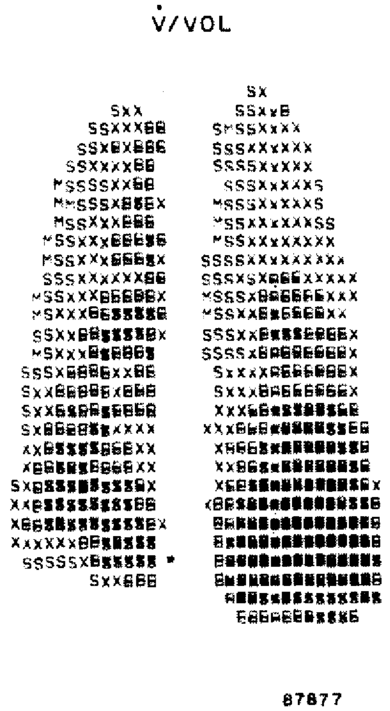

FIG.2. Pixel-by-pixel distribution of $\dot{V}/V_A$ (Kr-81m activity divided by Xe-127 activity) determined by computer analysis for the miner described in Fig.1. The horizontal and vertical distribution is non-uniform (ventilation index = 0.205).

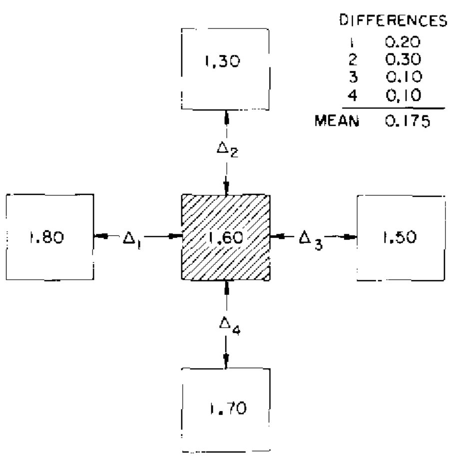

FIG.3. The mean $\dot{V}/V_A$ difference is determined between a pixel (shaded and its four nearest neighbors, as shown in the schematic. The non-uniformity index is then calculated from the overall mean for all the pixels in the lung.

TABLE I. PATIENT CHARACTERISTICS

Subject	Age (yr)	Exposure to Coal Dust (yr)	Chest X-Ray	Dyspnea*	Productive Cough
1	62	37	PMF A	2	-
2	58	39	Normal	3	++
3	54	27	q/q 0/1	5	-
4	61	30	Normal	4	-
5	62	40	q/q 1/1	3	+
6	63	40	p/p 0/1	2	-
7	64	38	Normal	2	-
8	65	32	q/q 0/1	2	+
9	54	33	PMF A	2	+
10	60	44	p/p 3/3	4	-
11	43	27	Normal	2	-
12	54	34	Normal	2	-
13	57	32	p/p 1/1	5	+
14	51	33	Normal	5	+
15	61	26	Normal	4	+
16	61	32	PMF A	4	+
17	65	40	Fine linear densities	3	++
18	65	32	Normal	5	+
19	58	35	Fibrosis	4	+
20	51	30	r/r 1/1	4	+
Mean ± s.d.	58.5 ± 5.8	34.1 ± 5.0			

*Grade 2: Short of breath walking up small hill.
3: Short of breath walking on level ground with people of same age.
4: Short of breath walking 1/4 mile on level ground in 15 min.
5: Short of breath performing minimal physical activity.

3. The overall mean of these individual adjacent-neighbor differences for all pixels was calculated and constitutes the index of nonuniformity of ventilation.

3. RESULTS

Twenty retired, non-smoking coal miners participated in this study. Their mean age was 58.5 ± 5.8 years (y) (range: 43-65 y), and they were exposed to coal dust for an average of 34.1 ± 5.0 y (range: 26-44 y) (TABLE I). Eight miners had normal chest x-rays, while the remainder showed varying degrees of

TABLE II. RESULTS OF PULMONARY FUNCTION MEASUREMENTS

	Percent of Predicted Values	
	Mean ± S.D.	Range
Vital Capacity	106.1 ± 12.6	93.6 - 118.7
Residual Volume	89.8 ± 23.8	66.0 - 113.6
Total Lung Capacity	99.3 ± 11.8	87.6 - 111.1
Functional Residual Capacity	90.9 ± 17.6	73.4 - 108.5
FEV_1	91.0 ± 11.5	79.5 - 102.5
FEV_1/FVC	95.6 ± 6.7	88.9 - 102.3
Maximal Voluntary Ventilation	96.2 ± 23.7	72.6 - 119.9
Airway Resistance	74.2 ± 21.9	52.3 - 96.0
Maximum Midexpiratory Flow Rate	74.1 ± 22.3	51.8 - 96.4
Maximal Expiratory Flow Rate at 50% VC	62.1 ± 21.0	41.0 - 83.1
Maximal Expiratory Flow Rate at 25% VC	61.5 ± 26.4	35.1 - 87.9
Closing Volume/Vital Capacity	134.0 ± 22.5	111.5 - 156.5
Static Compliance	132.8 ± 54.4	78.4 - 187.2

impairment, ranging from p-type opacities to complicated CWP (ILO 1979 classification). All patients had dyspnea on exertion, varying in severity between grades 2 and 5, as defined in the table. Only 12 of the group had a productive cough. The physical examination showed only minor pulmonary abnormalities.

Conventional measurements of VC, RV, TLC, FRC, FEV_1, FEV_1 as percent of the forced VC, MVV and RAW were all within normal limits (TABLE II). In contrast, one or more of the following measurements of MMF, $\dot{V}_{max\ 50}$, $\dot{V}_{max\ 25}$ or CV/VC was abnormal in 18 miners, indicating small airway obstruction. The static compliance of six of 12 miners tested exceeded 0.300 liters/cm H_2O, and the dynamic compliance decreased as the respiratory frequency increased to 60 breaths/min. in six of them. In normal subjects the compliance does not change with the respiratory rate.

Abnormal results were obtained in all gas-exchange studies, with increased alveolar-arterial O_2 gradients ($P(A\text{-}a)O_2$) and hyperventilation (TABLE III). Although $P(A\text{-}a)O_2$ was normal in five of the miners at rest, it became abnormal during physical exercise. In three of them the arterial CO_2 tension ($PaCO_2$) was also reduced at rest. However, steady-state DLCO was reduced in only four miners at rest. No clear trends were found from the exercise studies. In some cases, the miners' hypoxemia and hyperventilation became worse, with progressively

TABLE III. RESULTS OF GAS EXCHANGE AND VENTILATION MEASUREMENTS AT REST

Subject	PaO_2, mm Hg	$P(A-a)O_2$, mm Hg	$PaCO_2$, mm Hg	Ventilation Nonuniformity	DLCO, % Predicted
1	74	38	37	0.168	66
2	67	23	38	0.196	77
3	63	70	24	0.226	82
4	80	27	41	0.157	108
5	82	33	34	0.156	112
6	81	29	30	0.211	99
7	79	27	39	0.205	127
8	83	31	34	0.171	45
9	64	52	32	0.221	57
10	82	27	36	0.176	96
11	94	16	35	0.169	124
12	78	36	31	0.187	118
13	92	19	33	0.179	104
14	92	21	32	0.143	105
15	88	17	33	0.144	150
16	100	18	30	0.134	91
17	88	18	35	0.162	150
18	88	30	31	0.187	140
19	70	41	31	0.219	111
20	91	22	31	0.166	123
Mean±s.d.	82±10	30±13	33±4	0.179±0.027	104±29

decreasing arterial oxygen tension (PaO_2) and increasing $P(A-a)O_2$, while in other cases the opposite occurred.

The results of the ventilation and perfusion scans verified the minimally obstructive character of CWP (FIG. 1). Most of the Kr-81m ventilation images showed only peripheral irregularities. Significant segmental defects were found in only four miners. Of the 20 miners studied, only seven had minimal Xe-127 retention after 5 min. of washout, one with diffuse retention and six with retention primarily in the bases. The apical regions were underperfused, resulting primarily from the Tc-99m MAA injection while the miners were seated upright. The perfusion and ventilation images were matched in 15 cases. Ventilation but no perfusion defects were found in three miners, and perfusion but no ventilation defects in two miners.

The mean value for the ventilation indices for the 20 miners, indicating their degree of ventilation nonuniformity,

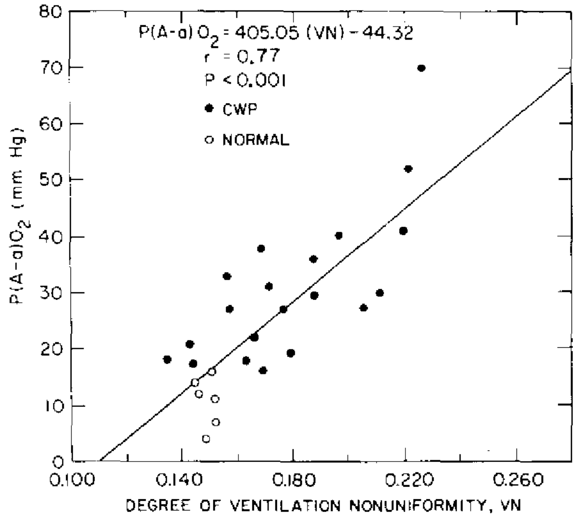

FIG.4. Correlation of alveolar-arterial oxygen gradient and the non-uniformity index for coal-miners and normals.

was 0.179 ± 0.027 (range: 0.134-0.226) (TABLE III). A similarly determined mean value obtained from six healthy volunteers, ranging in age from 31-67 y (mean of 42.7 ± 12.9 y), was 0.149 ± 0.003. The mean CWP value was statistically different from that of the normals at the level of $P < 0.001$. There was no overlap for 17 of the miners with the normal data. The other three miners will be discussed below.

The degree of nonuniformity in ventilation correlated well with such overall measures of gas exchange as $P(A-a)O_2$ and PaO_2 at rest. The $P(A-a)O_2$ for both miners and normals varied directly with the degree of ventilation nonuniformity (FIG. 4). The equation for the best-fit line through the data for the miners and normals was:

$$P(A-a)O_2 = 405.05(VN) - 44.32$$

$$r = 0.77$$

$$P < 0.001$$

where VN is the nonuniformity index.

The results for the normals are located in the lower left-hand corner of the graph, while miners with increasing

$P(A-a)O_2$ also had increased nonuniform ventilation. The three miners mentioned above, who had indices that fell into the normal range, also had normal $P(A-a)O_2$ at rest.

4. DISCUSSION

Functional pulmonary impairment, which could be quantitated, was found in all cases in a detailed examination of a group of 20 West Virginia coal miners. The miners' dyspnea and other respiratory symptoms were assumed to be caused by coal dust since they had an extensive exposure history, and no other abnormalities were found to explain them. It is important to note that all of them were life-long non-smokers, so that the effect of cigarette smoke either by itself or in synergism with coal dust could be eliminated as a potential factor influencing the results. This finding is in agreement with the study of Jacobsen and co-workers [13,14], who found significant associations between exposure to coal dust and CWP in non-smoking coal miners. On the other hand, Morgan et al. [15] and Rom et al. [16] believe that the effect of cigarette smoke is more significant than that of coal dust in causing pulmonary impairment. In addition, the miners exhibited the usual symptoms associated with CWP - significant dyspnea in all cases, productive cough in 12 men, and normal chest x-rays in eight men. Neither the grade of dyspnea nor the x-ray findings correlated well with the results of the physical examination or with the functional respiratory or gas-exchange measurements.

Impaired gas exchange was found to be the most significant functional measurement, being abnormal in all miners - 18 already at rest and the others during subsequent physical exercise. All had increased $P(A-a)O_2$ and were hyperventilating (reduced $PaCO_2$ and increased alveolar ventilation). We were able to verify the results obtained from overall blood and air sampling with a very sensitive technique using a computer analysis of the local distribution of ventilation with radioactive Kr-81m and Xe-127. Since CWP is characterized by relatively small but widespread focal damage, the degree of ventilation nonuniformity obtained from the individual pixel values throughout the lung could be related directly to their $P(A-a)O_2$. The miners with the greatest gradients also had the most nonuniform distributions of ventilation. Conversely, those with normal gas exchange at rest also had normal ventilation. Only four miners had diminished DLCO at rest in addition to the ventilation abnormality.

Our results agree with those of Robience et al. [3], who found that CWP does not result in a significant decrease in overall ventilatory function. Exposures to coal dust in underground mines ranging from 26 to 44 y produced no obstructive involvement of the large airways. The measured values for lung volumes, FEV_1, MVV, $\dot{V}_{max}$ and RAW fell within normal limits in all cases. On the other hand, we found measurable functional impairment of the small airways. In 18 of the 20 miners tested, measured values of MMF, $\dot{V}_{max}$ 50 and $\dot{V}_{max}$ 25 fell below 75% of predicted values, CV/VC exceeded the predicted mean values by two standard deviations, or dynamic compliance was frequency dependent. These findings were also confirmed by the ventilation and perfusion studies (FIG. 1). While abnormalities were found in the Kr-81m and Xe-127 images, these were generally peripheral irregularities in ventilation with minimal Xe retention during gas washout, rather than being segmental defects.

In summary, pulmonary impairment was measured in all miners at the level of the peripheral airways. These included a nonuniform distribution of ventilation as seen in Kr-81m scans; abnormal gas exchange manifested by increased $P(A-a)O_2$; and obstruction to air flow in the small airways. In contrast, conventional spirometric measurements were within the predicted normal limits, and DLCO was reduced in only four miners. Gas-exchange abnormalities correlated well and positively with the degree of nonuniformity in ventilation ($r=0.77$).

ACKNOWLEDGEMENTS

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DISCUSSION

S.R. BOZZO: Do you consider blood electrolytes to be an indication of a long period of hyperventilation?

H. SUSSKIND: I would expect this to be the case, although we have not compared the results for these subjects. However, we have measured the CO_2 partial pressure in arterial blood, which is below the normal values and which, together with an increased minute ventilation, indicates that the subjects are hyperventilating.

M. EL DESOUKY: Do you think that the technique you describe could be used as a screening test? What would be the requirement in capital and running costs for such an installation? The requirement in terms of manpower and training seems prohibitive.

H. SUSSKIND: Our studies are still in the early stages and we are attempting to identify changes in lung function as a result of CWP. In this connection it is important to point out that we were able to correlate the results obtained from overall blood and air samples with the local (point-by-point) changes in the distribution of ventilation measured with radioisotope tracers, thereby verifying that gas exchange, which was found to be abnormal in all 20 non-smoking miners, was the most significant functional measurement.

Incorporation of this approach into a screening procedure could follow later, since it could be used as part of an expanded nuclear medicine study to measure ventilation and lung perfusion. These studies are routinely used at the moment to assess a patient's pulmonary functional status, to detect pulmonary embolism, to determine whether a patient will benefit from lung surgery and so on. Many hospitals have nuclear medicine facilities including associated computers and the necessary supporting staff. I would estimate the cost of the gamma camera and computer at US $100 000 and the cost of an examination at US $500.

M. EL DESOUKY: Are the changes reported specific to coal-workers' pneumoconiosis, and what was the silica concentration of the dust to which these workers were exposed?

H. SUSSKIND: The changes we measured were caused by impairment of lung function, in this case from the inhalation of coal dust. However, we have also obtained similar results for impaired gas exchange in patients with other lung diseases. I'm afraid that I don't know the silica levels in the coal dust inhaled by these miners.

F.A. SEILER: Are you making any efforts to estimate the total exposure and, ultimately, the amount of coal dust deposited in the lungs of subjects?

H. SUSSKIND: We have no information that permits us to estimate their total exposure to coal dust other than the total number of years they worked in the mines. We are investigating possible techniques (such as whole-body activation and magnetopneumography) to measure the in-vivo lung burdens of coal dust but these require further study. We are also proposing to carry out longitudinal and cross-sectional studies of coal mines which will include accurate measurements of the concentrations and particle sizes of coal dust in the mine air.

SYNERGISM OF SO_2 AND BENZO(A)PYRENE CARCINOGENESIS
*Summary**

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Abstract

SYNERGISM OF SO_2 AND BENZO(A)PYRENE CARCINOGENESIS: SUMMARY.

The inhalation of pure polycyclic hydrocarbon by experimental animals has not yielded bronchogenic carcinoma to the extent expected. Operative and installation techniques which both damage bronchial epithelium and expose it to sustained dosage of polycyclic hydrocarbon have been productive of large numbers of tumours of the desired type. Reasoning that an inhaled chemical irritant might be an effective adjuvant to inhaled carcinogens, the exposure of animals to sulphur dioxide and benzo(a)pyrene (B(a)P) was combined. A variety of patterns or combination of these two materials was used with differences in the extent and duration of exposure to sulphur dioxide. Cancer incidents appeared to vary directly with the severity of irritant exposure when the exposure to carcinogen was held constant. The role of combined factors was further supported by a series of experiments in which intratracheal installation of carcinogen was combined with inhalation of sulphur dioxide in hamsters. Tumours induced by the inhalation of sulphur dioxide and B(a)P were squamous cell carcinomas of bronchial origin with striking invasive properties locally and occasional distant metastases. Concentrations were high (10 $mg \cdot m^{-3}$ of B(a)P and 10 ppm of sulphur dioxide). However, an incidence of 20% in a small group of 46 rats may have significant implication for larger population groups exposed to similar combinations occupationally or by way of more widespread general air pollution.

DISCUSSION

L.D. HAMILTON: I congratulate you on your clear exposition of these important observations. As they are based on small numbers of animals, I would like to have the results brought to the attention of the international community concerned with the health implications of energy generation in order to encourage the prompt repetition of your investigations with larger groups of animals. We are hoping to undertake this task at Brookhaven. Such studies are urgently needed.

* The complete text was not submitted for publication.

M. KUSCHNER: Thank you. I understand that the studies at Brookhaven you refer to envisage a determined examination of the effects of variations in the dose levels of SO_2, with exposure to the carcinogen kept at a constant level.

K.G. VOHRA: It is most interesting to see the results of the synergistic effects of B(a)P and SO_2 in experimental animals. I would like to ask a basic question regarding the relevance of animal experiments with doses of B(a)P almost a million times higher than environmental doses. There are large variations in the susceptibility of human beings to the induction of cancer by any carcinogen. The effect is essentially stochastic. How do we reconcile animal experiments at very high doses with this situation?

M. KUSCHNER: The problem of the extrapolability of high-dose animal data to long-term low-dose human exposure is one which haunts all our efforts at meaningful bioassay. I would hope that Dr. Albert's paper (IAEA-SM-254/66) in this session will help to shed some light on this very important question. I should point out, however, that while 10 $mg \cdot m^{-3}$ of B(a)P is indeed a very high dose, the 3–10 ppm of sulphur dioxide does not exceed some occupational levels by much. It may well be that the results we have reported are particularly relevant to workplace hazards.

A.P. LI: I am interested in the mechanism of SO_2 co-carcinogenesis. One model for this, based mainly on findings with skin carcinogenesis, is the two-stage initiator-promoter model. Do you plan to study the SO_2 co-carcinogenesis mechanism in this way?

M. KUSCHNER: I am not at all certain that the classic order of initiation followed by promotion would occur in the experiments we have reported. The simultaneous or concurrent exposures led us to use the less specific term 'co-carcinogenesis'. I think our original idea of simply altering the effective dose of the carcinogen by interfering with clearance was rather naive. I would not dismiss the direct genotoxic action of sulphite as a factor.

VALIDITY OF LONG-TERM AND SHORT-TERM ASSAYS FOR CARCINOGENESIS

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Abstract

VALIDITY OF LONG-TERM AND SHORT-TERM ASSAYS FOR CARCINOGENESIS.

The health hazards associated with energy systems by and large involve chronic low-level exposure to toxicants. The widespread assumption of a linear non-threshold dose-response relationship for carcinogens makes this class of agents of particular concern. In carrying out risk assessments for environmental exposures to suspect carcinogens, much reliance is necessarily placed on long-term and short-term laboratory bioassays as a basis for making both qualitative and quantitative judgements about the nature and magnitude of the carcinogenic risk. The validity of these tests as predictors of carcinogenesis in humans at low levels of exposure may depend on the extent to which the target organ is promoted either extrinsically or intrinsically.

The justification for the use of rodent bioassays for carcinogens is that, of the carcinogens known to cause cancer in humans, most cause cancer in rats and mice (1). Hence, positive results in rats and mice are taken to be evidence that the agents which produce these results are likely to be human carcinogens. Short-term bioassays, particularly DNA interaction and mutagenesis, are regarded as supportive evidence for animal bioassays and not decisive evidence in themselves (2).

Risk assessment for suspect carcinogens involves two aspects: (a) the qualitative judgment that an agent is or is not a likely human carcinogen and (b) a quantitative estimate of the amount of cancer likely to be produced by the agent given an estimate of exposure.

At the present time in the EPA and FDA, the quantitative assessments are done by use of the linear non-threshold dose-response relationship on the grounds that it has some basis in scientific fact and also that it provides a plausible upper limit for risk assessment (3).

The problem with using animal or human data to estimate risks is that the level of risks of cancer for carcinogens in the environment is far below the level which is detectable by direct observation. No dose-response model can be validated directly. In order to gain a better insight into the validity of using animal bioassays for quantitative risk assessment, we attempted to dissect the carcinogenic process by use of the two-stage initiation-promotion mouse-skin model. This model shows that the carcinogenic process can be divided into two stages: the initiation stage, which is immediate and permanent like a mutagenic event, and the promotion stage, which requires continual application of a promoting agent which is not necessarily a carcinogen.

The results of some experiments on mouse skin, presented here, show that the linear non-threshold dose-response relationship applies to the initiation stage but not necessarily to the whole carcinogenic process.

Figure 1 shows the time course of tumor formation with single graded doses of benzo(a)pyrene (B(a)P) followed by thrice weekly 5-μg applications of the promoting agent TPA (6-0-tetradecanoyl-phorbol-13-acetate). The curves show tumor formation beginning 50 days after the onset of promotion. The slopes of the curve are proportional to the dose of the initiator.

Figure 2 shows the dose-response relationship at 200 days after the beginning of TPA treatment for tumor incidence in the mouse skin according to the initiating dose of B(a)P. It can be seen that the dose-response curve is consistent with the linear non-threshold pattern.

Figure 3 shows the time course of tumor formation where the initiating dose of B(a)P was divided into various numbers of weekly fractions ranging from 1 to 64. TPA promotion, as above, was begun one week after the last initiating dose. It can be seen that there is no effect of dose fractionation.

Figure 4 shows a log-log plot of B(a)P dose with respect to the incidence of tumors. In view of the lack of a fractionation effect, the incidence is expressed as tumors per application of B(a)P with respect to the dose per application of B(a)P for both single and multiple doses of B(a)P. The slope of the line through these data points is essentially unity, indicating a linear dose-response relationship.

As shown in Figure 5, the dose-response relationship for guanine-DNA adducts formed in the mouse epidermis with single doses of B(a)P applied to the mouse skin also has a slope close to unity (4).

Although the dose-response relationship for B(a)P initiation of mouse skin has a linear non-threshold dose-response character, the dose-response relationship for the induction of tumors by B(a)P alone when applied chronically is highly non-linear. This is shown in Figure 6, which is the time course for carcinoma formation following weekly exposure to benzo(a)pyrene at the indicated doses ranging from 16 micrograms per week to 128 micrograms per week. It is clear that, with the lower dose rates, the appearance of tumors is shifted toward a later time. The observed response pattern is consistent with a dose-response relationship that is a second to third power of dose.

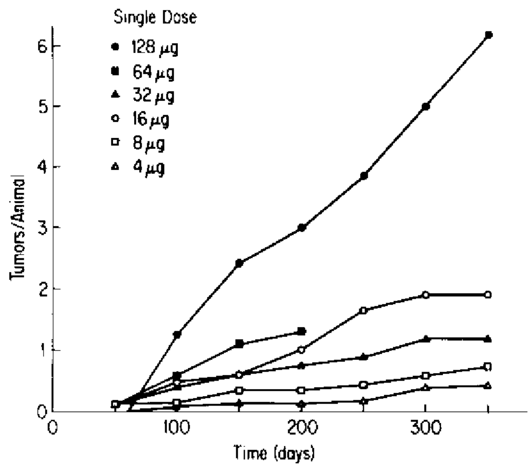

FIG.1. Cumulative tumor incidence after initiation by various single doses of B(a)P followed by thrice weekly applications of the promoting agent TPA (6-0-tetra-decanoylphorbol-13-acetate).

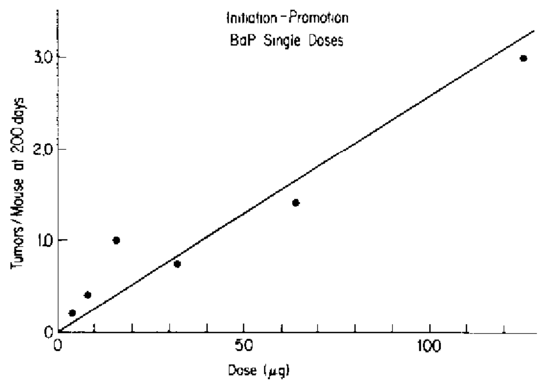

FIG.2. Tumor incidences at 200 days after the onset of promotion with respect to the initiating dose of B(a)P where the initiating doses were followed one week later by thrice weekly applications of 5 μg TPA per dose.

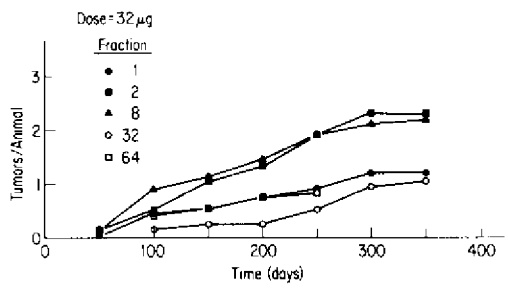

FIG.3. As in Fig.1 except that the indicated total initiating doses of B(a)P were divided into weekly fractions ranging from 1 to 64.

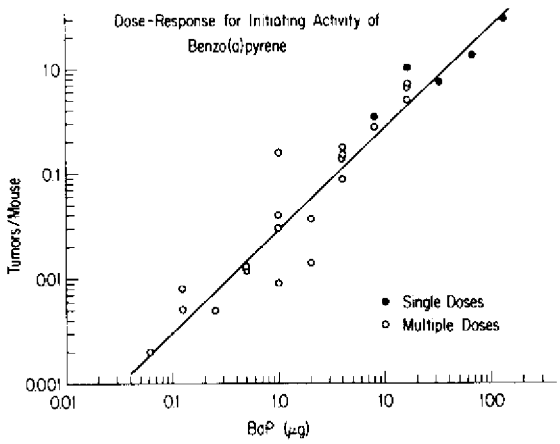

FIG.4. Tumors per mouse per application of B(a)P with respect to the dose per application of B(a)P for both single and multiple exposures to B(a)P 200 days after the onset of promotion with TPA.

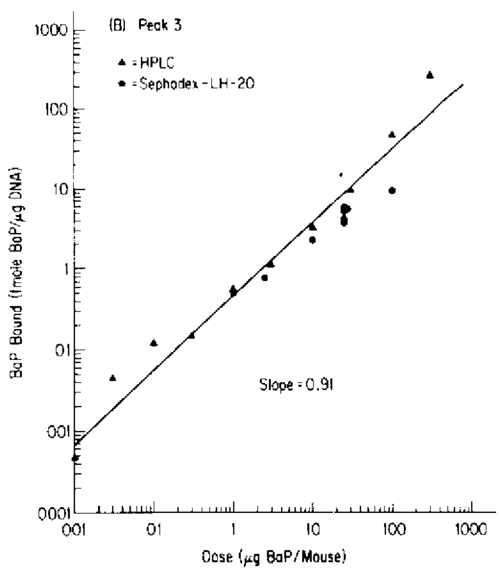

FIG.5. Dose response for the amount of B(a)P bound to epidermal DNA with respect to the dose of B(a)P applied to the skin. The data for two methods of determining the B(a)P-DNA binding are shown.

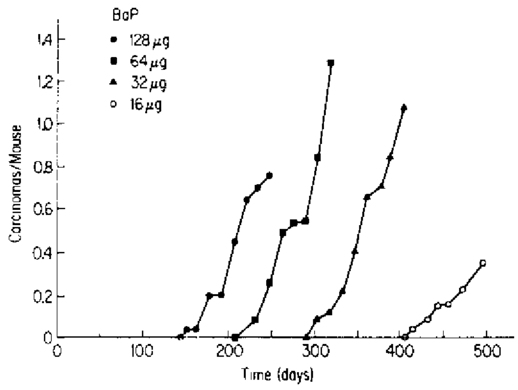

FIG. 6. Cumulative incidence of carcinomas with respect to time after the start of chronic exposure to B(a)P at the indicated weekly doses.

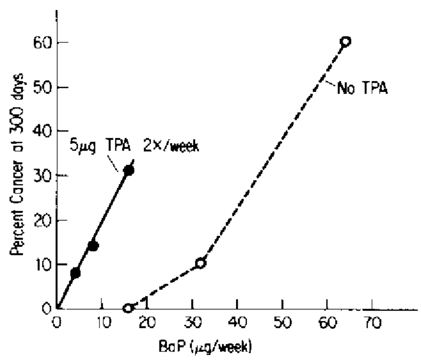

FIG. 7. Incidence of carcinomas at 300 days after the onset of B(a)P at the indicated weekly doses with and without concomitant exposure to TPA.

Figure 7 shows the tumor dose-response curves at 300 days with chronic weekly administration of B(a)P alone and with the addition of 5 micrograms of TPA given twice a week on days when the B(a)P was not given. As indicated in Figure 7, benzo(a)pyrene alone produces an upwardly curved dose-response relationship whereas the addition of the promoting agent, TPA, produces a linear non-threshold dose-response relationship.

The inferences from these results are as follows: agents which are genotoxic (DNA interactive and mutagenic) and carcinogenic in animals have a linear non-threshold dose-response relationship only in terms of their initiating action. At very low doses the promoting action of a carcinogen would tend to be negligible and only its ability to initiate would be of importance. Since initiation is not expressed except by the action of promoting agents, the only organs that are at risk of excess cancer are those which are promoted either by intrinsic or extrinsic agents; for example, intrinsic hormones in the case of endocrine organs (breast, ovary, thyroid, prostate) and extrinsic agents such as cigarette smoke in the lung, alcohol and other dietary factors in the gastrointestinal tract.

The extent to which one can generalize the findings in the mouse skin remains unknown. However, the liver, for example, shows some strong similarities in response patterns. There is a linear dose-response relationship for DNA carcinogen interaction (5). There is a linear dose-response relationship for the induction of enzyme-deficient islands in the liver which are analogous to papillomas in the skin (6). The dose-response relationship for tumor induction in the rat liver is highly non-linear as in the mouse skin (7).

If these inferences are correct, it means that positive animal bioassays will need to be extended by additional studies in order to be useful for risk assessment; for example, to identify by DNA binding in human tissue specimens which are the likely target organs and to determine, where feasible, whether agents which are known to be promoters in human organs (e.g. cigarette smoke in the lung) will actually act as promoters with the agent in question. Valuable as they are, we have been using animal bioassays as indicators of carcinogenicity in a relatively crude fashion. Emerging data suggests that the extent to which an agent is a dangerous carcinogen at low levels of exposure will be determined, amongst other factors, by the extent to which the target organ is promoted either extrinsically or intrinsically.

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DISCUSSION

I.M. TORRENS: Your data indicate a linear dose response when the initial dose is administered in fractions and non-linearity when the benzo(a)pyrene is administered chronically. What causes the difference and is there a transition between these two different techniques for carcinogenesis?

R.E. ALBERT: The data indicate linearity when the carcinogen is associated with the administration of a promoting agent regardless of whether the carcinogen is given once or repeatedly. The non-linearity is associated with the action of the carcinogen alone.

L.D. HAMILTON: Do you know of any evidence of a repair mechanism in chemical initiation?

R.E. ALBERT: No. Once initiation has taken place it appears to be irreversible.

R.M. BARKHUDAROV: What was the pathway of benzo(a)pyrene to the lung in your experiment?

R.E. ALBERT: The experiments I reported dealt only with skin.

R.M. BARKHUDAROV: In your paper you examined the two-stage development of tumours. You said that while there is no reaction threshold in the first stage, such a threshold does exist in the second stage. Could you explain this in greater detail?

R.E. ALBERT: I would speculate that skin initiation is a mutation-like single-hit process that is not expressed except by the action of a promoting agent whereas the action of the carcinogen alone is a multi-hit process which may be unrelated to initiation. I am suggesting that the initiation-promotion pathway is different from the carcinogen-alone pathway.

QUANTITATIVE COMPARISONS OF GENOTOXIC EFFECTS OF ATOMIC ENERGY AND FOSSIL-FUELLED ENERGY

Rad-equivalences for ethylene, ethylene oxide and formaldehyde — consequences for decisions at Government level*

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Abstract

QUANTITATIVE COMPARISONS OF GENOTOXIC EFFECTS OF ATOMIC ENERGY AND FOSSIL-FUELLED ENERGY: RAD-EQUIVALENCES FOR ETHYLENE, ETHYLENE OXIDE AND FORMALDEHYDE – CONSEQUENCES FOR DECISIONS AT GOVERNMENT LEVEL.

Rad-equivalences have been determined on the basis of data on the genotoxic effects of low linear energy transfer ionizing radiation and of three chemical pollutants – ethylene, ethylene oxide and formaldehyde – emitted from energy-producing power plants. In the case of ethylene and its metabolite, ethylene oxide, the conditions were particularly favourable because the equivalences could be based on the induction of total mutations in the mouse, which is the same genetic end-point used for the assessment of radiation risks. Once established, the rad-equivalences were used (a) to extrapolate the rules adopted for radiation to each of these two compounds and (b) to make recommendations for exposed workers at 'hot spots' and for the general population. Measurements of ethylene in power plants and in the atmosphere of Paris have indicated that in most cases the measured values fall within the recommended values. However, pollution by ethylene oxide in cold sterilization units should be reduced. Rad-equivalences obtained for lethal effects, and for the induction of chromosome aberrations by formaldehyde in human cells in vitro, suggest that the maximum admissible concentrations are far too high in most countries and must be reconsidered. In France, the Ministry of Health is taking the rad-equivalences into consideration for the preparation of a law regulating pollution by ethylene and ethylene oxide – as a first step. These results show that rad-equivalences can be used for risk assessments of genotoxic effects from power plants and that decisions can be made by extrapolating the rules adopted for radiation protection to some chemical mutagens, when certain strict conditions are fulfilled.

* Work financially supported by Electricité de France.

1. INTRODUCTION

In the modern world man is not only confronted with the technological problems of energy production but also with related unwanted side effects that are threatening human health and man's environment. Obviously, the use of different technologies for energy production involves, besides the advantages, certain disadvantages and risks that have to be assessed.

The aim of this paper is the quantitative evaluation of genotoxic risks to man and his environment caused by the presence of mutagenic (and carcinogenic) agents associated with the production of energy. This evaluation is achieved by undertaking a quantitative comparison of doses of certain genotoxic agents, in particular some of those produced by the combustion of fossil fuels such as ethylene, ethylene oxide and formaldehyde, with doses of ionizing radiation. In this comparison we have put special emphasis on long-term risks such as the genotoxic risks resulting from an attack of the genetic material of living organisms which lead to lethal effects, somatic and germ cell mutations as well as carcinogenic effects.

For the assessment of risks, the work established over the last five decades by the ICRP and various committees for ionizing radiation, such as UNSCEAR, BEAR, BEIR and MRC, can be taken as good examples. In spite of some inherent uncertainties and simplifications, this work has provided the necessary guidelines for radiation dose limitations which are accepted in most parts of the world (see Ref.[1] for review). In view of the impressive number of natural or man-made chemicals in man's environment, it has become necessary to carry out a similar work on risk assessments for genotoxic chemicals. The foundation of the International Commission for Protection against Environmental Mutagens and Carcinogens (ICPEMPS) in 1977 was a first step in that direction [2].

2. THE NOTION OF RAD-EQUIVALENCE

Since a sensible consensus on acceptable dose levels for ionizing radiation was reached, the idea of Crow [3] and of Bridges [4] in 1973, that the dose of a chemical agent producing a given genetic effect may be equated to the dose of ionizing radiation producing the same quantitative effect for the same genetic end-point, showed the way to a possible extrapolation of existing ICRP recommendations for radiation [5] to recommendations for chemical mutagens. This idea was developed further in recent years [6–10].

At the First Symposium on Rad-Equivalence at Orsay (France) in 1976 [9], it became clear that the evaluation between a dose of a chemical mutagen and a dose of radiation, on the basis of the same effect in one or several biological systems of reference, is only possible if some well-defined conditions are fulfilled. The unit for comparison does not have to be the rad. A given genotoxic chemical in widespread use might also serve as standard reference [10].

The following conditions (see Refs [8–11]) are prerequisites for the establishment of rad-equivalences:

(1) Rad-equivalences depend on quantitative measures, thus on the definition of comparable doses:

D (radiation dose) = ϕ (energy flux) × t (time)
D (chemical dose) = C (concentration) × t (time)[1]

(2) Reciprocity: the doses defined above have a meaning only if, when the two factors C and t vary inversely so that their product remains constant, the biological effect remains the same. The reciprocity law is verified for radiatons only in a certain domain [12, 13]. Reciprocity may equally be limited for chemicals.

(3) The shapes of the dose-effect curves should be considered. At low doses, linear relationships are often observed and a range may be found in which rad-equivalences can be meaningfully evaluated for a specific genetic end-point. This has been done in the case of ethylmethane sulphonate [14, 15] and furfuryl-furamide [16] and some epoxides [17].

(4) Rad-equivalences are only comparable for given genetic effects and experimental conditions.

3. RAD-EQUIVALENCES FOR CHEMICAL MUTAGENS

Since 1975, experimental work on rad-equivalences has been performed at the Institut Curie in Paris, with special emphasis on chemical mutagens such as formaldehyde [12, 19], ethylene and ethylene oxide [10, 20], vinyl chloride monomer [11, 21] and furocoumarins [22].

3.1. Ethylene and ethylene oxide

Ethylene is not mutagenic by itself; however, after inhalation it is metabolized in the liver to ethylene oxide (EtO) [23], an alkylating agent with mutagenic [24], carcinogenic [25, 26] and teratogenic [27, 28] properties.

Ethylene is produced by combustion: it constitutes a pollutant of the air in towns [24, 29]. Thus, as for radiation, it deals with 'hot spots' where workers may be affected, and with the diffused pollution affecting the general population.

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[1] For a more refined definition of the chemical dose see Ref. [10].

Since ethylene is a normal product of plant metabolism [30], there is a natural background level of ethylene in non-polluted air (as there is a background level of natural radiation).

The metabolic conversion of ethylene to ethylene oxide in vivo has been studied in mice [23]. In a first approximation and on average, 1% of inhaled ethylene is fixed in the organism in the form of EtO. This value is taken as a basis for the following results assuming the mouse to be a good model for humans, and assuming similar metabolic rates in the two species (which is still to be proven).

With regard to inhaled EtO, the US Food and Drug Administration, confronted with the problem of EtO pollution in cold sterilization units, arrived at the following relationships for a worker of 70 kg [31]:

1 ppm EtO $\times$ 1 hour $\rightarrow$ 2 mg of EtO absorbed

(This value depends on the breathing rate or effort of the worker and can be taken as an average value.)

Accepting the ratio of 1 per 100 for the metabolic conversion of inhaled ethylene into EtO, one can deduce

1 ppm ethylene $\times$ 1 hour $\rightarrow$ 20 μg of EtO absorbed

Following the method of Ehrenberg et al., the amount of EtO fixed can be quantitatively determined from the degree of alkylation of cysteine and histidine in the haemoglobin of erythrocytes [23].

The rate constant for the elimination of EtO from tissues is 4.6 h^{-1} in the mouse [32]. Reasonably reliable measurements of tissue doses of EtO in persons exposed to known or at least assessable average concentrations during the four-month life span (126 days) of erythrocytes used for the monitoring dose indicate that the rate constant for elimination of EtO from tissues, if it differs at all from the mouse value, could be somewhat faster [17, 33]. About the same dose was found in all organs of the body [32]. This is probably true regardless of whether the epoxide is absorbed by the organism or formed by conversion from ethylene [23].

The degree 10^{-7} of alkylation of a nucleophilic centre in DNA with the nucleophilic strength $n = 2$ corresponds to a genetic risk of 1 rad of γ-radiation [34]. Ten mrad equivalents per week would correspond to a degree of alkylation of about 10^{-5}.

The current permissible doses for radiation were set on the basis of results obtained on the total number of mutations induced in mice (for review, see Ref.[1]). Rad-equivalences for EtO, and hence for Et, were established on the same basis, with especially favourable conditions. As pointed out [34]:

(a) One can reasonably assume linearity of the dose-response curve at low doses;

(b) The mutation frequency is correlated with the number of alkylations in the nucleophilic centres in DNA, the effect being cumulative as it is for radiation;

(c) The same numbers of alkylations in DNA are associated with the mutational response to 1 rad of γ-radiation in bacteria, barley and mice (we can reasonably expect that it is also the same in Man).

In *E. coli* K12, the prophage induction versus mutation ratio for ethylene oxide approaches the value found for ionizing radiations [35].

Coming back to (a), although one can admit a linear dose-effect curve at low doses in the case of mutations induced by EtO in *E. coli*, barley and mice [17], as is the case for radiations [36], deviations from linearity can be observed with high doses [17, 37]. However, linearity is observed with the smallest doses which have been experimentally tested, and there is no evidence for a threshold.

Since the absorption of atmospheric ethylene depends on the breathing rhythm – hence on the physical activity of the individual – we propose to consider a 2.5-fold margin between light and medium activities.

According to Ehrenberg et al. [32], in the Mouse testes, and for ethylene oxide:

$$1 \text{ mM} \times 1 \text{ hour at } 37^\circ \sim 80 \text{ rads}$$

or

$$1 \text{ ppm} \times 1 \text{ hour (air dose)} \sim 40 \text{ mrads}$$

Recently, from measurements of N-3 alkylation in workers exposed to EtO, the exposure dose in ppm × hour can be reconstructed with some reliability; the tissue dose can be calculated, and the risk assessed [17, 33]. We propose for Man the values in Table I, taking into account the 2.5-fold range for breathing rhythms.

In a recent work [38], one of us (V.P.) induced mutations to thioguanine resistance in human amniotic cells in vitro by either EtO or ionizing radiation. From the linear portion of the dose-effect curves at low doses, a rad-equivalence value was thus measured:

$$4 \text{ mM} \times 1 \text{ h} \sim 65 \text{ rads}$$

or

$$1 \text{ ppm} \times 1 \text{ h} \sim 1 \text{ mrad}$$

(assuming 1 ppm = 2 μg EtO per litre of air [32]).

This result, although differing from the preceding one by an order of magnitude, can be considered as in rather good agreement if the in-vitro versus in-vivo differences in the conditions of treatment are taken into account.

TABLE I. RAD-EQUIVALENCES FOR ETHYLENE AND ETHYLENE OXIDE

		Light physical activity	Medium physical activity
EtO: 1 ppm × 1 hour	⟶	10 mrads	25 mrads
Et: 10 ppm × 1 hour or 1 mrad	⟶	1 mrad	2.5 mrads
		200 μg of EtO fixed	

TABLE II. MAXIMUM ACCEPTABLE CONCENTRATION IN THE AIR

Radiation		Light physical activity	Medium physical activity
For workers (40 h exposure per week):			
5 rads/year			
100 mrads/week	EtO	0.25 ppm	0.10 ppm
	Ethylene	25 ppm	10 ppm
		or 12 mg EtO fixed weekly	
For the total population:			
170 mrads/year	Ethylene	0.20 ppm	0.08 ppm
		or 0.5 mg EtO fixed weekly	

4. RECOMMENDATIONS FOR ETHYLENE AND ETHYLENE OXIDE

On the basis of the above rad-equivalences and the current rules adopted for ionizing radiations, the equivalences in Table II appear. Therefore, one of us (R.L.) has proposed the following recommendations to the French government advisory body, the Conseil supérieur d'hygiène publique de France:

(a) For workers subjected to a constant exposure during 40 hours per week, the average maximum acceptable concentration (MAC) of ethylene in the air should be 15 ppm. The MAC of EtO should be 0.15 ppm. For a 70-kg man working in these conditions the amount of EtO fixed in the body per week would be 12 mg or 2.5 mg/working day (5 working days/week).

(b) For the total population exposed continuously, the average MAC of ethylene should be 0.15 ppm. This corresponds to 0.07 mg of EtO fixed per day or 0.5 mg fixed per week.

When comparing these recommendations with the US Food and Drug Administration (FDA) decisions made in 1978 for the exposure to EtO of hospital workers in cold sterilization units, the acceptable dose of EtO absorbed each working day was 2.1 mg for a 70-kg man. This value corresponds remarkably well to the value of 2.5 mg recommended here. It is likely that the decisions of the FDA were reached on similar grounds, i.e. on rad-equivalences (see the report of C.W. Bruch in Ref.[3], pp 9–17).

5. FOREIGN REGULATIONS

The pollution by ethylene has not yet been regulated in any country. So far, only EtO exposures from cold sterilization units have been considered. Before the FDA published its decision, there was great confusion in the USA: concentrations of the order of 50 ppm were tolerated, i.e. concentrations 300 times higher than those recommended in France. In the Federal Republic of Germany (50 ppm of EtO) and in Sweden (10 ppm) the same 'complacent' attitude still predominated. The legislation in the USSR prescribing 0.5 ppm of EtO [39] is more cautious and almost consistent with the current radiation safety standards.

6. THE SITUATION IN FRANCE

To evaluate what the impact of the above recommendations would be, if adopted, the current status of ethylene and ethylene oxide in France has been studied on the initiative of the Ministry of Public Health.

The concentrations of ethylene were measured in 1978 in the environment of oil refineries and in Paris by Dr. M. Benarie's team (Institut national de recherche chimique appliquée, Centre de recherche, Vert-le-Petit, Département pollution des atmosphères). Using a chromatographic method, near to compressors (usually not frequented by people), average concentrations over 8 hours were between 2 and 4 ppm; within 300 m of the industrial complex, about 0.4 ppm were measured. These values turned out to be comparable to the average values measured at a point of heavy traffic in Paris (place Victor Basch) and similar to those reported for other big towns such as Berlin (West) (0.03 ppm), Los Angeles (0.02–0.12 ppm) and Delft (0.05 ppm). Within 1 km from the industrial complex, concentrations between 0.01 and 0.08 ppm were found, which are comparable to those of residential areas in towns of Europe and the USA.

The natural background in Western Europe is around 0.001 ppm ethylene. It is evident from these figures that the present values of pollution by ethylene lie in the acceptable range of the above recommendations. But the latter have the advantage of setting limits which do not yet exist.

The concentrations of EtO inside cold sterilization units and in their vicinity were measured in 1980 by Dr. B. Festy and his team. The results are now being computed. When available, they will, in particular, provide a fair assessment of the total exposure of French workers to EtO and allow an objective comparison with the total exposure to ionizing radiation.

6.1. Formaldehyde

Formaldehyde is a typical product of combustion found in our environment, e.g. in photochemical smog, in tobacco smoke, incinerator effluents, car exhausts and in the thermal degradation of epoxy thermoplastic materials (see Ref.[24] for review). It is a genotoxic product whose mutagenic activity is weak in comparison to its cytotoxic activity (see Ref.[40] for review).

Quantitative comparisons of the effects of formaldehyde with those of X-irradiation have been carried out on several biological systems and on different biological genetic end-points at the Institut Curie in Paris [8, 11, 18–20].

Several rad-equivalences were obtained for *E. coli*, yeast and mammalian cells for lethal effects [11, 19] the most interesting of which is the rad-equivalence for the lethal effect of formaldehyde on human fibroblasts in vitro [41]:

$$1\ \text{mM} \times 1\ \text{min} \sim 6.6\ \text{rads}$$

or

$$0.15\ \text{mM} \times 1\ \text{min} \sim 1\ \text{rad}$$

Recently, one of us (S.L.) obtained data concerning the induction of chromosome breakage in human fibroblasts. Using a normal fibroblast cell line (Jacot) in the 10–11 passage 1.5% chromosome aberrations were counted after either 220 rads of X-rays or 4 mM formaldehyde × 0.25 hours, giving the following equivalences:

$$4.5\ \mu\text{M} \times 1\ \text{h} \sim 1\ \text{rad}$$
$$0.27\ \text{mM} \times 1\ \text{min} \sim 1\ \text{rad}$$
$$1\ \text{mM} \times 1\ \text{min} \sim 3.66\ \text{rads}$$

When analysing the chromosomal stability after treatment of these fibroblasts in vitro (exponential phase) with increasing doses of formaldehyde or X-rays, deviations from the 2n normal chromosome set towards hypoploidy was observed.

TABLE III. MAC VALUES AND EQUIVALENCES FOR FORMALDEHYDE IN VARIOUS COUNTRIES

Country	MAC ($mg \cdot m^{-3}$)	Equivalent absorbed dose[a] ($rad \cdot a^{-1}$)
USA	4	170
Sweden	3	127
Czechoslovakia	2	85
German Democratic Republic	2	85
Federal Republic of Germany	1.2	50
USSR	0.5	21

[a] For an exposure to formaldehyde of 30 h per week for 40 weeks per year.

For this effect 600 rads of X-rays were about as effective as 8 mM × 0.25 hour formaldehyde, resulting in the following rad-equivalence:

$$2\ \text{mM} \times 1\ \text{h} \sim 600\ \text{rads}$$

or

$$3.33\ \mu\text{M} \times 1\ \text{h} \sim 1\ \text{rad}$$

or

$$0.2\ \text{mM} \times 1\ \text{min} \sim 1\ \text{rad}$$

It is satisfactory to find that the rad-equivalences on these genetic end-points in human cells approach those previously obtained for the lethal effects of formaldehyde on other human cells.

The above rad-equivalence for the lethal effect of formaldehyde on human cells may be used to evaluate some risks due to pollution by industrial plants:

(a) In a region in south-west France where natural gas is extracted, a permanent pollution of the air with formaldehyde is regularly measured. The measured values fluctuate around an average of 20 $\mu g \cdot m^{-3}$. Nobody has yet had any idea whether such a concentration should be considered harmful or not, and if so, to what extent. No accident or disease has yet been reported within the population that could be correlated with that pollution.

The above rad-equivalence tells us that, with regard to the killing of human fibroblasts (which could take place in the lungs), this pollution of formaldehyde is equivalent to an annual dose of 2500 mrads.

If only cell killing is considered, such a low dose is certainly safe and acceptable. However, if we take the equivalence on chromosome aberrations into account this dose appears to call for some caution.

(b) The local pollution in industrial plants should always remain below the MAC. Table III gives the MAC values adopted in several countries,and for each of them the corresponding equivalent dose in rads absorbed per year (cell killing) assuming that the workers are exposed 30 hours per week for 40 weeks per year. Since the values would approximately apply also for the induction of chromosome damage, the MAC values do not seem to be acceptable and should be reconsidered.

7. CONCLUSION

In conclusion, it may be said that for quantitative assessments and comparisons of risks the notion of rad-equivalence has turned out to be useful in some specific cases of chemical mutagens. These risks evaluations are, of course, far from perfect and are still open to criticism.

The rad-equivalences obtained for the three compounds described above are precise enough to elicit revisions of current decisions -- or lack of decisions – and to help governments to proceed towards the adoption of sound regulations. We are glad to point out that, thanks to the above studies and on the basis of their results, the French Ministry of Public Health has undertaken an administrative process which may lead to national regulations concerning ethylene and ethylene oxide. These would be the first regulations ever to deal with long-term effects of genotoxic chemical pollutants associated with the production of energy.

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DISCUSSION

B. SØRENSEN: I have a general remark concerning the use of analytic analogies in assessing the impacts of chemical agents and ionizing radiation. The radiation hits an arbitrary atom somewhere in the living organism and the quantal energy is almost always large enough to ensure demage. This effect is linear with dose. The crucial point is the subsequent response of the organism. In the case of a chemical agent, the factors involved are its ability to penetrate cell walls (possible threshold effect) and the immunological response of the organism. The latter is also non-linear and may lead to effects exceeding linear predictions.

It seems to me therefore that the use of the 'rad-equivalence' concept may be dangerous and that its use for legislative purposes is certainly premature.

D. AVERBECK: Your remark definitely does not apply to ethylene and ethylene oxide. I hope that I have been clear enough in stating that the evaluation and use of rad-equivalences are bound to certain strict and well-defined conditions which, in the case of ethylene and ethylene oxide, are fully met. The rad-equivalent method is thus a very valuable tool in this and other clearly defined instances.

K. SUNDARAM: I am aware of Professor Ehrenberg's work and the ethylene oxide story. Rad-equivalence has been vindicated using alkylations in testicular DNA,though there are some limitations since the technique does not separate DNA from different cell stages. Nonetheless, it has some useful attributes. I am not sure we can say the same about formaldehyde since somatic cell chromosome damage is not a surrogate for genetic mutations in the germ cells.

D. AVERBECK: Professor Ehrenberg's group has provided us with an extremely useful method for determining the dose of alkylating agents in tissues and organs. It can be used not only for ethylene oxide but also for other alkylating agents. In principle, it might even be possible to determine alkylations in the DNA of different cell stages.

With regard to formaldehyde, I agree that the ability to induce somatic chromosomal damage does not mean that the ability to induce germ cell mutations also exists.

AN ASSESSMENT OF HEALTH IMPACTS OF ELECTRICAL POWER TRANSMISSION LINES

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Abstract

AN ASSESSMENT OF HEALTH IMPACTS OF ELECTRICAL POWER TRANSMISSION LINES.

The types of investigations undertaken to test for possible biological effects of extremely low frequency electric fields have been numerous. However, neither animal and plant experimentation nor clinical studies nor experience with operating extremely high voltage transmission lines have to date provided convincing evidence of a harmful effect from exposure to electric fields associated with transmission lines in spite of numerous attempts to find such effects. Analysis of internal fields and currents supports these observations as the levels appear to be too low to affect mammalian cells. Thus, while one can never prove the negative (i.e. that there is *no* effect), the overwhelming body of evidence indicates that the electric fields associated with high-voltage lines have no deleterious biological effects.

As the demand for electricity has increased, the problem of transmitting larger amounts of electric power has been solved by building successively higher voltage transmission lines. One 345,000 volt (345 kV) transmission line (TL) can carry as much power as five 138 kV TLs. A single 765 kV TL -- the highest voltage in the U.S. -- can carry as much power as five 345 kV TLs. Test facilities exist for future TLs operating at voltages up to 2200 kV. At one end of these power delivery systems is a power generating facility, involving fossil, nuclear, geothermal, hydro, or solar procedures. Along the power delivery system are substations, where voltages are generally "stepped down" and power is dispersed through a number of smaller voltage TLs. The consumption of power occurs at the distant end of the system. In the United States the standard frequency for electric power delivery systems is 60 Hz (60 cycles per second), in Europe the standard frequency is 50 Hz.

Associated with these electric power delivery systems -- and virtually any electric device -- are man-made electric and magnetic fields. The magnitude of the electric fields

TABLE I. POSSIBLE IMPACTS OF TRANSMISSION LINES ON THE ENVIRONMENT

A. Effects of Construction and Maintenance
   1. Land clearing and maintenance within right-of-way
   2. Access road clearing and maintenance
   3. Soil compaction and other physical perturbations related to construction procedures
   4. Interference with agricultural procedures due to towers and lines
   5. Aesthetic impact of towers and lines

B. Hazard of Electric Shock from Contact with Line

C. Corona Effects
   1. Radio and television interference
   2. Noise
   3. Ozone and oxides of nitrogen production

D. Effects of Electric and Magnetic Fields
   1. Fuel ignition by spark discharge
   2. Induced electric shocks (transient and steady state)
   3. Cardiac pacemaker interference
   4. Biological effects of electric and magnetic fields

associated with these power delivery systems is a function of the voltage of the TL; the magnitude of the magnetic field is generally determined by the amount of current flowing in the TL. Both fields decrease with distance from the TL. At ground level directly under a 765 kV TL, the magnetic fields (~0.5 Gauss) are comparable to or less than those associated with normal household appliances. The electric fields at ground level beneath large TLs (e.g. 10 000 V/m) are generally much larger than those encountered in daily living. For this reason attention will be focused on the potential for these fields -- i.e. the electric fields -- to induce biological effects.

Table I lists possible impacts of alternating current transmission lines on the environment. The first set of impacts -- Effects of Construction and Maintenance -- are important to line-routing and tower-structure considerations. The second category -- Hazard of Electric Shock from Contact with Line -- is a real hazard, because of its generally lethal consequences, and is well recognized by utilities and regulatory bodies. Minimum clearances between conductors and the ground allow a safety margin for large objects to pass under the lines. The third class -- Corona Effects -- derives from the breakdown of air in the very near vicinity of the line; an occurrence most likely in foul weather. The effects of corona include noise and chemical oxidant generation, and possibly radio/TV interference. The noise levels are below those needed to cause hearing damage but may be bothersome. TL-generated oxidants appear to be too little to cause any significant biological effect. The last class of impacts -- Effects of Electric and Magnetic Fields -- includes direct biological effects, induced electric shock and effects on cardiac pacers. There is a "theoretical" possibility that a spark of sufficient energy to ignite gasoline could arise during refueling under a TL. Transmission lines are designed so that electric shocks from vehicles under TLs will not exceed 5 milliAmpères, a value taken by the Underwriter's Laboratory and the Canadian Standards Association as the maximum safe current for the public. Modern cardiac pacers are inherently sensitive to electric fields as they are dependent on sensing heart signals on the order of 1 mV. When interference is sensed, the pacer reverts to a fixed or "interference" mode of operation, providing a fixed rate signal to the heart. Interference of some pacers by exposure to maximum fields under a 765 kV TL appears possible but not life threatening. The last subheading -- Biological effects of electric and magnetic fields -- is the prime concern of this paper. Two approaches -- "biological" and "biophysical" -- have been used in arriving at a conclusion regarding the potential for TL electric fields to induce deleterious biological effects.

The types of "biological" investigations undertaken to test for possible biological effects of extremely low frequency (ELF) electric and magnetic fields and to understand the mechanisms whereby such fields induce biological perturbations have been varied and numerous. For convenience the "effects" literature will be discussed under the general headings of: 1) Working Situations, 2) Human Laboratory Exposures, 3) Animal Laboratory Exposures, and 4) Threshold Exposures.

1. Working Situations (Table II).

The results of these studies are uniformly negative in terms of relating any health or crop problems to exposure to transmission line electric and magnetic fields. The value of

TABLE II. WORKING SITUATIONS: TRANSMISSION-LINE EXPOSURE INVESTIGATIONS FOR BIOLOGICAL EFFECTS

Author(s)	Setting	Results
Kouwenhoven et al. Singewald et al. [1]	9-yr survey 345 and 765 kV transmission linemen	negative
Roberge [2]	735-kV switchyard personnel	negative
Strumza [3]	residents near, far from, 200-400 kV	negative
Krumpe et al. Houck [4]	5-yr survey Seafarer, controls	negative
Hodges et al. [5]	765-kV TL, crops	negative
Amstutz et al. [6]	TL, animals	negative
Knave, B. [7]	5-yr survey 400-kV substation workers, controls	negative
Greenberg et al. [8]	765-kV TL, bees	Behavior correlates with hive construction features
Stopps et al. [9]	High-voltage equipment and TL workers	negative
Greene [10]	765-kV TL, crop plants	negative

these studies is that they provide evidence that there are no pronounced effects by exposure to such fields.

Soviet investigators have reported a number of complaints such as listlessness, excitability, headache, drowsiness, fatigue and physiological differences attributed to exposure to high electric fields (Asanova and Rakov [11], Sazonova [12], Korobkova, et al. [13], Krivova, et al. [14] and Filippov [15]). The problem in interpreting such findings is that it is difficult to determine which factors in the working environment are responsible for the observed symptoms since all of them are found in a variety of occupational settings. Lyskov, et al. [16] has noted that for a 500-kV system there have been no problems concerning health and safety although from such a line the field intensity at ground level may reach 14 kV/m.

TABLE III. HUMAN LABORATORY EXPOSURES, INVOLVING ELECTRIC FIELDS, MAGNETIC FIELDS, ELECTRIC CURRENTS AND BIOLOGICAL ASSAYS

Author(s)	Setting	Results
Hauf, R. [17]	1-20 kV/m, 50 Hz, 3 hr: Behavior, blood	negative (some slight but within normal range)
Manntell [18]	3G, 50 Hz, 3 hr: Behavior, blood	negative
Johannson et al. [19]	30 kV/m, 50 Hz, 75 min: Behavior	negative
Beischer et al. [20]	1G, 45 Hz, 24 hr: Behavior, blood (including triglycerides)	negative
Eisemann [21]	200 μA, 50 Hz, 3 hr: Behavior, blood	negative
Tucker et al. [22]	15G, 60 Hz, 150 trials: Perception	negative
Gibson et al. [23]	1G, 45 Hz, 24 hr: Behavior	negative
Amon [24]	20 kV/m, 50 Hz, 5 hr: Physiology	negative
Rupilius [25]	3G, 20 kV/m, 50 Hz: Behavior, blood (including triglycerides)	negative

An electric field Soviet "standard" exists for personnel working in substations or transmission lines and provides for unlimited human exposure for electric fields up to 5 kV/m. It is difficult to determine the meaning or enforcement of the "standard". There appears to be some effort made to limit human exposure to the most intense fields. Lyskov, et al. [16] indicates that infrequent and nonsystematic exposure of the local population and agricultural workers can be disregarded.

2. Human Laboratory Exposure (Table III)

The few and slight (but within normal range) changes originally reported by Hauf [17] for reaction time and hematocrit determinations have not been observed in subsequent studies within his laboratory [vide 18,21,24,25]. Taken as a whole the data in this category, involving exposed/control or

TABLE IV. ANIMAL LABORATORY EXPOSURES

Author(s)	Setting	Results
Phillips et al. [26]	Rats and/or mice; 60 Hz, $\leq$100 kV/m	
pp. 47-94	Hematology (233)	1) female mice; higher w.b.c. 2) f3 mice; VPRC, hemogl.
pp. 95-109	Immunology (25)	Platelets high, decreased immune response
pp. 111-117	Pathology (24)	negative
pp. 119-137	Metabolic status, growth (42)	negative
pp. 139-145	Bone growth (20)	negative
pp. 147-171	Endocrinology (34)	negative
pp. 173-194	Cardiovascular function (29)	negative
pp. 195-227	Neurophysiology (52)	Sympathetic ganglia hyper-excitable
pp. 229-248	Reproduction, development (34)	negative
pp. 249-286	Animal behavior (15)	rats avoid >75 kV/m; prefer <75 kV/m
Kaune et al. [27]	Field perception (pigs)	threshold $\geq$30 kV/m
Grissett [28]	Growth, physiology (monkeys, 147 weeks, 76 Hz, 2G, 20 V/m) (>100)	exposed males faster growth 1st year
Smith et al. [29]	Growth (rats, 60 Hz, 25 kV/m, 30 days)	negative

TABLE IV (Cont.)

Author(s)	Setting	Results
Seto [30]	Growth (rats, 60 Hz, 20 kV/m)	negative
Knickerbocker *et al*. [31]	Growth, reproduction (mice, 60 Hz, 160 kV/m, P_1, F_1)	negative
Mathewson *et al*. [32]	Growth (rats, 45 Hz, 100 V/m, 28 days)	negative
Krueger *et al*. [33]	Growth (mice, 45-75 Hz, 100 V/m, 28 days)	negative
Grissett *et al*. [34]	Behavior, physiology (monkeys, 10-75 kV/m, 3G, 20 V/m)	negative
Graves, *et al*. [35]	Physiology, behavior (mice, 60 Hz, 50 kV/m, 6 weeks; pigeons, 21 kV/m)	Corticosterone level elevated first 5 minutes at 50 kV/m (maybe), pigeon perception
Marino, *et al*. [36]	Growth, physiology (rats, 60 Hz, 15 kV/m, 28 days)	decreased water consumption
Cerretelli, *et al*. [37]	Cardiac, physiology, growth, immunology (mice, rabbits, rats, dogs, 50 Hz, 100 kV/m, 2 months)	no effects at 10 kV/m
Marino, *et al*. [38,39]	Growth, reproduction, 3 generations (mice, 60 Hz, 3.5-15 kV/m)	increased and decreased weights, increased mortality
Marino, *et al*. [40]	Bone fracture healing (rats, 60 Hz, 5 kV/m, 14 days)	healing depressed, 5 kV/m

NOTE: The approximate number of comparisons between exposed and controlled animals is given in parentheses in Column 2.

"on/off" rite regimens, have not indicated that ELF electro-magnetic fields, including induced electric currents, comparable to those induced by transmission lines, affect humans.

3. Animal Laboratory Exposures (Table IV)

The studies listed in this Table are representative but not exhaustive. The vast number of comparisons between exposed and control organisms did not reveal an effect. The data from Phillips' laboratory are voluminous, and nearly every experiment (the approximate number of comparisons between exposed and control animals is given in brackets in Table IV) is replicated. A "confirmed effect" [p. 338 of 26] is generally one in which a statistically significant and consistent difference is observed in an experiment's replicate. The effects dealing with perception appear genuine. Humans generally detect 60 Hz field strengths of ~15 kV/m through hair vibration. It is not surprising that pigs perceive fields ≥30 kV/m or pigeons 22 kV/m. Because the data are based on dose-response relations, it is also clear that rats, if given a choice, will avoid areas with field strengths >75 kV/m. Whether or not the other "effects" in the Phillips' survey are genuine is not certain. The number of assays undertaken is extraordinarily large and the very few statistically significant differences do not establish cause-and-effect relations. Such differences may be chance events. Phillips indicates that the differences are "small and subtle" and "probably due to secondary factors" in the experiment. A large number of studies by other investigators have notdemonstrated deleterious effects on growth and development of rodents exposed to relatively high- and low-intensity 60-Hz electric fields.

Cerretelli et al. [37] report no effects on animals exposed to a 50-Hz 10-kV/m electric field. Serum chemistry changes (increased neutrophils, decreased lymphocytes) were reported for rats exposed briefly or chronically to 50-Hz 100-kV/m electric fields (the experiment was done once); such an effect was not noted by Phillips for rats or mice chronically exposed to 60-Hz 100-kV/m electric fields or in Cerretelli et al's experiments with dogs exposed to 50-Hz 25-kV/m electric fields. Graves et al. [35] report an initial "orientation" of mice to 60-Hz 50-kV/m electric fields. An elevated corticosterone level observed after a 5-minute initial exposure (the "new" environment - done only once) is not observed with analyses from chronic exposures (2-6 weeks). Grissett's [34] study of monkeys exposed to 76-Hz 20-V/m, 2.0 G electromagnetic fields is extensive in the number, variety and frequency of assessments for biological perturbation. "...[T]he most significant result is the difference in rate of weight gain between exposed and control males. The exposed males, which gained weight at a faster rate than did control males, are about 10 percent heavier. The difference in growth rate during the first year was significant at p=0.001. The divergence occurred during the first year. For the remaining period, the average difference between the groups has been stable." One hypothesis put forward by Grissett to explain the "effect" is that the monkeys' testes "were in contact with the cage-bottom bars generating the electric field and thus were directly stimulated to increase the secretion rate of testosterone." The monkeys were exposed in cages with 15 parallel steel bar electrodes as a floor. A potential of 0.76 V existed between each bar. "Postural observations confirm that the animals spent considerable time sitting on the bars in such a way that the scrotum was in direct contact with the bars."

There is little in the above phenomena which suggests a deleterious effect from exposure to electric fields. If such an effect is established then it will be necessary to determine a dose-response relationship and the mechanism, at least at a rudimentary level, causing the effect. Without such information it will be very difficult to extrapolate from such phenomenological observations to considerations of human health and safety.

4. Threshold Experiments (Table V).

The value of dose-response studies is that a measure of the magnitude of the electric field needed to perturb the organism is obtained. The highest field strength not producing an effect is considered a threshold exposure level. All of the studies in this category [except 47] have one thing in common: they were conducted with electrodes in the conducting medium which contained the organism (Table V). With electrodes in the conducting medium, very high current densities and electric fields in the extracellular fluid surrounding the cells can be obtained. Biological effects are observed to occur at current densities of the order of 1 mA/cm^2. A further commonality among these reports is that the field strengths and current densities used in these studies are orders of magnitude (factors of 10) greater than can be achieved by exposure to transmission-line electric fields. For example, the electric field strength in the medium necessary to produce effects on pea roots [43] or flatworms [41] could not be obtained if air were a series part of the path between the source and the organisms.

Leaf tip damage occurs in plants exposed in air to relatively high electric fields as a result of extreme perturbation of the field in the immediate vicinity of a tall

TABLE V. THRESHOLD LEVELS OF ELECTRIC FIELDS OR CURRENT DENSITIES FOR INDUCING BIOLOGICAL EFFECTS

Author(s)	Setting	Results
Marsh [41]	60 Hz, flatworm, bipolar regeneration	threshold 310 V/m, 0.6-0.8 mA/cm^2
Riesen, *et al.* [42]	60 Hz, brain organelle function	perturbation 155 V/m, 1.8 mA/cm^2 not at 0.07 mA/cm^2
Miller, *et al.* [43]	60 Hz, root growth	perturb. threshold ~300 V/m, 2 mA/cm^2
Friend, *et al.* [44]	1-100 Hz, *Amoeba* cellular alteration	threshold 300 V/m, ~1 mA/cm^2
Coate, *et al.* [45]	45-75 Hz, 1 and 2G, bacterium mutation	negative at 20 V/m, 0.5 mA/cm^2
Straub, *et al.* [46]	25-7500 Hz, marine organisms	perception at 1 mA/cm^2
Johnson, *et al.* [47]	60 Hz, leaf tip damage	inception at ~20 kV/m for pointed leaves

plant's tallest pointed leaves. The "burned" area is limited to the tallest leaves, is usually only a few millimeters long, and occurs with the onset of leaf-tip corona. The corona develops from an extreme concentration of the electric field at the very tip of the leaf. "Fleshy, rounded or blunt plant parts showed no damage in 50-kV/m [60-Hz] fields, whereas plant parts with a pointed shape exhibit minor damage at field intensities of 20 to 22 kV/m [47]."

When an object such as a human body is placed in an external 60-Hz electric field in air, the body is "coupled" to the field so that an electric field exists inside the object. In general, for biological tissues, the internal electric field will be many orders of magnitude smaller than the external field. The electric field at any point within the body is related to the current density (current per unit area) by the

$$\text{electric field (V/m)} = \frac{\text{Current density } (A/m^2)}{\text{Conductivity (mho/m)}}$$

equation in which soft tissue conductivity is 0.2 mho/m [48]. Current densities have been determined for a well grounded person in a 10-kV/m 60-Hz electric field and found to range from 0.003 mA/cm^2 (ankles) to 0.0006 mA/cm^2 (neck) [49], which translate to electric fields of about 0.1 and 0.01 V/m in the ankles and torso, respectively. Do these very small electric fields affect cells?

The system most sensitive to electric fields appears to be the nervous system, where there are excitable membranes. Internal field strengths of approximately 10-100 V/m are needed to affect firing. Field strengths of this magnitude also appear necessary to affect somatic cells. Such fields translate to current densities on the order of 0.1-1 mA/cm^2. This is approximately the current density at which one can perceive an electric current [48].

There is, then, a "safety factor" of about 100 between the electric field in the torso caused by an external field of 10 kV/m and the field which would be expected to cause biological effects and a "safety factor" of about 10 above the highest internal electric fields resulting from the 10 kV/m external fields. Internal fields for a 5-kV/m electric field exposure would be correspondingly less.

The above discussion is based on the magnitude of the electric field needed to cause nerves to fire. There is, however, some evidence that electric signals 1 or 2 orders of magnitude smaller than those needed to stimulate firing may exercise some influence on nerve cells [50,51]. However, the electrical field which would be produced in the torso by a 10-kV/m electric field in air would be only about 0.01 V/m [52]; or about 0.005 V/m for a 5-kV/m electric field. Thus, knowledge of the magnitudes of induced electric fields in bodies exposed to air electric fields plus an awareness of the fields requisite for cellular perturbation suggest that it is quite unlikely that the internal electric fields resulting from the electric fields under transmission lines will cause any significant biological effects. This suggestion is supported by the results of a wide variety of biological studies which tested for induced biological effects at electric-field levels comparable to or greater than those found under transmission lines (Tables II, III, IV, V). The results of these studies are internally consistent with the conclusion that, in general, transmission-line electric fields have not been shown to induce adverse biological effects. A similar conclusion has recently been published by the U.S. Environmental Protection Agency [53].

If indeed there are risks associated with exposure to transmission-line electric fields, the risks appear to be extraordinarily subtle. Also, from a mechanistic point of view (i.e. at least a rudimentary understanding of how electric fields interact with cells), it is difficult to see how such low internal electric fields can perturb cells, especially when there is very convincing evidence that cellular threshold levels for electric-field-induced effects are several orders of magnitude greater than those induced in a person under a transmission line.

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POSTER PRESENTATIONS

IAEA-SM-254/7

LEUKAEMIA MORTALITY IN BAVARIA CONSIDERED AS A POSSIBLE CONSEQUENCE OF RADIATION EMITTED BY NATURAL AND ARTIFICIAL SOURCES AND RELATED TO SOCIAL AND ECONOMIC FACTORS

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In recent years there have been many claims in the Federal Republic of Germany that the risk of contracting leukaemia in the neighbourhood of nuclear power installations is greater than elsewhere. The Institute of Radiation Hygiene of the Federal Health Office has therefore conducted a study of the distribution of cases of leukaemia deaths in Bavaria, so that these claims can be fairly answered. This was achieved by examining the mortality rates for leukaemia over the years 1970 to 1978 in varying districts and subdividing the population into classes for sex, age and cause of death on the basis of the four-figure ICD code of 1968.

The death rate for leukaemia in Bavaria as a whole has increased significantly, but this can be attributed to the increased mortality rate in the over-60-year-old group (Fig. 1). However, an increase in the death rate during the last ten years for leukaemia in children cannot be shown.

To test for a possible relationship between radiation levels and leukaemia incidence, regions with high natural radioactivity were compared with those having less. To do this, two groups of localities were isolated, where the difference in gonad dose between the two groups from natural radiation sources was 40 mrem per year. Table I gives the Standardized Mortality Ratios (SMR) for the period 1970 to 1978 for the group of localities with higher natural radiation levels. On the basis of these data, there is no substantiation for the theory that there has been an increase in the leukaemia death rate.

The leukemia death rate in districts with nuclear installations was compared with the average for Bavaria. In Table II the SMR for each district over the entire time span of 1972 to 1978 is given. No significant difference can be observed.

In addition, the districts were sorted with respect to the number of workers employed in industry. No significant difference in the results for districts with high levels of industrial employment and the average could be substantiated.

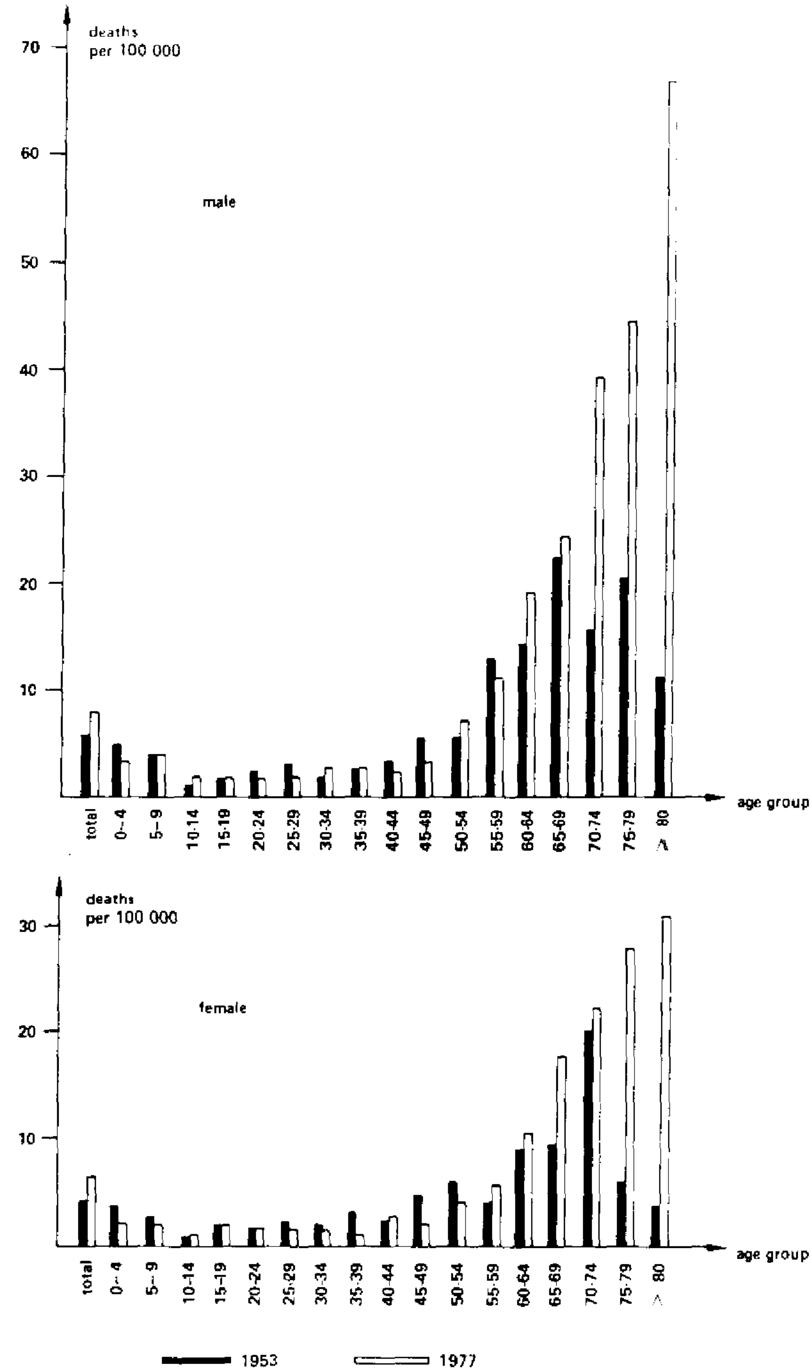

FIG. 1. Leukaemia mortality rates in Bavaria 1953 and 1977.

TABLE I. STANDARDIZED MORTALITY RATIOS FOR LOCALITIES WITH HIGHER NATURAL RADIATION LEVELS IN BAVARIA 1970 – 1978

Years of age	1970	1971	1972	1973	1974	1975	1976	1977	1978
0 - 14	1.37	0.64	1.07	0.41	0.68	1.16	1.76	1.74	0.84
15 - 59	0.55	1.09	1.21	1.42	1.23	1.31	2.17	0.55	0.81
60	1.02	1.12	1.11	1.35	0.81	0.87	1.42[a]	1.09	1.11

[a]SMR significantly different from 1 ($p < 0.05$).

TABLE II. STANDARDIZED MORTALITY RATIOS FOR DISTRICTS WITH NUCLEAR INSTALLATIONS IN BAVARIA 1972 – 1978

Years of age	Munich	Landshut	Aschaffenburg	Dillingen	Günzburg
0 - 14	0.65	0.47	0.00	1.55	1.39
15 - 59	1.36	1.49	1.08	0.62	0.92
60	1.13	0.77	1.49	1.00	0.84

IAEA-SM-254/83

HUMAN HEALTH AND THE ENVIRONMENT IN ELECTRICAL ENERGY UTILITIES

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The full text of the paper describes the experience acquired by FURNAS in health-related activities which contribute directly or indirectly to the preservation of the environment.

The paper was prepared by the Health Department of FURNAS, a company maintained by the Brazilian Government and which is responsible for the production and transmission of electric power in the south-east and part of the east-central region of Brazil, representing an operational area covering 1300 000 km^2. The company has 44 establishments (power plants, substations, offices and others) in seven states of the Federation. The Company's installed capacity for 1981 is 8.092 MW; the forecast for the next seven years, when all the plants of Angra's nuclear power station will have started working, is 10.584 MW. The Health Department operates in this extensive area, and has 18 working fronts of its own, complemented by a large number of institutions and professionals, authorized and contracted, established in 40 cities.

The material may be of use to other companies in the electric sector and to other institutions, including governmental, in formulating health policies, bearing in mind the development of measures leading to the protection of the environment, with special reference to Man as the main element to be benefited.

It is essential that health services in modern society be integrated into other areas of human activity. Such services must be conducted according to appropriate legislation and technical norms.

In large enterprises, particularly in areas with poor resources and serious health problems, due to precarious hygienic and security conditions to which large masses of workers are exposed, the responsibilities of the Company take many different forms. In such enterprises there is a very high rate of illness and accidents, including fatal ones. In the electrical utilities, which are responsible for large-scale construction and operation of the system for generating, transmitting and distributing power over a very large area, health activities must be effected by a special department responsible for planning and control.

For medical assistance at a nuclear plant, special rooms and installations for treatment in case of accidents involving ionizing radiation are necessary, as well as standardized equipment for radiation hygiene. For this purpose, plans for radiation emergency treatment and training and recycling of personnel are required.

The complete text demonstrates the intimate connection between health and environment and indicates the health activities that should be set up and developed through permanent programmes in all phases of every enterprise in the electric sector. It shows the need for active participation by the Health Department in the adoption of measures which are indispensable to improve local health conditions and to protect the environment. It is emphasized in the paper that such activities must be part of a systematic health concept. The Company's Health Department must work in co-operation with the local public and private medical institutions and hospitals. The paper also emphasizes the relationship between all health activities: medical and hospital assistance (secondary prevention), rehabilitation (tertiary prevention), and essentially preventive activities (primary prevention), for a better quality of life and the maintenance of adequate ecological conditions.

Guidelines are given for recommendations on the formulation and execution of programmes related to ecology and health, which appear to be particularly useful for developing countries.

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IAEA-SM-254/28

MOLECULAR ASPECTS OF CADMIUM DETOXIFICATION IN MAMMALIAN CELLS

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Cadmium is a ubiquitous, toxic pollutant that accumulates in the environment and in man, and it is expected that it will be mobilized with increasing energy production. We have approached the problem of cadmium (Cd^{++}) cytotoxicity by comparing the metabolism of Cd^{++} in cadmium-resistant and cadmium-sensitive cells. Three variant cell lines, derived from the parental Chinese hamster cell (line CHO), survive Cd^{++} concentrations 10–200-fold higher than that which kills the parental cell. Cd^{++} treatment of the variants induces the synthesis of a highly abundant poly A^+ RNA class which, in a cell free translation system, directs the synthesis of four low molecular weight proteins, including metallothionein (MT), not found in Cd^{++}-treated CHO cells. Nucleic acid hybridization was used to examine the information content and regulation of these inducible, highly abundant RNA sequences.

The results showed that:

(1) This class has a kinetic complexity of ~2000 nucleotides (NT) which is sufficient to encode five mRNAs the size of MT (400 NT);

(2) Cd^{++}-induction of each of the resistant variants increased the concentration of these sequences ~2000-fold above the pre-induction levels;

(3) Cd^{++} treatment of sensitive CHO cells induced the synthesis of only a subset of these sequences and then to a level <1% of that in the induced resistant variants;

(4) The genes encoding the induction-specific RNAs are differentially amplified in the genomes of the resistant variants, but the degree of gene amplification does not correlate with either the ability to synthesize the RNAs or the degree of resistance to Cd^{++}.

The data suggest that only a small number of genes are activated in response to heavy metals and that these genes are both differentially regulated and differentially amplified in sensitive and resistant cells. It is tempting to speculate that the failure of the sensitive CHO cell to synthesize all the RNA species induced in resistant cells may account for their extreme sensitivity. Although metallothionein, which presumably sequesters cellular Cd^{++} in a physiologically inert form, very likely plays a role in ameliorating Cd^{++} toxicity, other factors may also be involved. Detoxification may be more complicated than previously thought.

IAEA-SM-254/32

COAL LIQUEFACTION – HOW DOES IT MEASURE UP?

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Over the last few years, much research has been devoted to trying to determine the impacts of various synfuel technologies on human health and the environment. One group of synfuels being evaluated for commercialization is coal-derived liquids. Several processes are receiving increased attention: H-Coal, Exxon Donor Solvent, SRC I and SRC II. An evaluation of the research performed to date on the products, byproducts, effluents and emissions from these processes has shown a number of interesting conclusions concerning their relative toxicities and hazard potentials: of major concern is the mutagenicity found in liquid fractions thought to be attributed to aromatic amines.

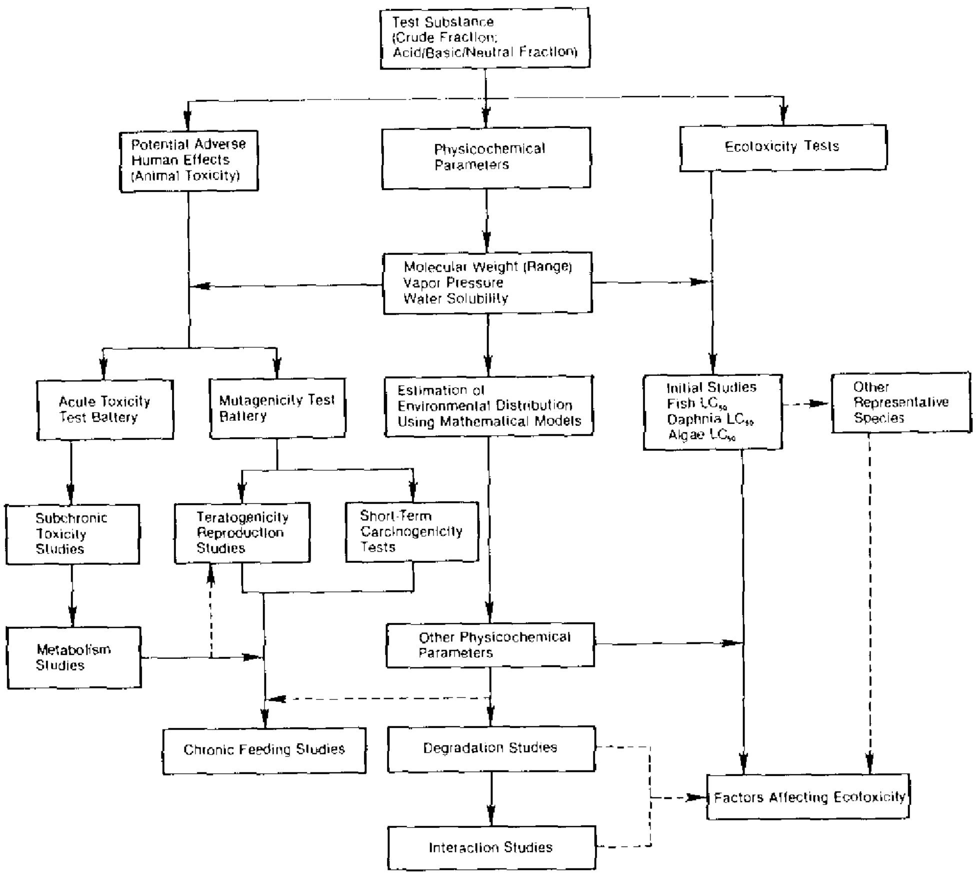

FIG. 1. Testing scheme for PMNs proposed.

It is apparent that the US Environmental Protection Agency (EPA) plans to consider most synfuel products, byproducts and isolatable intermediates as 'new chemicals' under the Toxic Substances Control Act. This means that, for each chemical or mixture, a PMN[1] will have to be submitted and accepted before commercial production can begin. Owing to the potential for large-scale production, the broad exposure, and some of the identified adverse toxic effects, the EPA is more concerned about synfuels than most other new chemicals. An increased burden will therefore be placed upon the industry submitting the PMN to ensure that the synfuels do not represent 'unreasonable risks to human health or the environment'.

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[1] Premanufacturing Notification.

TABLE I. TOXICITY-TESTING RESULTS FOR KEY COAL LIQUEFACTION STREAMS

	Ecology Tests				Reproduction Tests				Mutagen/ Carcinogen Tests			Subchronic Tests		Acute Tests			
	Decreased barley growth (TOC)[6]	Freshwater algae inhibition	Freshwater daphnids LC_{50} ppm	Freshwater fish LC_{50} ppm	Fetal Abnormalities	Peri- and postnatal Survival	Reproductive success	Decreased fertility	Skin painting	Chromosomal SHE[4]	Point mutation Ames	14 or 28 day (g/kg)	5 day (g/kg)	RAM[3]	CHO[2]	*In vitro* VERO[1]	*In vivo* LD_{50} (g/kg)
SRC II (Ft. Lewis) Raw Naptha (light distillate)					—	+	+	—	—		—		1			—	2.3
Middle Distillate		6.4[5]	2.0[5]	8.3[5]	—	+	+	—			—		1.2			—	3
Heavy Distillate					+	+	+	+	+	+	+		1.5			+	3.8
Recycle Process H_2O		+	+	+							+				+		
Wastewater Plant Influent		+	+	+							—				+		
Wastewater Plant Effluent		+	+	+							—				—		
Vacuum Bottoms		—	—	—							+			+			
SRC I (Ft. Lewis) Light Oil											—		2.4				3.0
Process Solvent										+	+					+	
SRC I (Wilsonville) Solid Product	+										+						
Process Solvent											+		1.0				2.8
H-Coal (PDU) blended distillate		+									+						
Severely hydrotreated blended distillate		+									—						
Crude Petroleums Prudhoe Bay											+				—		
Wilmington											+				—		

[1]Monkey Kidney Cell Cytotoxicity
[2]Chinese Hamster Ovary Cytotoxicity
[3]Rabbit Alveolar Microphage Cytotoxicity
[4]Syrian Hamster Embryo Transformation
[5]A blend of 2.9 medium distillate and 1 part heavy distillate
[6]TOC total organic carbon

To help the US Department of Energy (DOE) and industry prepare for the PMN process, Enviro Control has developed a testing plan (Fig. 1) that will generate the required data. We have also compared existing testing against the testing plan to show where additional work is required (Table I). Most of the additional work is required in the areas of whole-animal testing since existing efforts have focused on in-vitro screening studies. Work is needed on all aspects of synfuel combustion products. It is also essential that any PMN should present the hazards from exposure to synfuel-derived materials compared with those from petroleum or some other standard whose risks are accepted. All effects data should therefore be presented as relative effects per dose.

IAEA-SM-254/36

FOETAL AND INFANT MORTALITY AND CONGENITAL HYPOTHYROIDISM AROUND THREE MILE ISLAND

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Following the Three Mile Island (TMI) nuclear accident of 28 March 1979, we have evaluated possible health impacts of the low-level ionizing radiation and severe psychological distress upon human reproductive process, pregnancy outcome and infant mortality in the vicinity of the damaged nuclear reactor. Computed mortality rates were compared between the 10-mile TMI area and Pennsvlvania as a whole, while before-and-after observations were made within each of the 10-mile communities and Pennsylvania as a whole. In general, *foetal mortality rate* was lower for the TMI area than for the State as a whole in 1979; a similar cross-sectional difference was shown in 1978 and 1977. These differences were consistent throughout four quarters of each year under study. *Infant mortality rate* was slightly higher for the TMI area in 1979 as compared with the State as a whole. However, the higher rate was already manifested before the TMI accident and remained higher in the second quarter of 1979. There were no such differences during the latter half of 1979. There was no indication of a rising infant mortality trend during the latter part of 1979, which would have been expected if the TMI accident had a significant impact. The incidence of *congenital hypothyroidism* within the 10-mile radius of TMI was approximately one per 4000 live births during a period of one year following the accident. This is well within the range of normal expectation.

From the results of these epidemiological analyses, it may be concluded that the TMI nuclear accident did not have a significant impact upon foetal or infant mortality or upon the incidence of congenital hypothyroidism. These conclusions are also consistent with the observation that the maximum radiation exposure in the 10-mile radius during the 10-day crisis was no more than 85 mrem per person, which is lower than the annual exposure to the natural background radiation in the area.

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IAEA-SM-254/52

EVALUATION DES RISQUES MUTAGENES ENGENDRES PAR LES DIFFERENTES FORMES DE PRODUCTION D'ENERGIE

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Les différentes formes d'énergie utilisées dans notre société industrielle dégagent à des niveaux variés des émissions de polluants divers [1].

Pour circonscrire ces polluants, il est peut-être nécessaire de les identifier et de les mesurer mais il faut évaluer les détriments qu'ils causent, c'est-à-dire soit des dommages sanitaires, soit des dommages écologiques.

Pour apprécier les dommages sanitaires causés par les diverses formes d'énergie, nous avons choisi de déterminer l'effet global de ces nuisances par un test biologique; on peut ainsi approcher les effets éventuels d'inhibition ou de potentialisation des nuisances entre elles.

Dans ce travail nous avons en particulier tenté d'évaluer le potentiel mutagène des poussières émises par une centrale thermique à charbon.

1. ECHANTILLON ETUDIE

Pour l'étude d'une centrale thermique à charbon, nous avons recueilli des poussières dans l'environnement de la centrale. L'échantillon «Atmosphère» a été recueilli par aspiration d'air à travers un filtre Whatman en fibre de verre (GF/C) de perméance $< 0{,}02\%$ (ϕ 1,2 μm) avec un débit de 50 m^3/h durant 30 jours. Les points de prélèvement étaient placés sous les vents dominants de la cheminée et aux points de retombées maximum. L'échantillon «Végétaux» a été obtenu par lavage de végétaux prélevés aux mêmes emplacements que les échantillons d'atmosphère.

2. DETERMINATION DU POTENTIEL MUTAGENE

La caractéristique d'un produit mutagène est de provoquer des perturbations de l'ADN pouvant entraîner une évolution vers un processus cancérogène.

TABLEAU I. POUSSIERES DE VEGETAUX. RESULTATS

	SOUCHE			*Pouvoir Mutagène*		*Vitesse Pouvoir mutagène revers/mg*
	TA 98	*TA 1538*	*TA 100*	*Direct*	*indirect*	
Phase Hydrophyle	+	+	+	+		*4,8 TA 100 zone 16-10^3 160 µg*
Phase Aromatique	+	+		+		*1 TA 98 zone 55-10^3 55 µg*
Phase Aliphatique	+	+	+	−		

Les tests sur animaux étant longs et coûteux, les chercheurs ont été amenés à développer un ensemble de tests utilisant des microorganismes dont le plus connu est celui de Bruce Ames [2] pratiqué sur des souches mutantes de *Salmonella Typhimurium*. Nous avons utilisé ce test pour la détermination du potentiel mutagène des différentes poussières étudiées. Mais, préalablement au test d'Ames, l'échantillon global doit subir une séparation chimique qui n'entraîne ni dégradation, ni complexation des substances chimiques présentes.

Pour les deux types de poussières étudiées nous avons pratiqué une extraction par ultrasons [3] dans des solvants de polarité croissante [4] (benzène, dichlorométhane, acétone, nitrométhane, méthanol). L'extrait brut est évaporé à sec et repris par le cyclohexane qui subit une extraction suivant le protocole.

3. RESULTATS

Chaque phase d'extraction est testée à l'aide du test d'Ames. C'est avec la souche TA 98 avec et sans S9 Mix que l'on obtient les résultats les plus significatifs.

Nos résultats sont exprimés en vitesse de pouvoir mutagène, soit en revers/mg d'extrait brut, soit en revers/m^3 d'air.

La détermination de la cytotoxicité des deux phases permet de vérifier que l'effet toxique n'est pas suffisant pour inhiber ce pouvoir mutagène.

TABLEAU II. POUSSIERES D'ATMOSPHERE. RESULTATS QUALITATIFS

	Pouvoir mutagène		*Vitesse pouvoir mutagène direct*		*Vitesse pouvoir mutagène avec S9*	
	Direct	*Indirect*	*revers/mg extrait brut*	*revers m3*	*revers/mg extrait brut*	*revers/m3*
Phase Hydrophyle	+	+	*3875 zone 200 4 µg*	*25*	*1230 zone 4.10^2 4 µg*	*14,1*
Phase Aromatique	+	+	*210 zone 2550 25 µg*	*1,6*	*560 zone 2550 190 µg*	*2,8*
Phase Aliphatique	–	–				

Le tableau I indique pour les végétaux les résultats obtenus sur les différentes phases. Le tableau II indique les résultats obtenus sur les différentes phases d'extraction des poussières d'atmosphère.

4. DISCUSSION

Dans notre étude on constate que le pouvoir mutagène des poussières recueillies dans l'environnement d'une centrale thermique n'est pas négligeable. Il n'est pas dû uniquement au benzo-a-pyrène car, d'une part, la phase aromatique présente un pouvoir mutagène direct et, d'autre part, la phase hydrophyle a un pouvoir mutagène important.

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STILLAGE ADEQUACY THROUGH PRESS FILTRATION

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The National Alcohol Programme *Pro Alcool*, promoted by the Brazilian Federal Government, began as a feasible form of alternative energy, without, however, taking into account its impact on the environment. Every litre of alcohol manufactured generates 12–16 litres of stillage, a byproduct which unfortunately turned out to be a high-level pollutant.

To enable the National Alcohol Programme to be feasible in the State of Rio de Janeiro, the authors tried to find a practical way of utilizing the stillage. The requirements for the process were: low cost of equipment, ease of operation, and small working area.

The main objective was to make maximum use of the stillage, to reduce its pollution capacity, and to improve the cake which is the final product of the stillage press filtration process and which can be used for soil conditioning, fodder, gas production in biodigestors, and energy generation by incineration. The process would then be economically sound.

The coagulation process followed by press filtration is simple in concept and uses equipment that is already available in Brazil The initial financial investment is low and, owing to the degree of automation, only one operator is required. The equipment best suited for mechanical dehydration seemed to be the 'press filter'. The total costs (as at Feb.1981) were US \$0.075 per 1-kg cake and US \$0.204 per 2.7-kg cake.

Using this equipment on a pilot scale, with one single plate and one homogenization tank, the authors began tests to separate the stillage solids after adding the coagulant in order to produce a cake with 50–60% humidity and a clarified effluent with low biochemical oxygen demand (BOD).

From September 1980, industrial-scale experiments were conducted to optimize parameters such as coagulant dose and selection of the filter-cloth. The industrial prototype comprised a press filter 180 m long, with 10 fenolic resin plates (300 mm $\times$ 65 mm) reinforced with fibreglass to prevent corrosion, covered with a filter-cloth whose composition and dimensions would be decided during this investigation phase. The prototype enclosed a diaphragm pump with 689.5×10^3Pa (100 lbf/in^2) capacity. The structural frame of the filter was protected by surface treatment and corrosion-preventive paint. The unit included a 2500-litre polyester homogenization tank and a mixing device using diffused air provided by two blowers through a regulator tank. The prototype also had all the necessary command and automation devices and a 200-litre-capacity cart for collecting the cake.

In the operation tests, assays with stillage from six distilleries were conducted for four through 85 runs. Many types of filter-cloth were used as well as different concentration of coagulant agents. The operational press filter tests were mainly conducted with coagulant dosages of 2.0–2.7% concentration.

From the results of various tests it was concluded that:

(a) The presence of sinkable solids in the effluent did not induce a decrease in the organic load removal, indicating that these solids are predominantly inorganic and probably arise from some unidentified operational problem.

(b) A more significant decrease in the organic load, which is the basic goal of the project, occurred when coagulant concentration was equal to or higher than 2.0%. The samples at 2.5% reached the highest values: 63.47% and 60.33% in BOD and 76.66% and 73.89% in chemical oxygen demand (COD) removal (values taken as the minimum for the process optimization).

(c) When the effluent was clear, the results were always better than when the effluents, under the same operating conditions, were dark.

(d) The pressing time and the filtered volumes are the conditioning parameters for optimization of the process.

(e) Some of the filter-cloths tested showed uniform behaviour. The manufacturer agreed on the need for an improved filter-cloth.

Conclusions

The project of stillage adequacy through press filtration is now being developed in the northern region of the State of Rio de Janeiro, and has begun to show favourable results at the various phases, demonstrating the feasibility of the process.

The process has the advantage of low capital costs.

The economical and feasible process allows the stillage to be utilized with minimum pollution, while the cake produced can be used for soil conditioning, fodder, or energy production by incineration, with fertilizer ash as by-product. The stillage cake it the final product of press filtration; it has about 60% moisture and is rich in nutrients. The complete equipment requires an area of 20 m^2. The basic requirements of low capital cost, ease of operation, and small area have been satisfied; and a BOD and COD load reduction of about 60% and 75% respectively has been achieved. The stillage cake by-product is easy to handle and has physical and chemical characteristics that should make it widely used in Brazil and bring a quick return for capital expenditure.

IAEA-SM-254/56

LONG-TERM HEALTH IMPACTS OF ENERGY PRODUCTION

Preliminary assessment of environmental pollution by heavy metals and long-lived radionuclides for identification of research priorities

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The medium-term energy policy of the European Community foresees the integration and partial substitution of oil-fired power production by coal and nuclear energy.

The augmentation of coal combustion implies an increased mobilization of heavy metals into the environment and their incorporation into bio-geochemical cycles. This imposes the setting up of exposure limits and environmental quality standards for heavy metals released by fossil-fuelled power plants, establishing their maximum permissible release rate and maximum concentration in critical environmental compartments. Conversely, the augmentation of nuclear energy implies a potential mobilization of long-lived radionuclides from the nuclear fuel cycle.

Research has been under way since 1973 at the Joint Research Centre, Ispra, to improve the scientific information on which health and environmental protection criteria must be based. Model studies are used to identify critical areas and suggest priorities for the scientific research carried out at the Joint Research Centre on that subject, three of which are, briefly, as follows:

(a) The long-term implications of heavy metals released from coal-fired power plants in the European Community (local and regional implications);

(b) A comparison of the potential impact of long-lived radionuclides emitted from coal combustion and the nuclear fuel cycle (light-water reactors and liquid-metal fast breeders); and

(c) The long-term environmental implications of radioactive waste-management practices.

The assessment studies conducted at the Joint Research Centre indicate that the health impact of a greater use of coal and nuclear energy, as foreseen by the Community energy policy, is fully consistent with its environmental and health-protection policy. Research to improve the data base of assessment studies is continuing in the frame of the Community's multi-annual research programmes.

IAEA-SM-254/58

USE OF CHINESE HAMSTER OVARY CELLS IN EVALUATING POTENTIAL HAZARDS FROM ENERGY EFFLUENTS

*Application to diesel exhaust emission**

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Short-term bacterial mutagenicity tests have been used to estimate potential hazards due to emission of effluents from different energy production processes. However, the vast differences between mammalian cells and bacteria (membrane permeability, metabolic pathways, genome organization) make it necessary to confirm bacterial tests by the more relevant mammalian tests. Such an evaluation, based on the results of different tests, has proved useful in predicting the potential hazard of environmental pollutants [1].

At the Lovelace Institute, our method of testing energy effluents is to define the potential release sites and, working closely with the appropriate technology, to collect relevant samples, which are characterized physically,

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TABLE I. ADVANTAGES OF CHO CELLS

Well characterized	High plating efficiency
Genetically stable	Established quantitative assays
Near diploid	Coupling with exogenous metabolic systems

chemically and biologically. The biological toxicological tests include both bacterial and mammalian short-term toxicity and mutagenicity tests, as well as long-term whole-animal tests. Our mammalian short-term tests for characterizing the somatic and genetic toxicity of environmental complex mixtures are described here.

A well-characterized, genetically stable cell line, the Chinese hamster ovary (CHO) cells (Table I), is used in our mammalian short-term tests. Three end-points — cytotoxicity, mutation and sister-chromatid-exchange (SCE) — are used (standardized protocols, which are optimized for testing energy effluents, are outlined in Fig. 1). The cytotoxicity assay (Fig. 1(a)) measures the ability of the test substance to decrease colony-formation ability of CHO cells [2] and requires 7 to 9 days. The mutagenicity assay (Fig. 1(b)) requires approximately 16 days and measures the induction of mutation of the hypoxanthine-guanine phosphoribosyl transferase (HGPRT) gene locus. We use an assay that is modified [3] from that developed by Hsie and co-workers [4]. The SCE assay (Fig. 1(c)) measures the induction of exchanges between differentially stained sister chromatids. Our protocol is modified from that developed by Takehisa and Wolff [5]. Five days are required to conduct the SCE assay and 5 to 10 days for the chromosome analysis. As shown below, these tests respond quantitatively to the activity of known mutagens.

Application of these assays in evaluating toxicity and genotoxicity are illustrated in Fig. 2. Known direct-acting mutagens: N-methyl, N′-nitro, N-nitroso guanidine (MNNG), ethyl methane sulphonate (EMS); promutagens that required metabolic activation to be mutagenic: benzo(a)pyrene (B(a)P), dimethyl nitrosamine (DMN), and an environmental pollutant, a dichloromethane extract from diesel exhaust particles [2], were tested. The four mutagens used had a five orders of magnitude difference in specific activities. MNNG produced significant mutagenic and cytotoxic response at 0.1 μg/mltr, while DMN required 1000 μg/mltr for similar response. All the mutagens had dose-dependent cytotoxic (Fig. 2(a)) and mutagenic (Fig. 2(b)) effects. Dose-dependent induction of SCE was also observed (Fig. 2(c)), with the exception of B(a)P, which only produced a low response. The diesel exhaust extracts showed definite

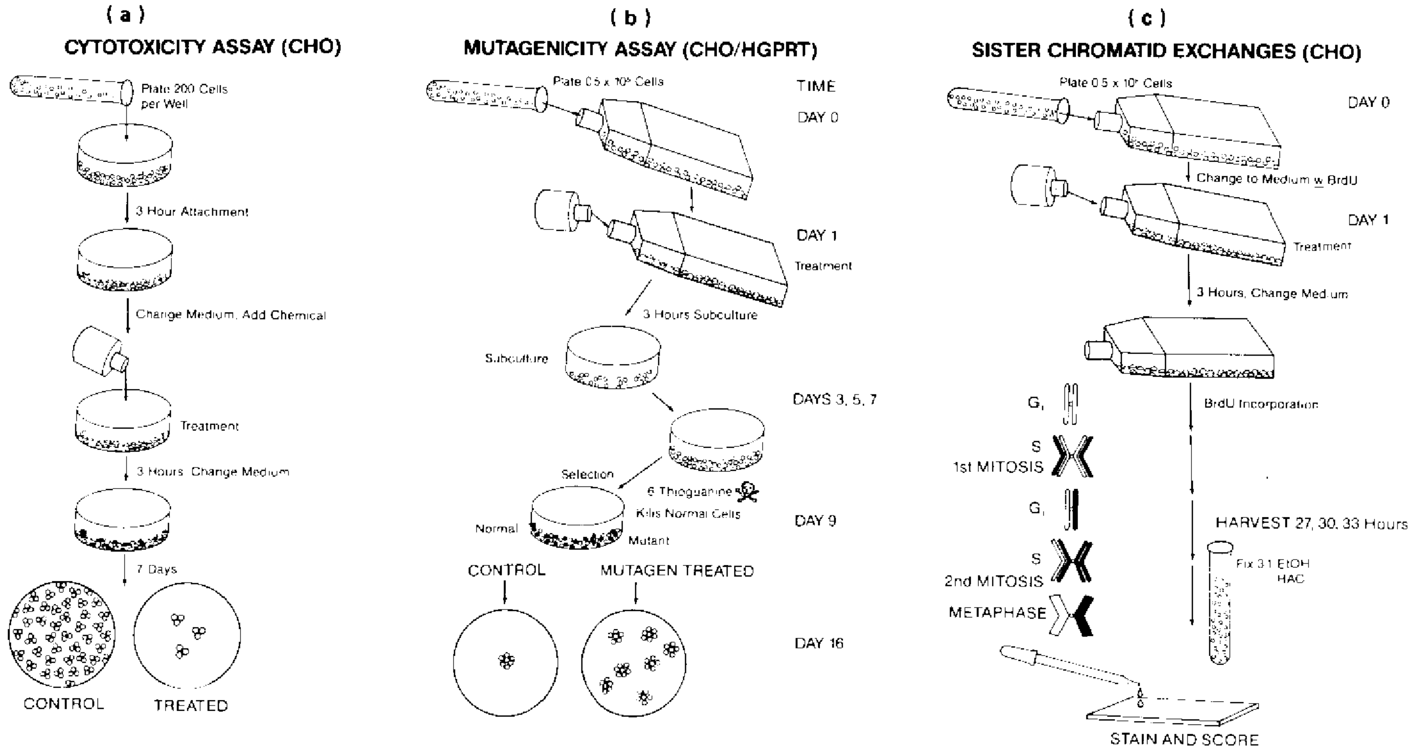

FIG. 1. Experimental design for the assays for cytotoxicity (a), mutagenicity (b), and sister-chromatid-exchanges (c) in CHO cells.

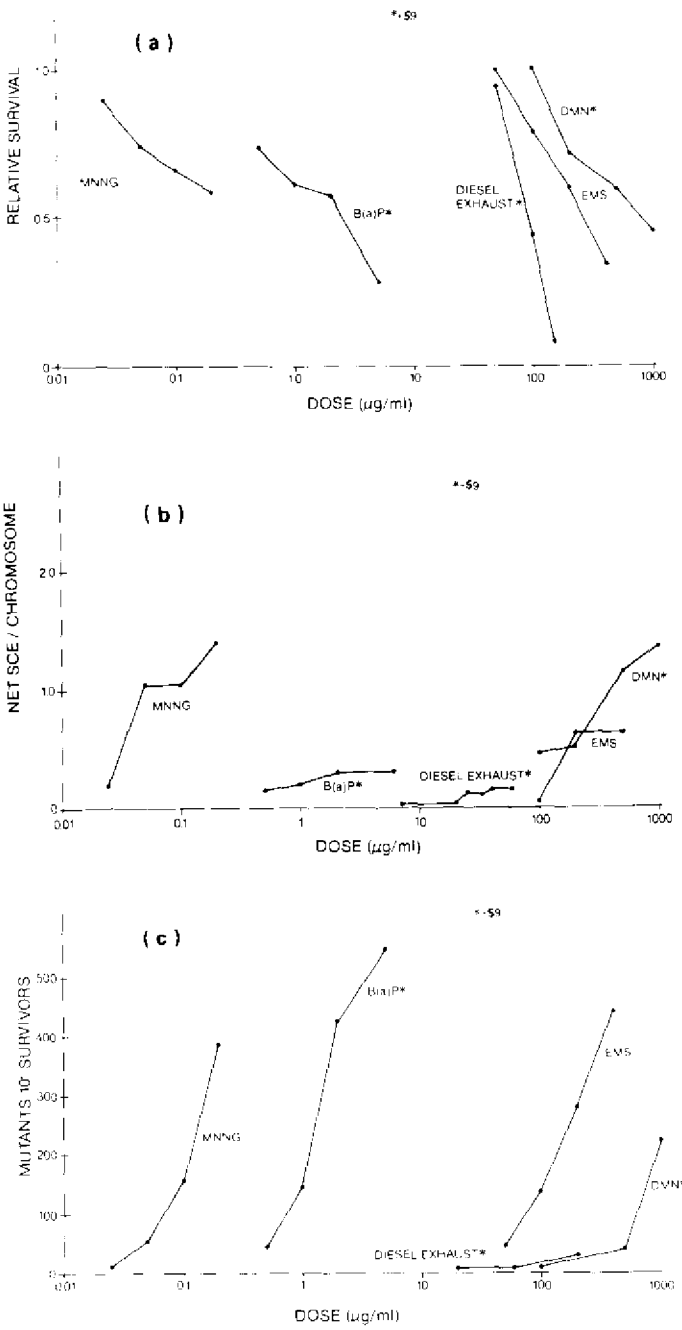

FIG.2. Application of the assays for cytotoxicity (a), mutagenicity (b), and sister-chromatid-exchanges (c) to known mutagens and diesel exhaust particle extract.

cytotoxicity (Fig. 2(a)), but only low genotoxic effects (Fig. 2(b) and (c)). For testing substances of unknown activities, like the diesel exhaust extract, a low genotoxicity is defined by lack of significant induction of genetic damage up to a dose that would yield 90% cell killing. The cytotoxicity assay therefore serves both to reveal the intrinsic toxicity of the substances tested and to yield the dose-range with which the more laborious genotoxicity assays will be carried out.

It is interesting that the diesel exhaust has less than 1/100th of the genotoxicity of a known environmental pollutant and animal carcinogen, B(a)P. This finding is very different from the bacterial mutagenicity studies [6, 7], in which the diesel exhaust extract was found to be highly mutagenic. We have evidence that the bacterial mutagenicity of the diesel exhaust extract may be due to the presence of nitroaromatic compounds [8]. The difference in activity of the extract on mammalian cells and bacteria may be a result of the deficient metabolism of nitrocompounds in mammalian cells. Thus the finding illustrates the need for both bacterial and mammalian tests to avoid over- or under-estimation of the potential hazard of the test substance.

Using the mammalian tests illustrated here, we have further shown that serum proteins, sulphydryl compounds, and lung and liver cytosols could detoxify the diesel exhaust extract [2]. In defining the genotoxicity, we found that although the diesel exhaust extract itself had low genotoxicity, it could enhance the mutagenicity of other chemical mutagens [9]. The possible interaction between diesel exhaust and other environmental mutagens should therefore be one of the important parameters involved in the estimation of human health risk.

Although results from the short-term assays as described here cannot be directly extrapolated to human risk, they serve as quantitative assays with which the biological effects of energy effluents can be studied under well-defined conditions. Data obtained from these established, highly quantitative assays can then be evaluated with data from other experimental systems as well as epidemiological data, allowing the potential hazard of energy effluents to the human population to be estimated.

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HEALTH ASPECTS OF WOOD-FUEL USE IN THE UNITED STATES OF AMERICA

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Wood-fuel is the largest biomass energy source used in the USA today: current use in residential heating is about 5×10^8 GJ [1]. Recent growth in use of residential wood-stoves has been great, and this growth has been associated with observed increases in wood-cutting and gathering accidents, house fires and air pollution. The object of this study was to develop quantitative estimates of health effects of residential wood-fuel use in the USA and provide some comparison with coal and oil alternatives. Health hazards of wood-fuel use in the USA differ from those in developing countries, e.g. in the more widespread use of chain saws, differences in gathering techniques, and differences in housing and combustion technology. A fuel-cycle approach was taken, estimating energy flows along a trajectory from extraction or harvesting to ultimate end use. Non-quantified health risks are burns and indoor air pollution. For comparison, an end use of supplying space heat for 1 million dwelling-years was taken. This is 6×10^7 GJ heat or 8.8×10^7 GJ wood input at 69% efficiency. Health effects were estimated at gathering, transport and combustion stages. Oil and coal alternatives were scaled from earlier work [2].

Wood-cutting and gathering in North America is a high accident-risk job, rivalling underground coal-mining. Risk depends on technology and method of operation. Based on estimates by Chockie et al. using 1976–78 statistics for the lumber industry in the state of Washington, there are $\sim 1 \times 10^{-7}$ accidental deaths per GJ wood for whole-tree removal [3]. Culling or collection of residue after removal of tree trunks for lumber involves more person-hours since the material is more dispersed and thus there are more injuries per unit energy collected. Figure 8 of the Keynote Address (p.14), shows coal-mine accidents for comparison; they are for a mix of 57% surface and 43% underground mining. Because of uncertainties in extrapolation of data from lumbering to fuel wood, the estimates for wood are probably accurate to no more than ± a factor of 2.

It was assumed that wood is transported in large trucks carrying 27 t for a 160-km round trip. The above figure is based on National Safety Council data yielding 4.4×10^{-6} accidents per km for fleet trucks and 3×10^{-3} deaths per accident. These estimates are probably minimal. Limited data on trucks carrying logs and poles and lumber indicate accidental death rates may be ten times higher [4]. The coal comparison assumes a mix of 53% rail with 400 km haul distance, 22% barge with 770 km haul distance, 11% truck with 80 km haul distance and 14% mine-mouth operation.

House fires are a serious accompaniment to the rapidly increasing use of wood-stoves in the USA. Most fires arise from unsafe installation of stoves and chimneys; a Massachusetts study found that 75% of wood-stove fires were due to this [5]. Nonetheless, since the stove is usually in the living area of the house and must be refuelled and ashes removed regularly, a wood-stove seems inherently more hazardous than a conventional furnace. The estimate of fires is based on 1976–78 experience in New Hampshire and Vermont from which specific data on wood-stove fires were available. To eliminate the effect of unsafe installation, only 25% of the fires were included. Two hundred and forty-four house fires were associated with combustion of $\sim 2.8 \times 10^{7}$ GJ of wood yielding $\sim 1 \times 10^{-7}$ deaths/GJ wood. No deaths occurred from oil heating. The risk of fire from electric heating was not calculated. Because of the rapid growth of wood-stove use and the difficulty of obtaining accurate statistics on wood consumption, the accuracy of the estimates is probably ± a factor of 3.

While wood is generally considered a clean fuel, its combustion emits particles, polycyclic organic matter (POM), carbon monoxide, and possibly nitrogen oxides at higher rates per unit energy than oil. Benzo(a)pyrene (B(a)P) is a commonly used index of POM in estimating chemical carcinogeneses. Lipfert estimated maximal seasonal concentrations of 1 ng · m^{-3} B(a)P over an 83-km^2 urban area burning 2.2×10^{6} GJ wood within the area [6]. This is an annual average contribution of about 0.3 ng · m^{-3}. For a population of 116 per km^2 (comparable to the population density in the coal alternative), wood consumption would indicate about 20% of homes with wood stoves. Population exposure would thus be 116 person/km^2 $\times$ 83 km^2 $\times$ 0.3 ng · m^{-3} = 2900 person-ng · m^{-3}. Fischer [7] estimates a dose-response function of roughly 2×10^{-5} cancer deaths per person-ng · m^{-3} B(a)P. This yields an estimate of 2.6×10^{-7} cancer deaths/GJ.

TABLE I. RISK IN SUPPLYING ONE DWELLING WITH WOOD-FUEL FOR 40 YEARS, SCALED TO A SINGLE INDIVIDUAL

Source	Risk of death
Wood-gathering	3.6×10^{-4}
Transport	1.2×10^{-5}
Fire	3.2×10^{-4}
Air pollution	9.2×10^{-4}
Total	1.6×10^{-3}

Comparison of air pollution effects is particularly difficult. It is subject to assumptions such as the fraction of those exposed who are also supplied with energy and the extent of long-range transport of pollutants considered. Both wood and coal estimates include only local exposure. The degree of transport of POM to more distant populations is unclear. Consideration of long-range transport of sulphate pollution, upon which the coal health effects are based, would lead to effects closer to those shown for wood. Owing to these factors, the error associated with using the B(a)P index and assumptions on population distribution, the level of uncertainty in the air-pollution--health-effect estimates is clearly greater than ± a factor of 10.

A convenient way to help understand the risk estimates is to scale them to a single person supplying one dwelling with wood over a 40-year period. This results in a total risk of fatality of $\sim 1.6 \times 10^{-3}$ (see Table I). For comparison, the risk of automobile-accident death in the USA for the same period is $\sim 9.5 \times 10^{-3}$. Thus, while the level of uncertainty in the estimates is great, wood fuel presents risks which are of the same order as more conventional heating technologies. Both are still well below the overall risk of other common activities. Nevertheless, many aspects of the risk are clearly amenable to preventive measures.

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HEALTH RISKS OF PHOTOVOLTAIC ENERGY TECHNOLOGIES

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Health risks of photovoltaic energy technologies arise from mining, processing and refining raw materials and from fabrication, installation, operation and disposal of devices used to convert sunlight into useful energy. Using an accounting approach, public and occupational health risks are examined for four different photovoltaic cell alternatives:

(i) silicon n/p single-crystal cells produced by an ingot process;
(ii) silicon metal/insulator/semiconductor cells produced by ribbon growing;
(iii) cadmium sulphide backwall cells produced by spray deposition; and
(iv) gallium arsenide cells produced by modified ingot-growing.

These alternatives cover a range of manufacturing options (e.g. ingot versus spray deposition) and materials (e.g. silicon versus arsenic) which might be used in future commercialization efforts.

The generic design for reference technologies was based upon a 25-kW_{pk} decentralized photovoltaic system. This system serves as a preliminary example of a photovoltaic installation but with two important qualifications regarding its applicability to the design of future installations: (a) structural material requirements greatly overestimate materials likely to be included in future design; and (b) fabrication technology is rapidly advancing towards thinner and more efficient solar cells. Because the system was conservatively designed, materials used exceed those likely to be present in future designs. As a result of these excessive demands, health risks subsequently calculated should be recognized to represent worse case estimates.

Public health effects stem principally from pollutants released during various stages of the photovoltaic energy cycle. This initial study explored only effects of arsenic, cadmium and silicon. Although the silicon technologies are nearest to commercialization, prediction of public health risks from exposures to silicon or silicon monoxide is not possible. The toxicology of silicon dioxide (silica) is well documented, but the toxic effects of elemental silicon and silicon monoxide have not been investigated and cannot be assumed to be the same as for silicon dioxide. In contrast, information on health effects of arsenic and cadmium compounds is extensive, and dose-response functions for kidney damage from cadmium and lung cancer from arsenic can be constructed.

Results of modelling efforts suggest that chronic exposure to cadmium emitted to the atmosphere during the photovoltaic energy cycle is unlikely to increase cadmium levels in the kidney cortex above $\sim 60\ \mu g \cdot g^{-1}$. This is well below the concentration (150 to $300 \mu g \cdot g^{-1}$) at which renal damage is expected. Arsenic released during the photovoltaic energy cycle could induce a lung cancer rate of 10^{-8} to 10^{-5} deaths/100 000 persons per year within 50 miles of photovoltaic-related facilities. The present lung cancer rate from all causes is ~ 43 deaths/100 000 persons per year. If long-range transport is considered, then total deaths in the USA from arsenic emissions in the photovoltaic energy cycle could range from 10^{-3} to 10^{-2} deaths per year per 10^{12} Btu.

Effects of acute exposures from release of arsenic and cadmium during fire or accidental consumption of these materials were also examined. Potential exposures and doses were calculated and then compared with acute doses found in the literature. In both, exposure levels were well below thresholds where death would be expected.

Thus, potential public health effects of the use of arsenic and cadmium in the photovoltaic energy cycle appear to be small in comparison with other hazards presently noted. Public health hazards related to silicon and silicon monoxide exposures cannot be assessed owing to lack of toxicologic information at this time. Measurements of exposures and compilations of some basic toxicologic information are required to improve the accuracy of the arsenic and cadmium damage estimates and to assess risks to be expected from silicon exposure.

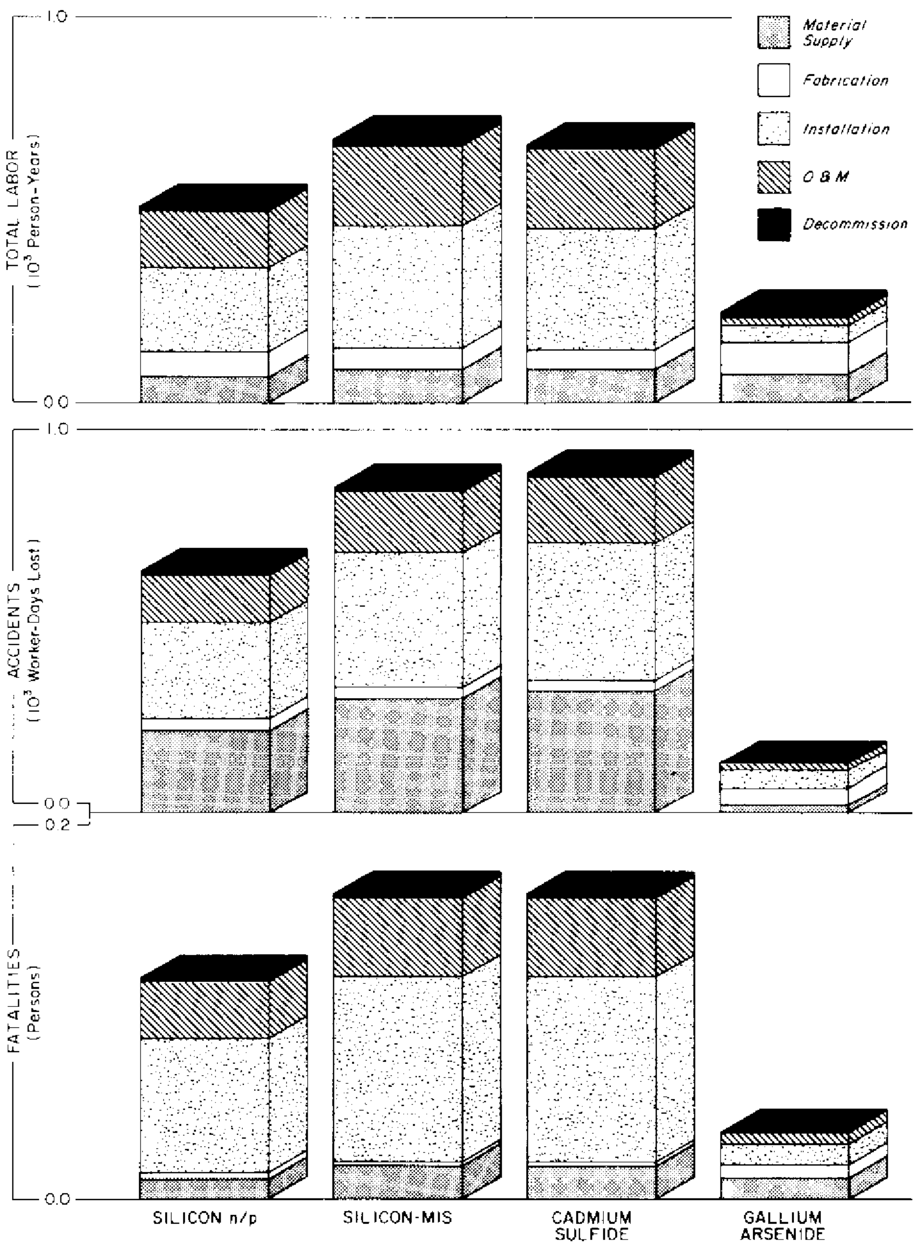

FIG. 1. Summary of labour and occupational health and safety impacts of photovoltaic energy technologies (per 10^{12} Btu output).

Chemical and physical hazards produced in the workplace are the source of occupational health risks, e.g. effect of trauma to a hand or carcinogenic hazard of exposure to arsenic. Risks from chemical hazards were assessed using approaches similar to those described for public health. As noted, the toxicology of silicon dioxide (silica) is well documented and clearly shows that hazards exist in dirty situations like quartz mining. But silicosis, commonly suggested as a potential hazard in the photovoltaics industry, is entirely preventable; its prevalence in existing industries has declined in the past decade. Exposures to silicon and silicon monoxide are expected from the different activities examined, but again no toxicologic information for these materials exists. Cadmium health-effects modelling suggests that chronic exposure in the workplace to cadmium levels above $\sim 10\mu g{\cdot}m^{-3}$ is likely to produce cadmium kidney burden levels ($\sim 200\mu g{\cdot}g^{-1}$) near or exceeding a concentration above which renal damage can be expected (150 to $300\mu g{\cdot}g^{-1}$). In-plant air quality measurements exceeding $10\mu g{\cdot}m^{-3}$ level are commonly found in smelters. Results of arsenic modelling analysis suggest that if workers were chronically exposed to in-plant arsenic levels of $10\mu g{\cdot}m^{-3}$ their cancer rates would be 10–50% of the background levels for all lung cancers.

Effects of other chemicals used in the manufacturing process were also explored, but only qualitatively. Many of the materials used could present either acute or chronic hazards, but actual exposures are unknown. More accurate estimates of these chemical hazards depend upon compilation of in-plant measurements describing type and frequency of exposure, analysis of toxicologic information, and subsequent analysis of the expected risks.

Most occupational mortality and morbidity effects probably relate to industrial risks similar to those encountered in the day-to-day operation of any industrial operation. Using an accounting method, risks from the extraction of the raw materials through to installation and eventual disposal of the photovoltaic devices were examined (Fig. 1). Among the four alternatives, there are only small differences in total health costs. Material supply, installation and operation appear to contribute substantial portions of the damage within the entire photovoltaic energy cycle. The contribution of the fabrication facilities is small. At fabrication facilities, the relative risk to workers (~ 50 worker-days lost (WDL)/100 man-years (MY)) is smaller than risks in other energy technologies; for example, in coal-mining the relative risk to workers is >100 WDL/100 MY. The labour intensity of the photovoltaic energy cycle, however, increases the absolute or societal cost (risk per worker $\times$ number of workers) of producing electricity by photovoltaic energy systems. Net health risks are nevertheless close to zero.

More detailed information on the results presented in this note are described elsewhere [1-3].

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Session VI

METHODS FOR QUANTITATION AND COMPARISON OF HEALTH RISKS

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Mémoire présenté sur demande

LA QUANTIFICATION DU DETRIMENT ET LA COMPARAISON DES RISQUES SANITAIRES
Problèmes méthodologiques

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Abstract–Résumé

QUANTIFICATION OF THE DETRIMENT AND COMPARISON OF HEALTH RISKS: METHODOLOGICAL PROBLEMS.

Some of the methodological problems involved in the quantitative estimate of the health detriment of different energy sources and in risk comparison are described. First, the question of determining the detriment is discussed from the point of view of the distortions introduced in the quantification when dealing with risks for which the amount of information available varies widely. The main criteria applied to classifying types of detriment are then recalled. Finally, the problems involved in comparisons are outlined: spatial and temporal variations in the types of detriment, operation under normal and accident conditions, and the risks to the public and workers.

LA QUANTIFICATION DU DETRIMENT ET LA COMPARAISON DES RISQUES SANITAIRES: PROBLEMES METHODOLOGIQUES.

On envisage quelques uns des problèmes méthodologiques posés par l'estimation quantitative du détriment sanitaire des différentes énergies et la comparaison des risques. La question de l'identification des détriments est d'abord discutée sur le plan des biais introduits par souci de quantification à propos de risques caractérisés par des niveaux de connaissance très hétérogènes. On rappelle ensuite les principaux critères utilisés pour établir une classification des détriments. Enfin, on envisage les problèmes posés par la comparaison: hétérogénéité spatiale et temporelle des détriments, fonctionnement normal et accidentel, risques pour le public et pour les travailleurs.

INTRODUCTION

Le thème du risque est souvent intervenu au cours des débats qui ont récemment concerné les choix énergétiques. Il constitue à présent une des dimensions indispensables à prendre en compte parmi les éléments qui interviennent dans l'appréciation des politiques dans ce domaine. Cet intérêt a suscité l'émergence d'études, qui sont à présent nombreuses et qui visent à une analyse systématique de ces aspects, en particulier en termes d'impact sur la santé et les écosystèmes naturels. Les recherches sur l'évaluation du risque procèdent d'un double objectif: assurer une clarification des données objectives en les présentant sous une forme systématique qui permette la comparaison, mais aussi, assurer à terme une meilleure allocation des ressources de sûreté et de protection en dégageant des priorités ou en soulignant d'éventuelles incohérences. Face à ces questions, de nombreux problèmes méthodologiques se posent qu'on abordera ici, en se limitant au domaine des impacts sur la santé de l'homme.

1. IDENTIFICATION DES DETRIMENTS

Dans sa définition de la santé qu'elle proposait voilà déjà plus de dix ans, l'Organisation mondiale de la santé (OMS) parlait d'un «complet état de bien-être physique, mental et social». Il est utile de reprendre un tel concept pour évoquer seulement les multiples dimensions potentielles qui pourraient être retenues dans une analyse des risques des systèmes énergétiques. Toutefois, puisqu'il s'agit surtout ici d'une perspective de clarification des débats, il semble raisonnable de limiter d'emblée le champ de ce qu'on conviendra d'appeler le détriment sanitaire à un sens restreint, celui du bien-être physique, et donc de laisser de côté le vaste domaine des effets d'ordre psycho-sociologique. Il ne s'agit pas d'ailleurs de nier l'importance de ces derniers aspects mais, plutôt, de considérer que leur nature implique un traitement différent et spécialisé, pour lequel il conviendrait que d'autres types d'analyses soient envisagés. Nous nous limiterons donc ici, pour notre part, à évoquer les problèmes posés par toute tentative d'objectivation et de quantification des risques de la première catégorie. Mais, déjà à ce niveau, apparaissent de nombreuses difficultés.

La question de l'identification des risques est la première qui se présente. En effet, les multiples activités industrielles actuelles ou envisageables pour la production d'énergie couvrent un champ immense, où le nombre de risques potentiellement envisageables est énorme. On observe d'ailleurs, sur ce point, un effet de convergence dans les études effectuées dont la plupart tendent à se polariser sur les mêmes catégories de risques. On comprend bien la logique d'un tel processus, mais il faut prendre conscience également des biais ainsi introduits au niveau des résultats obtenus. A ce propos, plusieurs remarques doivent être faites.

1.1. Les biais de la quantification

Le recours à des données quantitatives est bien sûr l'objectif immédiat. Mais on doit prendre garde au fait qu'il peut entraîner une série de biais. Le recours systématique à la quantification a accentué en effet l'importance du connu par rapport à l'incertain. Or, souvent, le fait de disposer d'une information riche sur certains aspects du risque signifie que des efforts particuliers de contrôle et de recherche lui ont été consacrés mais, pas nécessairement, que ce risque est important en valeur relative. A l'inverse, la difficulté à quantifier un effet ne doit pas conduire à en négliger l'importance.

1.2. Les enjeux sous-jacents

D'une façon générale, le fait de retenir ou non une catégorie de risque dans l'analyse n'est pas toujours suffisamment expliqué. En effet, le critère d'un tel choix est, théoriquement, l'importance relative de ce risque par rapport à d'autres qu'on négligerait et ceci est difficile à mettre en œuvre. Aussi, de multiples considérations interviennent, qui relèvent aussi bien de la disponibilité des données scientifiques et techniques (on retrouve alors le biais signalé plus haut) que de l'inquiétude que ce risque a suscité dans le public ou de considérations plus complexes. En effet, de multiples enjeux économiques, sociaux et politiques sont sous-jacents à des débats de cette nature. Ce ne sont pas de pures considérations d'ordre scientifique qui justifient à elles seules l'intérêt pour tel et tel problème. Dans la panoplie des risques potentiellement envisageables, il est toujours possible à tel ou tel intervenant de brandir la nouvelle forme de menace qui fera pendant à celle issue d'un débat antérieur. S'il n'y a pas de solutions toutes faites pour résoudre une telle question, il faut néanmoins y prendre garde et conserver à l'esprit cette difficulté générale dans ce type d'étude.

1.3. Les lacunes des connaissances

Au-delà de ces considérations, il faut rappeler également que ce vaste domaine de l'analyse et de l'identification du risque est soumis à l'incertitude inhérente à la connaissance scientifique et à son évolution. Les recherches nombreuses suscitées par ces débats font progresser cette connaissance et nous obligent à revoir de façon continue nos vues sur un certain nombre de problèmes et de priorités. Le débat et la controverse sont donc un état qu'on pourrait dire normal à propos de ces résultats qui ne peuvent être tout à fait convergents. Certains risques sont d'ailleurs négligés du simple fait, non pas des incertitudes, mais d'une lacune complète des connaissances scientifiques disponibles.

2. CRITERES DE CLASSIFICATION DES DETRIMENTS

Face à l'extrême variété des risques susceptibles d'être envisagés, un certain nombre de catégories sont généralement proposées conformément à un usage qui repose, dans une large mesure, sur l'expérience du risque radiologique. Ces catégories sont établies à partir des critères suivants:

– Population ou structure affectée: risque professionnel, risque pour le public (effet de voisinage des installations mais aussi d'usage de l'énergie), pour l'écosystème naturel (considéré soit comme milieu de transfert ou comme système biologique final), etc.

– Extension spatiale et temporelle des impacts: il y a lieu de distinguer à ce propos six combinaisons au moins selon qu'il s'agit d'effets locaux, régionaux ou mondiaux à court terme ou d'effets à long terme, c'est-à-dire, par exemple, qui affectent la descendance des individus exposés (effets génétiques).

– La nature de l'effet biologique: effet somatique, génétique par exemple. On notera à ce propos que la pratique consistant à tenir compte des effets génétiques semble encore privilégier le cas des radiations ionisantes alors que de nombreux mutagènes chimiques sont négligés.

– Le mécanisme biologique de l'effet également, selon qu'il peut être considéré comme stochastique ou non par exemple.

– L'importance et la gravité de l'état: *affection mortelle*, susceptible ou non de faire l'objet de traitement approprié ou *morbidité*, donnant lieu à des soins médicaux, arrêt de travail et dont l'importance relative ne peut qu'être appréciée sur la base d'un ensemble de critères variés.

– Le mécanisme d'apparition du risque: circonstances normale de fonctionnement des installations et cas accidentels.

A ce niveau, il convient surtout de noter que l'exigence principale que l'on doit attendre des études comparatives n'est pas seulement liée à l'exhaustivité des dimensions retenues. Il importe également qu'un traitement de même nature soit opéré à propos de l'ensemble des risques comparables qu'on prend en compte et, si cela n'est pas rendu possible du fait de l'absence des données nécessaires, il est souhaitable qu'une telle particularité soit clairement indiquée.

3. L'EVALUATION COMPARATIVE DES DETRIMENTS SANITAIRES

3.1. Les objectifs possibles

Le problème de l'évaluation du risque et du choix des méthodes appropriées pour y parvenir ne peut être traité dans l'absolu. Il ne peut être envisagé qu'en fonction des objectifs poursuivis. Ceux-ci doivent être précisés. En effet, historiquement, les études ont commencé par se limiter à une comparaison des effets d'installations industrielles isolées: par exemple les centrales thermiques classiques et nucléaires. On a ensuite inclus l'ensemble des opérations industrielles liées à l'extraction, à la préparation et au transport du combustible, introduisant ainsi le concept de cycle complet. Des études plus récentes incluent les opérations de démantèlement et de gestion des déchets. On est passé ainsi d'un objectif de comparaison de techniques électrogènes à celui de comparaison de filières énergétiques complètes. Toutefois, à partir du moment où on a voulu intégrer également des technologies décentralisées, une extension devenait nécessaire à l'ensemble des activités d'utilisation de l'énergie finale dans la mesure où celle-ci devenait indissociable de sa production. Par ailleurs, la prise en considération de techniques présentant une large diversité du point de vue de leur contenu capitalistique (charbon, nucléaire, hydraulique par exemple) entraîne la nécessité d'un calcul incluant les risques induits par la construction. Dans cette évolution, l'idée d'une comparaison de «technologies» alternatives perd de sa consistance et disparaît donc progressivement au profit d'un objectif de comparaison de scénarios énergétiques complets établis par combinaison en proportion variable de différentes filières (en termes d'exploitation et d'investissement).

Quels que soient l'objectif retenu et son degré d'ambition, les technologies analysées doivent être nécessairement spécifiées dans le cadre de scénarios précisant:

- la localisation retenue par les installations (et donc les caractéristiques des populations exposées),
- l'état des réglementations de sûreté et de protection,
- l'origine des matières premières et leur caractéristique,
- l'état précis des technologies,
- l'énergie *nette* produite (afin de prendre en compte l'autoconsommation des chaînes énergétiques considérées).

3.2. La prise en compte du temps

Trois caractéristiques au moins sont à considérer sur le plan temporel:

- La durée de fonctionnement des installations.
- La durée de vie de polluants non dégradables dans l'environnement, problème souvent soulevé dans le cas des radioéléments de longue période et des déchets radioactifs, mais qui se pose de la même façon pour tous les produits chimiques partiellement non dégradables, c'est-à-dire qui conservent leur toxicité dans l'environnement sous leur forme initiale ou à la suite de transformations physico-chimiques diverses. Le concept d'engagement de «dose collective» introduit à propos de radionucléides constitue un moyen pratique pour rendre compte de ces phénomènes d'irréversibilité au cours du temps pour des produits

partiellement ou non dégradables. Il serait souhaitable que l'usage d'une telle notion soit étendu au cas des substances chimiques présentant des caractéristiques de cette nature.

– La durée des temps de latence qui peuvent intervenir entre le moment de l'exposition et celui de l'apparition des effets pathologiques induits.

3.3. La spécification du choix des sites

Elle est très importante pour l'estimation des effets sanitaires au niveau du public. Sur le plan de la sûreté, comme de la protection, la même technologie placée dans des environnements naturels et humains différents aura des effets «sanitaires» très variables. La définition des sites constitue donc une phase importante du travail d'évaluation; on a recours souvent pour cela à la notion de «site moyen» défini empiriquement pour des conditions générales de transfert (mer, lac, rivière) et de population compatibles avec les situation nationales moyennes qu'on envisage. Il est certain en tout cas que ces divers éléments peuvent être très différents selon le contexte national qu'on étudie et aboutir en conséquence à des résultats très variables en termes d'effets sanitaires sans que ces différences soient imputables à des méthodes ou à des données de base divergentes. Ces considérations suggèrent également qu'on ne peut pas directement extrapoler sans précaution les résultats obtenus dans un pays donné à un autre.

3.4. L'expression quantitative du détriment sanitaire

Chaque fois que l'objectif qu'on poursuit est de nature comparative, on est amené à tenter de trouver une échelle de mesure commune à l'ensemble des détriments sanitaires considérés. Cependant, on se trouve confronté dans ce cas à l'extrême hétérogénéité des effets sanitaires rencontrés: accidents du travail, mortels ou bénins; pathologies cancéreuses mortelles ou guérissables, effets d'aggravation de pathologies respiratoires, etc. Deux grandes catégories d'indicateurs peuvent alors être utilisées pour résoudre ce problème:

a) ceux qui s'expriment en termes «épidémiologiques» (nombre de décès, années de vie perdues, jours de travail perdus, cas de maladies imputables, etc.);

b) ceux qui s'expriment en termes monétaires (coûts médicaux induits par telles ou telles affections, coûts des compensations obtenues au titre d'une maladie professionnelle, ou même pertes de revenus, de production consécutives à l'absentéisme pour fait de maladie, etc.).

Dans le domaine du risque considéré, il semble hautement préférable, pour des raisons liées à l'effort de clarification qu'on poursuit dans ces travaux, de se limiter aux indicateurs du premier type. Ceux-ci ont en effet pour mérite principal de ne pas dépendre entièrement du contexte national où ils ont été établis (à la différence des seconds) et donc de pouvoir faire l'objet de confrontations internationales valables. Par ailleurs, compte tenu du nombre finalement minime d'effets sanitaires qu'on peut attribuer à la production d'énergie, il est inutile de placer cette question dans le champ de l'économie de la santé et de s'engager sur une autre catégorie de problèmes délicats et encore mal résolus concernant l'évaluation du coût social du détriment.

3.5. Le cas du risque d'origine accidentelle

Celui-ci présente une double particularité. D'une part, les études quantitatives concernant l'évaluation des probabilités d'occurrence et des conséquences d'accident sont encore peu développées en dehors du cas de certaines installations (centrales nucléaires, barrages hydro-électriques); par conséquent, il faut se garder des effets de polarisation sur les résultats disponibles, alors que des études analogues n'ont pas été faites sur des installations présentant également de tels risques (raffineries, méthaniers, terrils, etc.). D'autre part, il semble indispensable de conserver dans la présentation des résultats concernant le risque accidentel son caractère spécifique. En particulier, il conviendrait d'éviter de calculer des valeurs moyennes du risque obtenu en multipliant des probabilités d'occurrence par des estimations de conséquences et de comparer sans précaution ces valeurs à celles qui correspondent au détriment en fonctionnement normal. Il semble donc qu'à ce propos l'effort de comparaison globale trouve une limite et qu'il soit préférable, pour des raisons de clarification réelle, de conserver une distinction nette entre les détriments en fonctionnement normal et accidentel, quitte à conserver la possibilité de comparaison «interne» au sein de ces catégories. Des remarques analogues pourraient d'ailleurs être faites à propos du risque pour les travailleurs et du risque pour le public d'une part, et de la distinction construction-exploitation d'autre part.

CONCLUSION

Au-delà de ces quelques remarques méthodologiques générales qui ne prétendent en aucune manière constituer une revue exhaustive, il paraît important d'insister sur la nécessité de présenter de façon relative, aussi souvent que cela est possible, les résultats des études, c'est-à-dire de trouver des éléments de référence permettant de situer les résultats obtenus par rapport au contexte dans lequel ils se situent, afin de dégager clairement les ordres de grandeur des phénomènes qu'on évoque. Par exemple, valeur de la pollution de fond sur laquelle vient se surajouter la pollution due à l'activité considérée, fréquence totale des pathologies et supplément apporté par l'activité en question, valeur moyenne des risques professionnels dans le contexte qu'on considère, etc.

Il importe, en effet, que ces études parviennent aux objectifs qu'elles poursuivent, c'est-à-dire clarifier les débats sociaux, rétablir à leur juste niveau des perceptions erronées des risques technologiques et aussi dégager des priorités ou des incohérences dans notre façon de gérer la sûreté et la protection.

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DISCUSSION

I.M. TORRENS: You made the point that adding together the health effects from accidents and from normal operation creates difficulties. The greatest factor as regards accidents in the case of nuclear power is uncertainty. It is thus difficult to envisage a comparison of the health effects of nuclear power and other energy sources in which the effects and probability of accidents are not included. Do you not think that this problem makes any risk comparison between nuclear and other sources of energy somewhat suspect?

H. JAMMET: No. Separate comparisons can be made for the detriment associated with normal operation and that due to accidents. Uncertainties exist in all power generation systems – hydro, gas and nuclear included.

F.A. SEILER: One important problem area is missing in your discussion of the temporal aspects. You proceed from releases and exposures directly to health effects, but omit the important aspect of dosimetry: we need to know how much of a given toxicant is deposited and is responsible for the health effects observed. Unfortunately, a large number of publications in this field adopt the same approach, with the result that there are great uncertainties in our risk asessments. A major effort is needed in this area if we are to improve the quality of our assessments.

H. JAMMET: The dosimetric aspects are indeed important if we wish to use exposure indices. We have used the effective collective dose as an index of exposure and it is representative of radiological detriment. The same methodology can be used for chemical detriment if the intake of toxic substances is known.

R. WILSON: The term or concept of rad-equivalence has been used by you and others at this Symposium, but I think it is both unnecessary and undesirable: unnecessary because exposure indices can be directly related to a risk level, and undesirable because the level of risk associated with 1 rad (or 1 rem) can change if the dosimetry changes, as has happened recently in the case of the Japanese atomic bomb survivors. Could you comment on this?

H. JAMMET: Exposure indices are a useful and simple tool in assessing radiological detriment. It is also possible, from the methodological point of view, to use them for chemical detriment if we have a good scientific equivalence for the stochastic effects between chemical and radiological substances.

Invited Paper

COMPARATIVE RISK ASSESSMENT OF TOTAL ENERGY SYSTEMS

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Abstract

COMPARATIVE RISK ASSESSMENT OF TOTAL ENERGY SYSTEMS.

The paper discusses a methodology for total impact assessment of energy systems, ideally evaluating all the impacts that a given energy system has on the society in which it is imbedded or into which its introduction is being considered. Impacts from the entire energy conversion chain ('fuel cycle' if the system is fuel-based), including energy storage, transport and transmission, as well as the institutions formed in order to manage the system, should be compared on the basis of the energy service provided. A number of impacts are considered, broadly classified as impacts on satisfaction of biological needs, on health, on environment, on social relations and on the structure of society. Further considerations include impacts related to cost and resilience, and, last but not least, impacts on global relations. The paper discusses a number of published energy studies in the light of the comparative impact assessment methodology outlined above.

1. INTRODUCTION

The purpose of this paper is to place health impacts of energy systems in the context of a total impact assessment. In many cases health and non-health impacts have common causes and cannot easily be treated separately. Furthermore, it is useful to have a notion of the relative importance of health and other impacts for different energy systems, in order not to be tempted to draw comparative conclusions on the basis of health effects alone, when a total impact assessment may lead to quite different conclusions.

In the following sections 2 - 5, some methodological elements will be discussed, and in sections 6 - 8 their application to individual energy systems will be taken up. Only examples of such applications will be dealt with, since no complete analyses have yet been performed.

2. TOTAL IMPACT ASSESSMENT

A full comparative assessment of different energy systems must consider all impacts of each energy system on the society into which its implementation is considered. Among these impacts are usually some that may be labeled positive, while others are labeled negative. Some impacts may be of a mixed character: e.g. the discharge of waste heat to a water body may improve living conditions for some of the flora and fauna of the water body, but living conditions may deteriorate for other parts of the biota. The positive impacts of energy supply are primarily the benefits associated with the tasks for which the energy is applied. These include production of services and goods. Some of the positive effects may be indirect, through a general stimulation of economic activity, through stabilisation of energy supply, or through positive changes in attitudes towards a given society.

The main areas of potential (negative) impacts of energy systems may be classified as related to the physical, the human and the social environment. The physical impacts include modifications of micro- or macro-climate, e.g. caused by carbon dioxide or heat releases, deterioration of terrestrial or marine ecosystem (e.g. by disposal of pollutants) and resource degradation (such as landscape effects of strip-mining). The impacts may be site-displaced (air pollution traveling across national borders) or time-displaced (migration of radioactive waste or other pollutants from burial site to drinking-water sources), and the impacts may depend on complex interactions exhibiting threshold effects (triggering of Arctic ice melting or, conversely, extension of glaciation).

The effects on individual human beings include health effects caused by work or public exposure. The causes may be safety related (accidents), they may be noxious substances, noise and stress-producing working conditions, and they may be radiation exposure. In the case of accident risks a special problem is presented by events with small probability but large consequences. This extends into the social impacts, since such events present not just individual but additional social risks, if the consequences are disruptive, or if the social risk perception is enhanced by the character of the accident consequences. Examples of impacts on the social environment are modifications in the distribution of burdens laid on different social groups (preponderance of poor people living close to a polluting power plant, while the power-consuming high-income groups have moved away), altered employment opportunities, shifts between different types of regional development (e.g. centralized versus decentralized), changes in control structure (establishment of novel institutions, perceived need for anti-terrorist protection of energy installations, etc.), demand on foreign currency and imported technology, modification of supply security (which may mean different things for the individual and for society, resilience meaning access to more than one supply option, in addition to reliability of a given energy supply system). And to conclude the list, any other interference with a society's range of goals will constitute an impact of the social environment.

3. RISK CONCEPTS

Risk is a common-language word which is used to express at least the following three different concepts:

Direct risk is the probability of experiencing one unit of damage (e.g. the loss of one human life) per unit of time [1]. It can be expressed in the product form

$$\underset{\text{(Consequences/year)}}{\text{direct risk}} = \underset{\text{(events/year)}}{\text{frequency}} \times \underset{\text{(consequences/event)}}{\text{damage}}$$

Social risk is a measure of the implications for a given society of an activity entailing a direct risk specified by frequencies and corresponding damage magnitudes of individual events. Of specific social concern are events with small probability but large consequences. This is a reflection of the differing capability of a given society for handling small, frequent accidents (such as automobile accidents) as compared with large-consequence events for which society is ill prepared (due to their low probability) or which society is simply incapable of handling due to its finite resources (e.g. number of hospital beds) or, at the extreme due to a collapse of vital institutions (e.g. in case of a major nuclear reactor accident affecting the administrative capital of a small country) [2].

As a crude way of quantifying the social risk it has been proposed to use an expression of the form

$$\text{social risk} = \text{frequency} \times (\text{damage})^p$$

where the damage is raised to some power p (larger than one), which depends on the structure of the society in question [3].

Perceived risk is the risk of a given enterprise as perceived by the public. It may differ from both the direct and the social risk, and may exhibit short-term fluctuations and may be strongly influenced by coverage in media. It also reflects public confidence/mistrust in authorities and in many cases a large discrepancy between perceived risks and officially stated direct or social risks may reflect a history of previously experienced inaccuracy of official reassurance statements. In this sense, perceived risks would have to be taken seriously in risk assessment. Generally, emotional expressions are of course basic in defining social priorities.

4. COMPARATIVE RISK ANALYSIS

Some of the impacts of an energy supply system can be quantified, others cannot. Of the quantifiable impacts, some may be evaluated in monetary units, others may not. A complete assessment of a given system includes a political evaluation of all the impacts, and a comparative assessment of different system choices involves a weighting of the impacts of the systems against each other. Since the impacts of different systems according to the definition used here may be incommensurable, such a weighting involves value judgements, i.e. a political decision. If the subset of impacts lending themselves to economic interpretation is considered in isolation, the term 'full costing' is applied, in order to make a distinction from the direct costs alone.

A comparative assessment of energy systems must deal with two major obstacles: the existence of unacceptable impacts and of impacts with large uncertainties. The definition of impacts which a society will consider unacceptable should be made open to democratic debate. Examples may be catastrophic accidents of disruptive size and major impacts occuring with a time- or site-displacement (so that other - eventually future - societies would bear the burden without sharing the benefits). Once a society has agreed which impacts it considers unacceptable, it would accept only such designs of energy systems for which unacceptable impacts are absent. The problem of uncertain impacts may also be dealt with by excluding energy systems with uncertainty intervals stretching into the unacceptable regions. More problematic is the intercomparison of different systems with large uncertainties of different nature. Again the rules for comparison should in such cases be derived by full public participation, since such rules are necessarily politically debatable. Under the heading 'uncertainty' should also be considered the possibility of future changes in the attitude to selection of acceptable and unacceptable impacts. Elements of a methodological approach to comparative assessment of energy systems may be found in [4-8].

In identifying the impacts of a given energy technology, the full energy cycle must be considered: from construction of equipment, inputs of energy and materials, extraction and refining of fuels - if such are involved - through operation under normal and abnormal conditions, to eventual dismantling of the equipment and disposal of any byproducts emerging as a result of the energy conversion scheme. Furthermore, impacts may arise from the use of the transport, transmission and use of the energy produced, from the institutions created for management of the supply system, and so on.

Many studies have compared some impacts of different energy systems on the basis of equal energy production, e.g. considering various fuel cycles from resource extraction to the generation of electric power, on the basis of given power production. This would ignore any differences occuring in the further transportation, conversion and use of the energy all the way to the final services provided. The procedure would not catch differences between centralized and decentralized systems in terms of power transmission requirements, and it would not be capable of treating energy systems not based on a simple energy production, such as passive solar houses or systems involving changes in end-use structure leading to the elimination of a certain energy use.

The proper basis for comparing energy systems is thus the service provided by the energy system (the term "end-use" could be applied but unfortunately it is often used for "energy delivered to the final customer", a quantity that may exceed the energy spent on the desired service by a large factor, cf.[9]). In many cases there would be alternative ways of providing the service in question, with different amounts of energy input corresponding to different system designs and technology. Energy input may then be substituted by measures leading to a higher efficiency, in which case the impacts of energy provision must be compared to the impacts associated with the measures of efficiency improvement. This can only be done on the basis of an identical service delivered. This concept will be further illustrated in sect.7.

5. MARGINAL AND COMPLETE ENERGY TRANSITION

If only a small part of an existing energy system is replaced by a new one, the evaluation of indirect impacts is simplified. In this case, the gross social structure is given, and impacts associated with e.g. energy or raw materials inputs to the new system part may be evaluated using the existing production methods, for which impacts are in principle known or could be measured (except for the possibility of latent time-displaced impacts). The impact on employment, balance of foreign payments etc., is similarly determined, and social impacts - for instance through changes in energy use styles - would be accessible through means such as interview studies.

As the change in energy systems becomes more than marginal, it would no longer be acceptable to base the evaluation on the present surrounding system, i.e. the presently employed methods of providing process energy, materials, etc., for the new energy technology, and the present social structure as an indicator of the impact that the new energy technology may have on its surrounding society. As the energy system changes in a major way, new approaches to obtaining energy and materials inputs will come into play. This does not necessarily mean that each new energy source will provide the energy for its own establishment, because it may furnish other forms of energy than those which it requires during the construction phase, but the changing mix of energy sources will define the energy inputs drawn upon at any given time. Similarly, a dynamical approach to materials provision, and to social impacts, must be used. This is highly demanding, since it demands a model for the social development to go along with the plan for replacement of the energy system. And clearly, the energy system does not define the social structure (although it may be one factor influencing it), so it may be required to view the emerging energy system in the light of several social development models, in which the same energy system may give rise to different impacts.

6. TOTAL IMPACT CRITERIA

The different methodological elements described in the previous sections may be summarized in a 'checklist' of impacts to be assessed for each energy system. Such a checklist may take the form suggested in Table I, containing a number of criteria with which a given total energy system may be more or less compatible. The table indicates a scale of compatibility ranging from 'highly incompatible' (negatively correlated with) over 'neutral' to 'highly compatible' (positively correlated with). The set of criteria reflect basic individual and social goals, ranging from biological needs over human relations and concern for the environment to broad questions of international relations and potential conflicts. This allows the energy systems to be checked for their degree of consistency with different types of social and political organization. Some of the criteria directly or indirectly affect health, others do not.

Furthermore, some of the criteria reflect issues on which there is a general consensus, while other ones probe into cultural and economic organization of society, in some cases providing antithetic criteria, for which simultaneous compatibility of a given energy system would eventually be excluded.

The 'degree of compatibility' scale is provided as a first attempt at making the assessment quantitative. Only in a few cases can a truly quantitative impact assessment be obtained in comparable units. The idea of the checklist, which does not in its present form claim to be exhaustive, is to avoid unquantifiable aspects of the assessment being left out of the analysis.

Many of the criteria formulated in Table I are subject to interpretation. For example, item 18 (the energy system's compatibility with having high material standards) may include questions on the possibility of launching demand stimulating campaigns (by the energy-providing utility companies or fuel-selling companies), and item 16 (avoiding redundant institutions

TABLE I. CHECKLIST FOR EACH CONTEMPLATED ENERGY SYSTEM

Degree of compatibility with:	(neutral) − 0 +
1. Satisfaction of basic biological needs	+---------+---------+
2. Acceptable health impacts	+---------+---------+
3. Ensuring individual security	+---------+---------+
4. Having meaningful social relations	+---------+---------+
5. Having meaningful work activities	+---------+---------+
6. Acceptable accident risks	+---------+---------+
7. Concern for physical environment	+---------+---------+
Specifically: 7a. Climate impact	+---------+---------+
7b. Air, water and soil	+---------+---------+
7c. Mineral resources	+---------+---------+
7d. Biota, ecosystems	+---------+---------+
8. Concern for work and mental environment	+---------+---------+
9. Goals of a competitive society	+---------+---------+
10. Goals of a society based on equity	+---------+---------+
11. Goals of a non-competitive society	+---------+---------+
12. Emphasis on strong division of work	+---------+---------+
13. Goals of a tradition-based society	+---------+---------+
14. Goals of a pluralistic society	+---------+---------+
15. Encouraging democratic participation	+---------+---------+
16. Avoiding redundant institutionalization and infrastructure	+---------+---------+
17. Avoiding power concentration	+---------+---------+
18. Having high material standards	+---------+---------+
19. Having high non-material standards	+---------+---------+
20. Acceptable energy provision share of economy('acceptable cost of energy')	+---------+---------+
21. Acceptable cost uncertainties	+---------+---------+
22. Resilience of energy supply system	+---------+---------+
23. Political independence	+---------+---------+
24. Keeping future development options open	+---------+---------+
25. Minimizing potential for conflicts and wars	+---------+---------+
26. Improving international relations	+---------+---------+
27. Improving conditions for development in the world's poorest regions	+---------+---------+
28. Overall evaluation being insensitive to uncertainty in individual impact estimates	+---------+---------+

and infrastructure) may include questions of the feasibility of stimulating energy efficiency. In many cases, the energy institutions and infrastructure are barriers against efficiency improvements (e.g. high fixed fees for electricity or district heating provisions, combined with low fees proportional to energy use).

7. A HEALTH IMPACT CASE STUDY

In this section, the difference between system comparison based on equal energy production and, alternatively, on equal energy service delivered will be illustrated by considering occupational health impacts of lighting.[1]

The average U.S. delivery of light amounts to 540 lumen per capita. To provide this an average electrical power input into lamps of 40 W/cap is used [10], corresponding to a primary energy input of about 120 W/cap. The lighting service is likely to be smaller than the lumens delivered from the light bulbs, due to only a fraction penetrating lamp-shades and being directed towards useful areas. Assume that the actual lighting service is 150 lumen/cap, a figure that may be high relative to the light made useful when reaching proper surfaces for reading, working and leisure activities, but a figure which recognizes that also some of the diffuse light from lamps is performing a service in providing a pleasant environment for people.

Table II summarizes the steps involved in providing the lighting service by conventional incandescent light bulbs powered from a coal-fired electricity plant. The table also gives an estimate of health impacts for each step, as provided by [11,12]. The attempt here is to include the manufacture requirements throughout society for each fuel-cycle step, using present (U.S.) industrial structure and bio-medical data. In this sense the health impact evaluation is a 'total' one, but it only includes fatalities and work-days lost as a result of injury, not social impacts and mental injuries. It thus does not provide a complete impact evaluation even for the items 2 and 6 in Table I, and not at all for item 8.

In Table III, a similar evaluation is made for a different energy system, proposing to provide the same lighting service by high-efficiency light-bulbs [10], by improved light-guidance to relevant areas (more transparent shades, more reflecting walls, etc.) and to provide the electric power by wind generators feeding into the existing grid. In order to use the same health impact data as for the coal-based system, a 'marginal system change' approach was used, assuming that the introduction of wind energy does not lead to institutional changes, and that specific back-up facilities would not be required (corresponding to a wind energy share in the system of up to about 25 %, cf. [13]).

Fig.1 compares the occupational impacts of the two systems, on the basis of equal (electric) power production and alternatively on the basis of both providing a lighting service of 150 lumen/cap. Clearly the 'equal power production' comparison gives a completely false impression of the relative impacts.

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[1] If the purpose had been to compare energy sources alone or efficiency improvements alone, only one of these items would have been altered at a time.

TABLE II. HEALTH IMPACT OF PROVIDING LIGHT BY INCANDESCANT BULBS / COAL POWER STATIONS

Step	Average energy input flux	Impacts	
		Injury (10^{-6} MDL)*	Fatality (10^{-9} deaths)
Mining (surface)	154 W	54[a]	12[a]
Processing/transport	139 W	43[a]	9[a]
Power generation,	126 W		
occupational hazard		67[a]	10[a]
public hazard[b]		?	430^{+1720}_{-430}
Environmental protection (desulphurization)		75[a]	14[a]
Power transmission	44 W	~ 200[c]	~ 40[c]
Institutions & back-up facilities		?	?
On-site installations	40 W	?	?
Light bulbs (incandescant)	40 W	5[d]	3[d]
Light distribution	540 lumen	-	-
Lighting service	150 lumen	-	-
Sum of center estimates:		444	518
(of which occupational:)		(444)	(88)

* MDL = Man-days lost.
[a] Ref.[11].
[b] Ref.[12]; assumes low-sulphur coal and 90% desulphurization.
[c] Very rough estimate based on cost of power production and of transmission being similar.
[d] Ref.[12]; assuming cost of (3) bulbs is US $2 and using category of 'control equipment'.

TABLE III. HEALTH IMPACT OF PROVIDING LIGHT BY ADVANCED LIGHT BULBS / WIND POWER

Step	Average energy input flux	Impacts Injury (10^{-6} MDL)*	Fatality (10^{-9} deaths)
Power in wind	12 W		
Power generation	3.9 W		
occupational hazard		29[a]	3[a]
Power transmission	3.9 W	~ 20[c]	~ 4[c]
Institutions & back-up facilities		?	?
On-site installations (incl. transformers)	3.5 W	?	?
Light bulbs (advanced)	3.3 W	35[d]	20[d]
Light distribution (improved)	300 lumen	-	-
Lighting service	150 lumen		
Sum of center estimates:		84	27

* MDL = Man-days lost.
a,b,c As in Table II.
d Assumed cost of advanced light bulb: US $15 (1976 level).

Table II indicates that the coal-based system has a public health impact that is poorly known but which may exceed the occupational hazard in magnitude.

Similar comparisons could be made for alternative ways of providing e.g. domestic hot water (for instance, comparing the cycle 'nuclear power → electricity → resistance heating → hot water → rejected water' with alternative cycles: 'solar collectors → small heat storage → hot water → waste water heat exchanger', etc.) and space heating (for instance, comparing the cycle 'fossil co-generation → direct heating lines → existing building' with: 'solar rooftop collectors → district heating lines → community heat storage → back through district heating lines → retrofitted building with lower heat requirements', etc.).

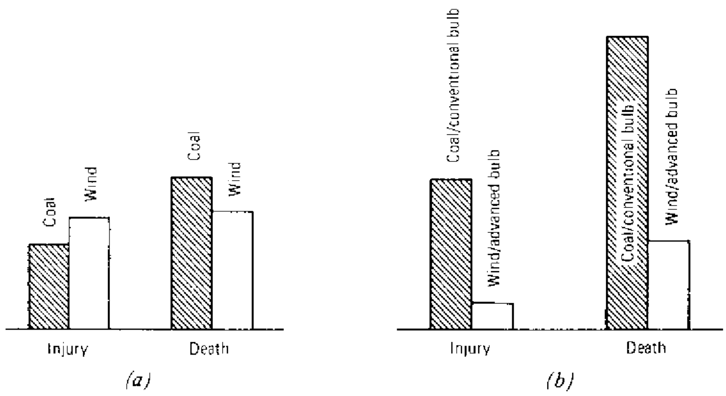

FIG. 1. Comparison of occupational health impacts based on:
(a) equal power production
(b) equal lighting service.

8. CRITIQUE OF PUBLISHED STUDIES

A number of studies have been undertaken to compare the impacts of different energy systems or of different system components. In some studies, cost was the only impact studied, and in other studies, health impacts alone were at issue [14,15,12]. These studies were made on the basis of equal electric power production and they all suffer from the substitution of fairly arbitrary estimates for data not immediately available (with the underlying philosophy that bad numbers are better that no numbers).

Studies attempting to broaden the range of impacts considered, as well as considering alternative ways of a full transition between the present energy system and possible future ones, have tended to restrict most of the impact discussion to a qualitative level [16-19]. Some attempts have been made to systematize the methodology of impact assessment [20,4,7,8]. They have been drawn upon in setting up the set of compatibility criteria described in sect.6.

A few recent studies have moved in the direction of quantitative assessment in some of the impact areas neglected by early comparative studies. Two such studies will be further discussed in this section [21,22]. The range of impact areas covered and the nature of coverage in these two studies are summarized in Table IV. The impact areas of Table I have been lumped together in broad categories for the purpose of Table IV, which gives the relevant item numbers for Table I in parentheses.

TABLE IV. IMPACT TREATMENT OF TWO ENERGY STUDIES

Impact category	study: IIASA[21]	CIS[22]
Biological needs satisfaction (1+3)	-	Q
Health (2+6)	-	-
Environment (7,8)	qQ	m
Social relations (4,5)	-	q
Structure of society (9-19, 23, 24, 28)	m	qQ
Cost (20,21)	Q	mqQ
Resilience (22)	(m)	Q
Global relations (25-27)	mqQ	q

Key to treatment given:
Q = quantitative
q = qualitative
m = mention
– = not considered (or only sporadically mentioned)
Impact categories give items of Table I included in parentheses.

The IIASA study considers two world-demand scenarios, corresponding to different economic development and different allocation of investments. All possible energy sources are invoked, but with emphasis on nuclear and coal technologies [21,23,24]. The CIS study, on the other hand, constructs a scenario for an all-renewable energy system for Sweden, considered to be dictated by a Swedish policy of leaving the best options for a rapid improvement of the situation in the world's poorest countries[22].

Neither study explicitly considers health effects, and the distribution of emphasis on the different impact categories is rather different. While the IIASA study assumes that the present value system will be kept rigidly over the next 50 years, the CIS study explicitly assumes a change in value system towards more emphasis on global solidarity and on placing a ceiling on material consumption in the rich countries. A similar explicit modelling of alternative future value systems were undertaken in the Stanford study [19], and has been suggested in other studies [5,25].

Although Table IV suggests that quantitative indications can be obtained in most of the impact categories, this is clearly not possible for each individual impact item. The importance of unquantifiable impacts have been recognized in earlier studies [26], as has the existence of thresholds in public acceptance of impacts, e.g. an upper threshold beyond which impacts are unacceptable [27].

Satisfaction of biological needs is used in the CIS study to define a minimum of food and energy supply.

The little mention of health impacts in the IIASA and CIS studies is surprising, since the IIASA alternatives rely on nuclear systems with little known but potentially significant risks, in addition to the also little known - but qualitatively well established - health impacts of coal and other fossil technologies, and the CIS study relies heavily on biotechnologies, including forestry and methanol production industries, which may be associated with substantial risks and health impacts.

Environmental issues are discussed in both studies. However, the CIS study is mostly concerned with work and mental environment, while the IIASA study makes a fairly thorough investigation of the possible climate impact of carbon dioxide from fossil fuel combustion (and yet draws the little qualified conclusion that the indicated temperature rise associated with the fossil fuel use in the IIASA scenarios is unlikely to be a problem). The IIASA study also looks into the pollution aspects and the depletion of resources. On the other hand, no discussion of the broad ecological impact of the envisaged futures is undertaken, and the work and mental environment is not mentioned at all. There could be good reason to mention work conditions, e.g. in coal mines and nuclear reprocessing plants, as well as the impact on the mental environment of the security measures, that today are felt necessary in order to deter nuclear terrorism and sabotage. Clearly, the mainly decentralized energy system advocated by CIS would not have serious problems of this kind.

Only the CIS study embarks on a discussion of the impact of energy systems on social relations. Indeed, this is a central theme of the study, which claims that its energy system is particularly suited to a society with social relations better that those prevailing today, and with work conditions similarly improved, by means of a closer link between consumption and production (i.e. production is directly controlled by people's desire for consumption, in contrast to the presently detached production structure, which invites production of some products that could not be sold without heavy promotion).

Actually, the CIS study treats all the questions of the structure of society listed in Table I as items 9-19 and 23-24. It specifically explores the goals of a non-competitive society and attempts to construct an energy supply system compatible with such a society. A quantitative assessment is made of the energy saving resulting from the study's proposed move towards a minimum of institutions and infrastructure. On the other hand, small and decentralized systems are preferred only when they are seen to provide clear advantages over larger and more centralized systems. The proposed energy system ensures political independence and its minimum of institutions and infrastructure ensures that any change in policy can easily be made at a later stage. The CIS study comments that the opposite is true for nuclear power systems.

The IIASA study deals only sporadically with questions of social structure. It does treat the development of material standards (through the gross national product indicator), and claims to have devised energy systems compatible with high material standards. In reality, it has only considered GNP and may in fact represent the trap exposed in the CIS study: creating a growing 'structure' taking up more and more of the total GNP, and thus leaving less and less to contribute to real material standard at the level of people. The IIASA study also discusses political independence, particularly in relation to fuel trade.

Both studies estimate the direct cost of each scenario, and in the IIASA study the principle of minimum direct cost is even used to select the mix of energy sources [24]. The indirect costs do not play a role in the choice of system.

The question of system resilience is only receiving parenthetical attention in the IIASA study, which explicitly assumes a 'surprise-free' world development, with presently decided oil production ceilings as the only 'political constraint'. In contrast, the CIS study makes a detailed analysis of the technology chains for each system component, i.e. it keeps track of where the materials and skills come from and which degree of dependence is involved. The preferred energy systems of the CIS study are described as sometimes 'complex' but never 'complicated', implying a mix of small- and large-size systems, a mix of centralized and decentralized systems, but never systems depending on asymmetric relations between regions. The degree of local rooting may be taken as one resilience measure, other aspects being associated with the impacts that occasional technical failures may have.

As mentioned, a picture of Sweden in the global development process underlies the CIS study. Key criteria are global elimination of poverty, of asymmetric dependence, of resource and environmental exploitation and of the arms race. By relying on its own renewable resources, Sweden may take away some of the global pressures.

The IIASA study also deals with global relations, but in quite different terms. Its central assumption is that economic development of the Third World countries is only possible if there is further economic growth in the rich countries [23]. It is acknowledged that economic growth as defined by GNP does not adequately represent structural changes which may occur in different regions, and that the model used does exclude such changes [24]. It is of course precisely these structural changes that the CIS study attempts to model.

In summary, the look into two fairly detailed energy studies has shown that the full range of impacts suggested in Table I is beginning to receive attention, but that their treatment can be approached in quite different ways. While the CIS study tries to investigate an energy system based on an assumed change in value system and social structure, the IIASA study rigidly adheres to the socio-economic system that happens to prevail at present. The IIASA assumptions are probably shared by most political decision-makers today, while the new value system proposed by the CIS study has a fairly widespread acceptance by the younger generation in many western countries. The question is then whether this generation gets adapted to the views held by the present rulers, as they eventually take over the decision-making responsibilities, or if the new value system will be replacing the old one along with the generation shift.

Regarding those studies which propose to do quantitative comparisons of selected impacts, one should be very careful in drawing conclusions at the present state of the art. In the case of the nuclear energy systems, the most uncertain risk appears to be associated with large reactor accidents. The methodology used by the studies undertaken so far ([14] and related studies, e.g. [28]) has been discredited as incomplete [29,2], and it is not even clear if the distribution of risk on accident size is correctly assessed or not [2]. Furthermore, the impacts associated with large-area land contamination has only recently been devoted detailed attention [30,31].

Similar criticism can be directed at the impacts associated with the air, water and soil pollution from fossil fuel and wood-burning energy systems. The full range of health impacts have hardly been identified, and numerical risk estimates suffer from large uncertainties (cf. e.g. [12]).

More satisfactory is the assessment of occupational risks associated with construction and manufacture processes inherent in energy-system equipment and operation. The problem is here that the calculated impacts are not properties of the energy systems themselves but of the particular industrial methods by which the systems are produced in given regions and at given times. In this respect, the studies are useful in indicating those areas where a change in manufacturing process is called for. But in terms of comparative evaluation of different systems one should be very careful in assessing whether the industrial method underlying given impact figures is essential or not, i.e. if it would be technically and economically justified to use alternative industrial methods with smaller impacts.

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DISCUSSION

L.D. HAMILTON: The health impacts of energy sources represent only one of the factors taken into account in energy policy. This Symposium focuses essentially on quantitiating those health impacts, and that is a necessary precursor to the type of analysis carried out by you, which depends not only on changing the kind of energy used (coal to wind power, for example) but also on increased end-use efficiency (in the form of improved light bulbs, for instance).

B. SØRENSEN: I agree with you that detailed studies for limited impact categories are necessary inputs to a total impact assessment. My reservation was directed at scientists who present the results of limited analyses to the public and decision-makers in such a way that the limitations are not clearly described.

S.C. MORRIS: You say that risk = probability × damage but that for low-probability accidents with high damage the relationship should be risk = probability × $(\text{damage})^p$, raising damage to a power to account for society's lower ability to cope with such accidents. It is my understanding that, in decision and utility theory, the more conventional approach in such cases is to use a multiplying factor rather than a power, that is risk = p × probability × damage (where p is a factor). Could you comment on why you think damage should be raised to a power rather than simply multiplied by a factor?

B. SØRENSEN: My suggestion is that social risk is still proportional to probability but that it increases more than linearly with the amount of damage per event. This becomes clear if you take, for example, a traffic accident causing one death, an aircraft accident causing 100 deaths and a nuclear accident causing 10 000 deaths, and consider society's capability of handling these accidents.

QUANTIFICATION OF THE HEALTH HAZARDS ASSOCIATED WITH DIFFERENT ENERGY SOURCES

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Abstract

QUANTIFICATION OF THE HEALTH HAZARDS ASSOCIATED WITH DIFFERENT ENERGY SOURCES.

Comparisons of health hazards which may result from the operation of different types of electrical power-producing systems are a necessary input to the decision-making process of planning future supplies. Although other factors have played a dominant role in the past and will continue to be a major influence, much greater attention is now devoted to a consideration of detriment to health associated with large-scale industrial development. The paper considers only this health aspect of the comparison and concentrates on one aspect of that, namely on how the impact on health of workers and public can be expressed to represent the detriment. Two measures are discussed: the number of deaths and the effective loss of life, both evaluated per GW(e)·a. The latter is extended along the same lines as in the ICRP publication 'Problems Involved in Developing an Index of Harm'. The index of harm is a measure of hazard to a worker in a particular industry; the analogous quantity here is a measure of hazard of operating a 1-GW(e) power plant. For illustration, the hazards of coal-fired and nuclear power stations are compared although certain factors are omitted from both cycles which it will be essential to include if the method is extended to bring wind, wave and solar energy sources into the comparison. Inevitably some contributions are very difficult to quantify and it may be more realistic to consider these qualitatively rather than attempt to fold them in with artificial numerical values. The procedure described for the inclusion of quantifiable factors seems to be a reasonable basis for comparison but it is not suggested that any such procedure is adequate by itself. Clearly comparisons should be made on several distinct bases.

1. INTRODUCTION

There is global interest in the attainment of an assured energy supply for industry, hospitals, domestic and other requirements. The need is strong for both economic and social reasons. For technical reasons any energy strategy will always be based on several sources but within certain practical bounds the proportion contributed by any one is adjustable. The decisions involved are of a long-term nature since the development and implementation times are long compared with the life of most governments. The factors influencing the decision are economic (the relative costs of producing a unit of electricity), practical (the availability of the energy source) and social (the environmental and health consequences of each possibility). The wider social question of changing society to be less dependent on energy, regarded by some as the ultimate solution, will not be considered in this paper even though there may be associated health consequences; neither will the economic or practical considerations. Here we shall discuss only the implications to health of both workers in the power industry and of members of the public from the exploitation of an energy source. In particular, we shall address ourselves to the problem of the expression of the impact.

The hazards associated with the use of energy sources may be of varied character - there may be short-term minor discomforts or injuries; there may be major injuries or fatal accidents; and there may be long-term health effects such as chronic diseases or late effects. These late effects include death due to induced respiratory or circulatory system damage, malignancies induced by exposure to some carcinogenic agent, and genetic effects. The pattern of these various types of health detriment will be different for each energy source. Therefore, the comparison of the use of different energy sources will not be simple even when restricted to only health considerations. This is complicated still further by the obvious requirement that the whole fuel cycle associated with each source must be brought into any comparison. The pattern of injury in each part of the same fuel cycle will often be distinct and hence the combination of detriment from different parts of a cycle requires some care.

One possibility is to calculate the number of deaths and the number of non-fatal health effects expected to result from the production of a given amount of electrical energy. This may be based on statistics collected on the health of workers involved in a given fuel cycle or on predictive modelling using parameters based on knowledge from experimental research. A fuller assessment would include the distribution of the deaths in time and by age together with a classification of the seriousness of the non-fatal injuries and diseases.

Alternatively, one may seek a measure of impact which embraces the time and age at death such as useful life lost as a result of the processes involved [1, 2, 3, 4]. For fatalities, this has the advantage of providing a simple way of taking account of effects which range from instant death of a young man to a late-effect death which may occur up to 50 years after some exposure. For non-fatal effects, the extent of the disease or injury will be classified according to the period of inhibited activity associated with the effect.

This paper draws on data related to electricity production from the coal fuel cycle and the nuclear fuel cycle. It should be emphasised that while every attempt has been made to find and use actual data, the objective here is to illustrate different representations of health effects rather than to present a definitive comparison of the fuel cycles chosen.

2. METHODOLOGY

In recent years, a number of authors have surveyed the difficulties of objective comparisons of the health consequences of exploiting different energy sources [5, 6, 7]. The difficulties extend beyond those of quantification of effects, about which there is often very little data. Identification of a health-related aspect of a fuel cycle does not bring universal agreement that it should be included in a comparison. For example, some have argued that in the UK the coal burned in power stations is mainly from the larger deposits which are more easily mined and therefore general data on hazards in coal mines over-estimate that part of health impact of coal-fired electricity production. However, this is countered by the argument that if less of these easily mined deposits were consumed in power stations, a smaller output from more hazardous pits would be required for all other coal utilisation. Therefore, it could be argued that fuel extraction for additional coal-fired power stations should be assessed at the level of the most hazardous mines.

Another problem arises when inevitably we have to make comparisons between directly observed risks such as those derived from accident statistics and calculated risks such as those deduced from estimates of population radiation doses and risks per unit absorbed dose extrapolated from observations at doses three or four orders of magnitude higher. There are two facets of this to be taken into account, one being the need to make fair comparisons — that is, not between historical plant of one kind and design specification of future plant of another. The second is the sensible interpretation of the uncertainties involved. Clearly comparisons have to be made even though many pointers indicate the impossibility of being able to do so definitively.

In this paper we shall cut across the kind of difficulties illustrated since our objective focusses on only one aspect of the attempt to compare. ICRP 1977 [1] discusses the establishment of an index of harm for workers which suggests the need to bring together into one index hazards of death (I_D), non-fatal accidents (I_A), industrial diseases (I_S) and hereditary effects (I_H):

$$\text{INDEX OF HARM} \equiv I_D + I_A + I_S + I_H$$

The components are dimensionless and correspond to the number of years of life lost per thousand worker-years. The numerical examples given are based on averages; for example, 30 years life lost from a fatal accident and 15 years from a radiation-induced fatal malignancy. Accidental injuries are assumed to occur at a rate which varies as the square root of the number of deaths in the same industry (justified by considering figures for US industries) and are weighted to be 10% of the period of disability. Occupational disease is considered, but evidence presented suggests that its contribution to harm is small compared with the other hazards and it is included as 5% of accidents, ie. $I_S = 0.05\ (I_D + I_A)$.

Estimate of hereditary damage is considered only for radiation at a rate of 3×10^{-3} per sievert (3×10^{-5} per rem) and weighted equivalent to one death. In addition, exposure to the foetus is included on the assumption that 50% of the workers are female who become pregnant at standard population rates; this adds a sixth of the sum of somatic and genetic effects to the index (see paragraph 63, ref. [1]).

Examples of the value of the index are given below:

(i) Fatal accident rate 300 per million worker-years at 30 years loss of life per death I = 10.2.

(ii) Similar to (i) but rate of 30 per million worker-years I = 1.2.

(iii) Radiation workers in job with fatal accident rate 10 per million worker years, annual exposure of 6×10^{-3}Sv (0.6 rem), I = 1.75.

(iv) Similar to (iii) but exposure which leads to 5×10^{-2} Sv/yr (5 rem/yr), I = 11.5.

Clearly, the numerical estimates for each factor in the index of harm are open to debate and the procedure which leads to the combination of fatal and non-fatal effects is unlikely to find universal acceptance. However, provided the presentation permits the components to be compared separately as well as combined into a single index, the ICRP document [1] does suggest one basis for making comparisons and different values for the weighting of the components may be assigned if a consensus view on their relative importance emerges [5].

In this paper we suggest that, subject to the availability of adequate data, it is unnecessary to make assumptions on average loss of life. Figure 1 shows a comparison of the age distribution of British coal miners and Canadian uranium miners. Obviously a uniform distribution of accident probability with the age of an individual will lead to a different average loss of life. We shall include such effects in our calculations of reduced life expectancy.

Reduced life expectancy in an individual or loss of life years among a group as quoted in sections 3 and 4 are calculated as follows. Normal life expectation is evaluated from age-specific mortality rates in England and Wales as recorded between 1950 and 1978. Life expectancy at age i is defined as

$$L(i) = (n_{i+1} + n_{i+2} + \ldots)/n_i + 0.5$$

where n_i is the number in a population alive at age i, n_{i+1} at age i + 1 etc. Thus a death at age i causes a loss of life of

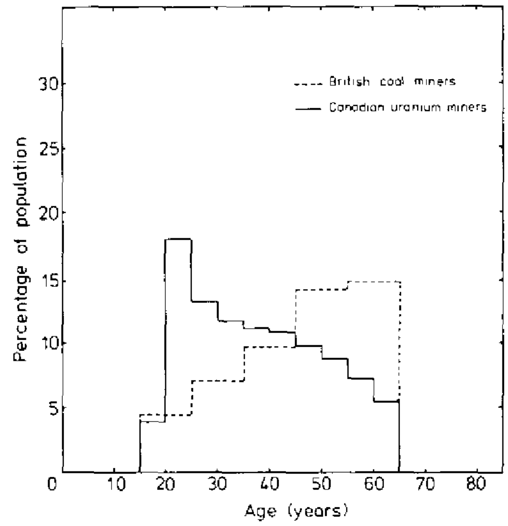

FIG. 1. Comparison of age distribution of two work forces.

L(i) years and total life lost is evaluated by summing the products of the number of deaths at each age and the corresponding life expectancy.

As in reference [1], the probablity that an individual has a fatal accident is assumed to be age-independent between ages 20 and 65. It has been assumed also that national mortality rates apply to working populations with specific job hazards superimposed on them. The method could equally well use occupation-specific mortality rates if they are available.

Life lost due to radiation exposure is calculated in a similar way except that the effects are induced malignancies and death occurs some years after exposure. For a unit dose of radiation we need to know the distribution of time elapse between exposure and death and a value for the risk. Given these, life tables can be generated with and without exposure and life lost derived. ICRP values are used for risks to workers, 1×10^{-2} deaths per Sv (1×10^{-4} per rem) with a distribution of deaths due to solid cancers in time which is zero for 15 years following exposure and then constant until 45 years after exposure. For leukaemia, the corresponding period of finite risk is from 5 to 25 years and it is assumed that 20% of excess deaths are due to leukaemia. For the general public a figure of 1×10^{-2} Sv^{-1} is used for the radiation-induced cancer risk to males and 1.5×10^{-2} Sv^{-1} for females to take account of the greater proportion of women and children who are possibly more sensitive to radiation carcinogenesis than working-age males.

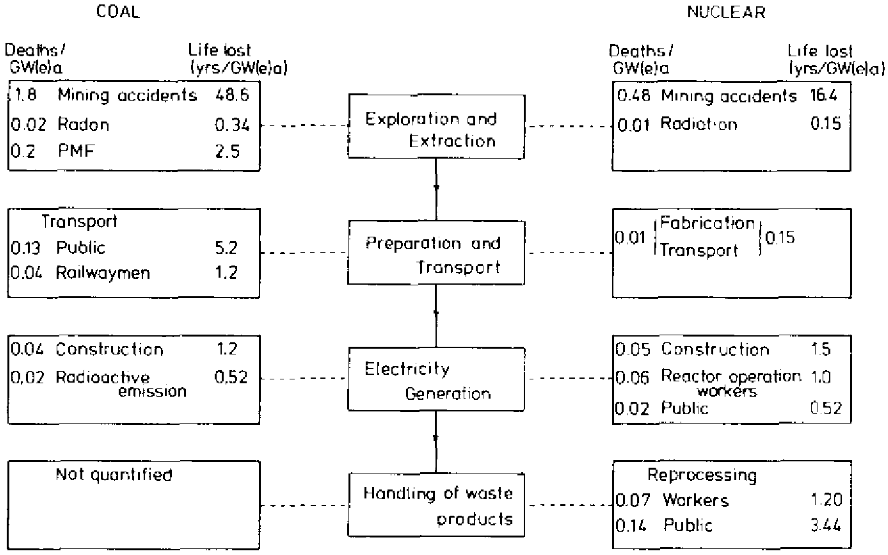

FIG.2. Schematic simplified fuel cycle with hazard data for coal and nuclear fuel cycles. Numbers on the left in each box are deaths per GW(e)·a and those on the right are years of life lost per GW(e)·a. (PMF = progressive massive fibrosis.)

3. FUEL CYCLES FOR COMPARISON

The comparisons to be made are based on the simplified fuel cycle shown in figure 2. The figures shown for the numbers of deaths are per gigawatt electrical year (Gw(e)·a), which seems a reasonable basis for comparison - although some argue that it discriminates against coal since nuclear reactors normally operate on a base load scheme and thus produce electricity closer to their optimum output. However, the output of a coal-fired power station is closely related to the mass of coal consumed and it can be seen from figure 2 that the dominant contribution to health detriment also is dependent on the mass of coal delivered to the power station. Thus given that neither will be used alone to provide all electricity required, expression of effects per GW(e)·a is adequate for our purpose.

The following two subsections comment on the data and their sources.

3.1. Coal

3.1.1. Exploration and Extraction

Exploration for the location of fuel deposits does not represent a significant hazard for coal or nuclear fuel and no

TABLE I. AGE DISTRIBUTION OF POPULATIONS INVOLVED IN THE COMPARISON

POPULATION	AGE GROUP						
	0-14 %	15-24 %	25-34 %	35-44 %	45-54 %	55-64 %	65 + %
British Rail Employees [12]	-	12.2	23.4	23.3	22.6	18.5	-
Coal Miners [12]	-	8.9	14.1	19.4	28.2	29.4	-
Construction Workers [12]	-	14.5	22.5	22.6	20.7	19.7	-
Radiation Workers [1]	-	7.8	23.7	30.7	25.6	12.2	-
Uranium Miners [20]	-	21.8	24.9	21.9	18.6	12.8	-
Public (Males) [13]	25.1	15.1	13.0	12.1	12.5	11.7	10.5
Public (Females) [13]	22.5	13.9	11.9	11.3	12.2	12.2	16.0

attempt has been made to include it here. Similarly the initial mine construction is a negligible hazard when taken over the lifetime output of the mine. Using information given by the UK National Coal Board (NCB) [10] for the years 1970-1980 (the coal output per man-shift is 2.23 tonnes; there are 1.06 deaths per million man-shifts and 1 GW(e)·a requires 3.85 million tonnes) yields a figure of 1.8 deaths per GW(e)·a. Conditions and safety in mines have improved during the decade so future mining operations should reduce this rate.

Coal-miners are subjected to some increase in γ-radiation (perhaps up to 0.2 mGy/a) but they have a corresponding decrease of natural cosmic irradiation. Radon exposure of miners is estimated to cause about 0.02 deaths per GW(e)·a from induced lung cancers [8, 9].

The reduced life expectancy (see section 2) among the miner population in Table I corresponds to 27 years life lost per fatal accident. According to NCB reports, the mean age of miners has decreased from 43.9 years in 1970/71 to 39.6 years in 1979/80. Thus the life lost per accident will be an underestimate of future values but since the figure of 1.8 deaths per GW(e)·a may overestimate future mining hazards, the figure of (27 x 1.8 =) 48.6 years lost per GW(e)·a may be regarded as a fair representation. An additional 0.34 years per GW(e)·a may be associated with radon.

Dust conditions in mines have led to high incidences of pneumoconiosis and the more extreme condition, progressive massive fibrosis (PMF). Considerable advances have been made in controlling dust levels in modern mines which will reduce such disease among miners. This changing situation, together with the difficulty in assessing if the condition caused or contributed to death, does not permit a reliable quantification. However, a figure has been derived for present purposes based on modern conditions of 0.2 deaths and 2.5 years of life lost per GW(e)·a. An explanation of these figures is given in the appendix.

3.1.2. Preparation and Transport

Most preparation and sorting of coal for power stations is carried out before the fuel leaves the colliery, and accidents are included in the figure for mining operations. Transport of coal from mine to power station is almost entirely by rail and this results in 0.04 deaths per GW(e)·a among British Rail employees and 0.13 deaths per GW(e)·a among the general public (omitting suicide) [5, 11].

Translating these numbers of deaths into life lost among British Rail workers (Table I) yields 30.8 years of life lost per fatal accident or (0.04 x 30.8 =) 1.2 years lost per GW(e)·a.

Information on railway accidents [14, 15] does not provide an adequate age distribution; here it has been assumed uniform up to age 80 years and zero thereafter. 83% of the fatal accidents on railways are to males (public and railway workers), which corresponds to about 75% males and 25% females for the general public. Life lost is 39.2 years per male accident and 41.9 years per female accident. Thus we have (0.13 x (0.75 x 39.2 + 0.25 x 41.9) = 5.2 years lost per GW(e)·a among members of the public.

3.1.3. Electricity Generation

Inhaber [3] gives construction times for coal and nuclear power stations in terms of man-hours per MW(e)·a over the systems operational life (coal 505 and nuclear 633 man-hours/MW(e)·a). However, the bases of these have been disputed [5] on grounds as to the extent to which worker and public risks arising from the production of materials used in the construction should be included in the health assessment. This is not important when considering coal or nuclear but can dominate for wind, wave or solar energy technology. Inhaber's figures correspond to about 7 500 man-years work for a 1-GW(e) coal-fired power station and 10 000 man-years for the same size nuclear plant. This seems not unreasonable and a fatal accident rate among construction workers of 160 per million per year [1, 19] leads to 0.04 deaths per GW(e)·a. Using the age distribution for construction workers in Table I yields the life lost per accident as 30.8 years and hence (0.04 x 30.8 =) 1.2 years lost per GW(e)·a.

A coal-fired power station may discharge to the atmosphere products of incomplete combustion (for example, polycyclic aromatic hydrocarbons), toxic metals or pollutants such as SO_2, particulates and oxides of nitrogen. While these are potentially harmful, effects have not been observed at permitted levels and such hazards are not quantified here, but general considerations are available in the literature [5, 16, 17]. In addition, coal burning does discharge radioactive material into the environment [18, 19]. This has been considered in some detail in other papers in these proceedings (session III) but the information was not available for inclusion in the estimates here. Camplin estimates that the collective effective dose equivalent commitment (truncated at 500 years) to the UK population is 340 man-Sv from 30 years of operation of a modern coal-fired power plant - taking into account a negative contribution due to the release of aged carbon. A representative figure is 10 man-Sv per GW(e)·a to the UK population, which is in reasonable agreement with Corbett [8]. This figure becomes 3 man-Sv per GW(e)·a on a global basis since the stable CO_2 negative contribution increases more rapidly than the positive contribution from all other radionuclides. Use of the UK population distribution given in Table I will yield only approximate global effects since life expectancy in UK is considerably greater than the world average, but the uncertainty on these numbers is large since the estimates are arrived at as the difference between two large numbers. UNSCEAR are in the process of compiling data of this type; in the meantime 3 man-Sv per GW(e)·a is used which leads to 0.02 deaths and 0.52 years lost per GW(e)·a.

3.1.4. Handling of Waste Products

Corbett [8] has reported that concentration of radionuclides in coal are similar to those in soil and that burning concentrates them in the ash by about a factor of five. Bulk ash is used as land fill and in man-made building materials. The former results in negligible increases in radiation doses considered over 500 years while criteria for protection against radiation effects from building materials can be drawn up independently of the fuel cycle generating the waste. Other aspects, such as leaching out of corrosive material from ash tips into ground water, have also been ignored.

3.2. Nuclear

3.2.1. Exploration and Extraction

As with coal, exploration for uranium deposits is not included but mining hazards include two components: fatal accidents and significant radon exposure. Hamilton [7] quotes 0.48 deaths per GW(e)·a for Canadian uranium mines. The population distribution in Table I yields 34.2 years life lost per accident and hence (0.48 x 34.2 =) 16.4 years lost per GW(e)·a. UNSCEAR [21] appendix E, para. 81, gives 1 man-Sv per GW(e)·a as the collective lung dose to uranium miners. External irradiation during mining is given as about 0.5 man-Sv per GW(e)·a (para. 48). The public are exposed if mine waste is used for building purposes but since the decision to do so is independent of energy production, effects of such exposure are not included here. The lung dose and the external radiation to uranium miners correspond to 0.01 deaths per GW(e)a or to 0.15 years life lost per GW(e)·a on the basis described in section 2.

3.2.2. Preparation and Transport

Milling, fuel fabrication and transport lead to 1.5 man-Sv per GW(e)·a [21, 22] with negligible contribution from fatal accidents during transport since the volumes being transferred are small relative to the coal equivalent. This collective dose leads to an estimated 0.01 deaths per GW(e)·a, corresponding to 0.15 years life lost.

3.2.3. Electricity Generation

UNSCEAR [21] assess the collective dose per GW(e)·a due to reactor operations as 10 man-Sv to workers and about 3 man-Sv to the public. Using the relevant risks and age distribution, these lead to 0.06 deaths (1.0 years life lost) per GW(e)·a for workers and 0.02 deaths (0.52 years life lost) per GW(e)·a for the general population. Construction adds 0.05 deaths and 1.5 years lost per GW(e)·a.

3.2.4. Handling of Waste Products

Reprocessing spent fuel to separate plutonium and uranium from fission and activation products is the largest contributor to the collective doses which result from the nuclear fuel cycle. UNSCEAR estimate these from UK experience to be in the range 12 to 39 man-Sv to the world population and 12 man-Sv to workers per GW(e)·a. These are based on historical data, and UNSCEAR suggests that future doses are likely to be lower than these - here we have used 20 man-Sv to the public and 12 man-Sv to workers, which leads respectively to 0.14 deaths (3.44 years life lost) per GW(e)·a and 0.07 deaths (1.2 years life lost) per GW(e)·a.

Following UNSCEAR no assessment is made of doses that may result from future disposal of high active waste.

4. SUMMARY OF COMPARISON

Figure 2 summarises the components included to illustrate the comparison procedure. The values for life lost are analogous to the index of harm [1] outlined in section 2. Index of harm measures the hazard to a worker in a particular industry - the quantity here is a measure of the health detriment of operating a 1-GW(e) plant. The index of detriment is the sum of the components and this is summarised together with the number of deaths in Table II. Also shown in this table is the modification to the index to include non-fatal accidents, occupational diseases, hereditary effects and radiation dose to the foetus. These are incorporated by the same scheme adopted in the index of harm.

TABLE II. COMBINED ESTIMATES OF HAZARD PER GW(e)·a TAKEN FROM FIG.2

	Coal	Nuclear
Deaths	2.25	0.84
Life lost due to deaths (years)	59.6	24.4
Index of detriment of 1 GW(e) power station	65	29

NOTE: The 'index' includes modifications described in Section 2 to take account of non-fatal accidents, occupational disease, genetic effects and effects associated with irradiation during pregnancy.

We see from Table II that the comparison between coal and nuclear-produced electricity looks the same whichever of the three measures of health impact is considered. This will not necessarily be so for all comparisons and occurs here because of the young ages of Canadian miners and the large estimated dose to the public from reprocessing, both of which contribute a high value for the life lost per death. Although all types of effects for both public and workers are combined into a single index, the authors would like to emphasise that this is exploratory only and that it is essential to look at each component. The comparison here is incomplete and therefore is to be regarded only as illustrative of a method for comparison which can be applied in more detail and to the exploitation of any source of energy.

ACKNOWLEDGEMENTS

The authors are grateful to Linda Butler for her rapid production of the manuscript of this paper and to Mr. J. Buckley for the diagrams. We are also grateful to Dr. M. Jacobsen for making his data on PMF available to us and to Drs M. Jacobsen and S. Rae for helpful discussions. However, responsibility for the crude approximations made here rests with the present authors.

APPENDIX

Pneumoconiosis incidence and life shortening under modern coal-mining conditions

The median level of dust in modern British coal mines is about 3.7 $mg.m^{-3}$ [23]. The work of Jacobsen et al. [24] suggests that 2% of workers exposed to this level will progress to category 2 simple pneumoconiosis or higher in 35 years. McLintock et al.

[25] report that the attack rate of progressive massive fibrosis (PMF) on workers with category 2 pneumoconiosis is 1.5% per annum (based on a 5-year study). Although it is clearly an oversimplification, we estimate the incidence of PMF in miners by combining these figures, assuming that the annual risk of progressing from simple pneumoconiosis to PMF continues for 15 years. This combined risk is thus 0.45%.

Estimates of life shortening due to PMF are based on the 24-year survival figures for workers of various ages and degrees of pneumoconiosis (private communication with M. Jacobsen). The data for men without pneumoconiosis was reproduced quite well by using national age-specific mortality rates. The effect of PMF was modelled by adding an increment to the death rates for the ages above 50. The loss of life due to a death from PMF is taken to be the difference between the life expectancies of 20-year-olds from the two populations, 3.4 years.

Taking a figure of 6500 man-years to mine the coal to produce one GW(e)·a, the total number of deaths is 0.2 and the total life lost 2.5 years.

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DISCUSSION

B. SØRENSEN: You give your results with three significant digits and no error estimate. Does this mean you know the risk from major accidents to three significant figures?

J.A. REISSLAND: Only two or three of the higher values are quoted to three significant figures. But your implication is in order: such accuracy is not justified, nor is the use of two significant figures in the smaller contributions.

However, one significant figure gives misleading results when the numbers are combined. You will notice that we have quoted the final results only as integers.

B. SØRENSEN: You discuss the concept of 'useful life lost'. Is the loss of life of unemployed or retired persons not considered?

J.A. REISSLAND: The term 'useful life lost' refers to non-fatal detriment such as three months of debilitating illness. There is no judgement involved about the relative usefulness of lives. All life is included in the assessment which I outlined, whether the detriment occurs during normal working life, other types of employment, unemployment, or at any age during retirement. The loss of life is the period of inability to lead a normal life (in the case of illness), or the period from the age of death covering the expected further life for a person of that age (in the case of death).

I.M. TORRENS: Have you done any work on the inclusion of the more efficient use of electricity (through better light bulbs or heat pumps, for example) in the fuel cycle risks?

J.A. REISSLAND: No. It is unclear to me why the more efficient systems discussed in the previous paper (SM-254/105) by Dr. Sørensen cannot be used irrespective of the method of electricity generation.

R.M. BARKHUDAROV: I think you said that your values for loss of life were based on numerical values from ICRP Publication 27. I should like to know whether you relied entirely on the figures in that publication or whether you in fact have your own values for loss of life due to various causes.

J.A. REISSLAND: No numerical values were taken from ICRP Publication 27 in calculating the index of hazard. We used the principles outlined in that publication to estimate effects, but not the numbers. All values for loss of life expectancy were calculated on the basis of the actual age distribution of the relevant working population. The source data for risks are given in the references quoted.

W. PASKIEVICI: You indicated that the number of deaths per GW(e)·a due to radon is greater in coal-mining than in uranium-mining. How did you calculate the effect of radon in coal-mining?

J.A. REISSLAND: The references in the paper will make this clear, but I think I should emphasize that these values are per GW(e)·a and not per worker. In other words, the risk to an individual worker from radon in coal mines will be less than that for a uranium miner, but more time is spent mining the equivalent amount of coal.

DIRECT AND INDIRECT HEALTH AND SAFETY IMPACTS OF ELECTRICAL GENERATION OPTIONS

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Abstract

DIRECT AND INDIRECT HEALTH AND SAFETY IMPACTS OF ELECTRICAL GENERATION OPTIONS.

This report is an analysis of the health and safety risks of seven electrical generation systems, all of which have the potential for commercial availability after the year 2000. The systems are compared on the basis of expected public and occupational deaths and lost workdays associated with average unit generation of 1000 (MW(e) per year. The risks and associated uncertainties are estimated for all phases of the energy production cycle, including fuel extraction and processing, on-site construction and system operation and maintenance. Also included are the risks of direct and indirect component manufacture, materials production and energy inputs, all of which are major contributors to the risks of the more capital-intensive solar technologies. The potential significance of major health and safety issues that remain largely unquantifiable are also considered.

1. INTRODUCTION

This report summarizes the results of an assessment comparing the health and safety risks of seven electrical generation technologies. The assessment was performed at Argonne National Laboratory under the auspices of the Satellite Power System (SPS) Concept Development and Evaluation program established by the Department of Energy and the National Aeronautics and Space Administration to generate information by which a decision could be made regarding the development of an SPS. The seven technologies chosen for assessment are projected to be commercially viable within the time frame for implementation of the SPS (2000-2030). The technologies include the SPS; a low-Btu coal gasification system with an open-cycle gas turbine combined with a steam topping cycle (CG/CC); a light water fission reactor system without fuel reprocessing (LWR); a liquid-metal fast-breeder fission reactor system (LMFBR); a central-station terrestrial photovoltaic system (CTPV); and a first-generation fusion system with magnetic confinement. For comparison with the baseload technologies, the risk from a

TABLE I. MAJOR CHARACTERISTICS OF THE SEVEN ENERGY TECHNOLOGIES

Characteristic	LWR	CG/CC	LMFBR	CTPV	DTPV	SPS	Fusion
Unit Capacity (MW)	1250	1250	1250	200	0.006	5000	1320
Total Direct Commodity Cost per Unit ($\$10^6$)[a]	333.7	356.2	535.2	90.1	0.00717	13 421	1253.8
Average Annual Load Factor (%)	70	70	70	25.8	12.2	90	70
Indirect Capital Cost per Unit ($\$10^6$)[b]	197.1	132.7	262.6	20.0	-	-	628.6
On-site Construction Labor per Unit (10^6 person-hr)	13.1	15.2	14.5	1.7	96×10^{-6}	(c)	22.1

[a]Delivered costs for components, structures and materials. Land and labor costs excluded. Values are 1978 dollars.

[b]Temporary site construction facilities, payroll insurance and taxes, and other construction services such as home and field office expenses, field job supervision, and engineering services. Specifically excluded are fees for permits, taxes, interest on capital, and price escalation. Values are 1978 dollars.

[c]Rectenna - 15.0; Construction in orbit - 0.7; Launch area maintenance - 4.2; Launch area operations - 2.8.

decentralized 'roof-top' photovoltaic system with 6 kW(e) peak capacity and battery storage (DTPV) was also evaluated. The basic design parameters for these systems are given in Table I.

Detailed descriptions of the alternative generation systems were compiled on a consistent basis for comparison [1, 2]. The design of the coal system with low-Btu gasification was based on an SO_2 emission factor of 86 kg $SO_2/10^{12}$ J for gas or 140 kg $SO_2/10^{12}$ J for coal. The light water reactor considered was that of a typical U.S. commercial design using enriched uranium without reprocessing. The fusion system was based on a preliminary design utilizing a tokamak reactor with deuterium/tritium fuel cycle. Silicon photovoltaic cells, at an array cost of $\$35/m^2$, were assigned to each of the solar energy systems. The design of the decentralized solar energy system included 20 kWh(e) of storage capacity and advanced lead-acid batteries with a 10-year lifetime, although system storage and utility system back-up were not included for any of the other systems.

2. ESTIMATION OF HEALTH AND SAFETY RISK

From the technology characterizations and other related information, we identified all major known and potential health and safety issues that could be unambiguously defined and discussed. Each segment of the energy cycle was considered, including component fabrication, plant construction, fuel extraction and processing, operation and maintenance, and waste disposal.

2.1. Direct effects

Public and occupational risks directly related to construction, operation and maintenance of each energy cycle were estimated primarily on the basis of available literature. (See reference [3] for full literature citation.) Whenever possible, a quantitative estimate of fatalities and person-days lost was made. While these measures do not define the total adverse impact of a health and safety issue, they do provide a means of comparing the technologies as a whole and by phase of the energy cycle.

A range of estimated impact is included in each quantification, reflecting the uncertainty associated with the magnitude of impact. However, for some potential health and safety issues it was not possible to provide any quantification. Lack of information such as dose-response relationships at low-dose levels, siting patterns, populations exposed, uncertainties regarding probability of event occurrence, and characterizations of advanced technologies limited the estimation of risk magnitudes for some issues to qualitative discussion of potential severity or possible mechanisms for occurrence of the risk.

The risk of electricity generation differs between technologies not only in the magnitude but also in the manner in which the impact is incurred. These distinctions affect societal perceptions of the acceptability of each risk and need to be preserved in the analysis. Catastrophic events constitute a prime example of the need for categorization. Because of the engineered low risk of occurrence of catastrophic events, the number of expected deaths per year, averaged over the lifetime of the plant, may be lower than that from continuous low-impact risks, but the public perception of the significance of these potential events may critically affect the viability of a technology. For this reason, plus the inherent difficulty in predicting occurrence rate and impact level, catastrophic events were not included in the quantified sum of technology risks in this study but were included in a separate semiquantitative discussion.

2.2. Indirect effects

Compared to the more conventional coal and fission technologies, the advanced solar and fusion technologies present

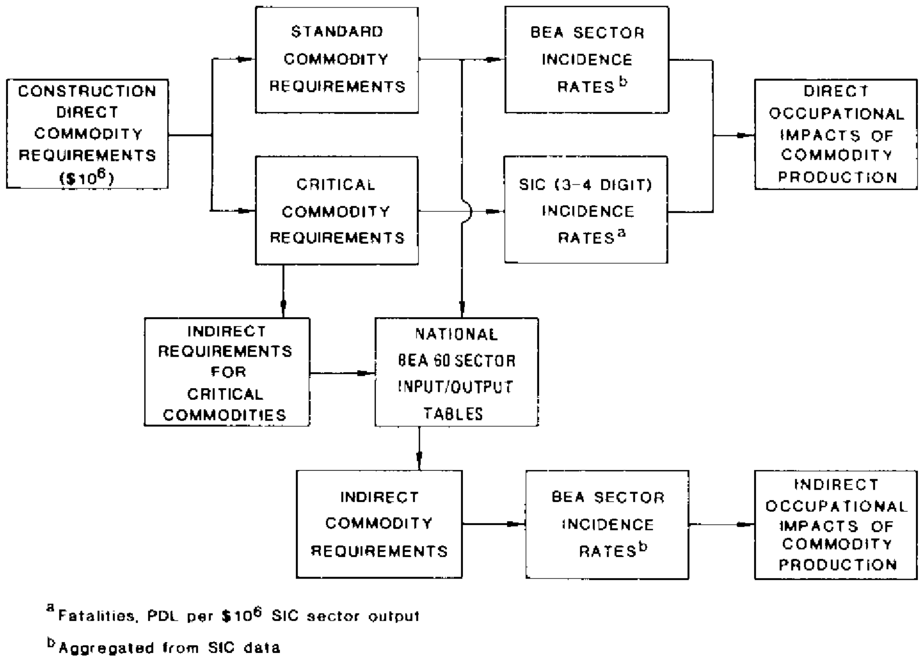

FIG. 1. Analysis of occupational health and safety impacts (PDL = person-days lost).

a tradeoff of reduced fuel requirements but higher initial capital and construction requirements. Furthermore, the industries producing the energy system components in turn require certain commodity inputs (e.g. copper mining to produce electrical equipment), and the risks associated with the production of these indirect requirements must be considered in the overall risk analysis.

The procedure for estimating occupational health and safety risks associated with off-site manufacturing of components and materials used in facility construction (Fig. 1) entailed use of the percentage of system capital costs for commodities produced by the industrial groupings specified by the U.S. Department of Commerce Bureau of Economic Analysis (BEA). The indirect commodity requirements in all BEA sectors are determined by economic input-output analysis with the direct commodity requirements as input [4]. The coefficients for fatalities and nonfatal person-days lost per $\$10^6$ output for each of the BEA categories were averaged from the more narrowly defined categories of the U.S. Standard Industrial Classification (SIC). The total 1975 output of the SIC sector [5,6] was used as a weighting factor. The parameters used to obtain the SIC sector incidence rates per $\$10^6$ output were the 1972 productivity levels (employees/$\$10^6$) [5] and the 1975 statistics for ocupational health and safety risk [7] per employee-year.

Similar procedures were utilized to estimate indirect energy requirements in the various industrial sectors that produce the system components.

Because the categories in the BEA grouping cover a broad range, the risks of production of major commodity requirements of individual energy technologies may be inaccurately estimated by the procedure described above. For this reason, the production risks of specific critical commodities were evaluated with the slightly revised procedure indicated in Fig. 1. For these critical commodities, the direct production risks were evaluated by means of more specific incidence rates for SIC categories. Manufacture of photovoltaic cells for the solar energy systems and of lead-acid storage batteries required in the DTPV system are examples of commodities evaluated with this more detailed approach. Risks, both direct and indirect, associated with producing inputs to the manufacture of these critical commodities (e.g. lead for storage batteries) were then evaluated by means of the broader BEA categories.

The procedure for estimating direct and indirect occupational risks of commodity production contains various uncertainties, including use of the Bureau of Labor Statistics data for occupational injury and illness [7]. Although these data are considered the best for these factors, they reflect large error bounds because of underreporting and misdiagnosis. In particular, these statistics do not adequately reflect chronic disease. Other uncertainties include use of the historic input-output structure of the economy to estimate indirect requirements for facilities to be constructed after 2000, uncertainties in plant construction requirements, potential changes in employee productivity, and potential changes in risk levels per worker.

3. RESULTS

A summary of estimated fatalities per year per 1000 MW(e) of average generation is given in Table II.

3.1 Risks of the construction phase

For every $\$10^6$ of direct industrial output required to supply system components to each of the energy systems considered, a combined indirect output of $\$0.5–0.9 \times 10^6$ is required

TABLE II. SUMMARY OF QUANTIFIED AVERAGE FATALITIES PER YEAR PER 1000-MW GENERATION, 30-YEAR PLANT LIFETIME

Characteristic	LWR	Coal (CG/CC)	LMFBR	CTPV	DTPV	SPS	Fusion
Total	0.26-1.4	6.6-79	0.24-1.1	0.43-0.73	1.92-4.4	0.26-0.67	0.22-0.44
Population Affected							
Public	0.03-0.18	5.4-76	0.03-0.18	U[a]	U	U	0.0001
Occupational	0.24-1.2	1.3-3.1	0.21-0.94	0.43-0.73	1.92-4.39	0.26-0.67	0.22-0.44
Impact Period							
Manufacture and Construction[b]	0.10-0.16	0.11-0.18	0.12-0.20	0.31-0.55	1.04-1.94	0.19-0.55	0.16-0.38
Operation and Maintenance	0.16-1.2	6.5-79	0.12-0.92	0.12-0.18	0.88-2.45	0.07-0.12	0.03-0.06
Impact Cause							
Accidents and Non-Radiation Disease	0.21-0.67	6.6-79	0.17-0.51	0.43-0.73	1.9-4.4	0.26-0.67	0.22-0.44
Radiation	0.05-0.70	0.0023	0.07-0.61	U	U	U	U

[a]U - Unknown or negligible.

[b]Total impacts averaged over 30-year lifetime.

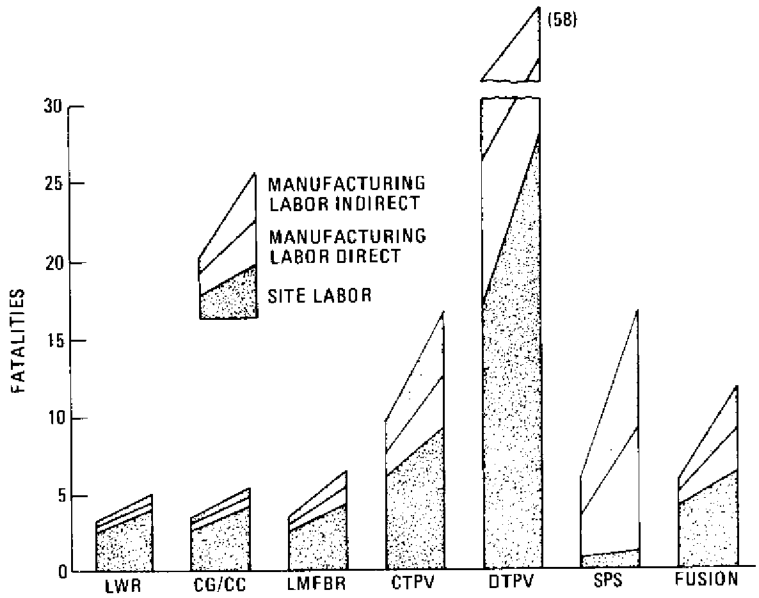

FIG. 2. Total occupational fatalities in construction phase of system with 1000 MW(e) average generation.

TABLE III. INDIRECT ENERGY REQUIREMENTS AND RELATED HEALTH AND SAFETY RISKS FOR ENERGY SYSTEM CONSTRUCTION[a]

Energy System	Input Energy		Associated Risk (Fatalities)		
	Electrical [1000 MW(e)-yr]	Coal (10^{15} J)[b]	Coal Electric[c]	Nuclear Electric[d]	Coal Mining[e]
Coal Gasification Combined Cycle	0.041	0.9	1.6	0.03	0.4
Light Water Reactor	0.040	1.0	1.6	0.03	0.4
Centralized Terrestrial Photovoltaic	2.8	6.2	110	2.2	2.6
Decentralized Terrestrial Photovoltaic	6.5	9.5	260	5.2	3.9

[a]For number of systems defined in Table I required to generate 1000 MW(e) average.

[b]Coal for purposes other than electrical generation.

[c]Based on mean total risk of 40 fatalities/1000 MW(e)-yr from Table II.

[d]Based on mean total risk of 0.8 fatalities/1000 MW(e)-yr from Table II.

[e]Based on 0.41 fatal accident and disease incidence per 10^{15} J of coal extracted in underground mining.

from other industries. The combined direct and indirect impact per $\$10^6$ of component production is within the same range for each technology; as a result, the total component requirement per 1000 MW(e) of generation is an overriding factor in determining component production risk. The total occupational risks of component production, combined with on-site construction risk, are illustrated in Fig. 2. As the figure shows, higher construction-phase risks are incurred in the solar technologies because they are more capital intensive. This is most clearly demonstrated by the decentralized solar technologies, which require a large number of small dispersed facilities to generate an average of 1000 MW(e) annually.

The production of energy utilized in component manufacturing represents a further indirect risk not included in Table II or Fig. 2. Input-output analysis indicates that the total electrical energy requirement for direct and indirect component manufacture for the coal and nuclear systems is equivalent to a small fraction of the equivalent energy produced in one year of operation [1000 MW(e)-yr] of those systems (Table III). On the other hand, the input energy for component manufacture for the centralized and decentralized photovoltaic systems is equivalent to 2.8 and 6.5 years of output, respectively. These large 'payback time' estimates are in large part due to the electrical energy requirements for production of silicon photovoltaic cells [2360 kW(e)h/kg semiconductor-grade silicon] [8]. The risks associated with the production of this quantity of electrical energy are highly dependent on the generation technology assumed. The mining of coal utilized in component manufacture for purposes other than electrical generation (Table III) does not represent major additional risks.

3.2. Occupational risks of the operation and maintenance phase

The occupational risks of operation and maintenance (O&M) are largest for the coal technology, primarily because of the risk of accidents and illness from coal mining. A major uncertainty in mining risk estimates derives from uncertainty about the long-term effect of recent regulations for reducing accidents and the levels of dust in coal mines. Additional occupational O&M risks are related to rail transport of coal, accidents in the coal-processing and electrical-generating plants, and exposure to potentially carcinogenic emissions from the coal gasification process. The estimate of the risk from in-plant gasification emissions [0.0 - 0.2 fatality/1000 MW(e)/yr] is based on the estimated number of workers in the plant and historical data from pilot plants with limited control measures [9]. For the fission systems, approximately 70-80% of the risk is related to conventional occupational hazards, and the remaining 20-30% is due to low-level radiation exposure, the impacts of which are uncertain. The O&M occupational risks of the advanced fusion, SPS, and centralized terrestrial solar systems have no historical basis and are projected from conventional risk levels for existing similar occupations and from estimates of the number of O&M employees required [1,2]. The significant O&M occupational risk estimated for the decentralized solar energy system is based on 3-9 hours annual maintenance for each of the numerous small units, plus replacement of storage batteries every 10 years.

3.3. Public risks of the operation and maintenance phase

The public risks of the O&M phase are also largest for the coal technology, and these risks are almost entirely due to coal transport accidents [0.8 - 1.9 fatality/1000 MW(e)/yr] and air pollutants [4.6 - 75 fatalities/1000 MW(e)/yr]. The air pollutant impacts include those from long-range transport, and the uncertainty range is based on a 60%-confidence level for estimating the incidence rates of health effects (adapted from Ref. 10). It should be noted that a similar procedure with 90% confidence levels results in a lower limit of zero impacts. For the fission and fusion systems, only low levels of public impacts [less than 0.1 fatality/1000 MW(e)/yr] can be attributed to normal O&M, and these are primarily due to low-level radiation, which has a high uncertainty level. The quantified public impacts of the O&M phase are negligible for the solar technologies.

3.4. Unquantified health and safety issues

In contrast to the apparent public willingness to accept limited known risks of energy systems, recent experience with light water fission systems indicates that perceived major risks that are less quantifiable or predictable may restrict or completely halt energy system deployment, if adequate assurances of very low impact probability cannot be given. For this reason, potentially major but unquantified risks should be given prominence comparable to the quantified risks discussed here. Table IV is a listing of potentially major unquantified issues identified for the seven technologies considered. Catastrophic events (i.e. events of low occurrence probability but high impact per event) are included in the unquantified category because of the inherent difficulty in predicting occurrence rate and impact level. Furthermore, averaging expected catastrophic impacts over plant lifetime does not indicate the full significance of these potential events.

TABLE IV. POTENTIALLY MAJOR UNQUANTIFIED ISSUES

Solar Technologies (CTPV, DTPV, SPS)	Nuclear Technologies (LWR, LMFBR, Fusion)
1. Exposure to cell production emissions	1. System failure with major public radiation exposure
2. Hazardous waste from disposal or recycling of cell materials	2. Fuel-cycle occupational exposure to chemically toxic materials
3. Chronic low-level microwave exposure of large populations (SPS only)	3. Diversion of fuel or by-product for military or subversive uses (LWR, LMFBR only)
4. Space vehicle crash into urban area (SPS only)	4. Liquid-metal fire (LMFBR, fusion only)
5. Exposure to launch-vehicle emissions (SPS only)	

In general, the more defined technologies (e.g. CG/CC, LWR) have a greater number of quantifiable risks and fewer unquantifiable risks. The opposite is true for the less-defined technologies (e.g. fusion, SPS). Table IV does not attempt to rank the unquantified issues, although, for example, potential radiation release from fission is expected to be greater than that from fusion.

4. CONCLUSIONS

Of the various systems considered, the coal technology has the largest overall quantified risk, primarily due to coal extraction, processing and transport, and air-pollutant emissions, although large uncertainties remain in the actual effect of the air pollution. The decentralized photovoltaic system has large associated risks due to the large labor and material requirements of small, dispersed units. The quantified risks from the remaining technologies (fission, fusion, SPS and centralized terrestrial photovoltaic) are comparable, within the range of quantified uncertainty. The occupational risks for component production, both direct and indirect, are a substantial fraction of the total risks; in particular, for the advanced, capital-intensive solar and fusion technologies. The energy requirements for component production can also be associated with substantial risks, depending on the source of energy.

Of potential major significance for public acceptance of new energy systems, but not included in the quantification, is the possibility of catastrophic incidents associated with fission and fusion systems. Unique unquantified issues of concern also exist for the SPS in relation to the use of microwave transmission of energy and extensive space travel.

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DISCUSSION

J.A. REISSLAND: Thank you for an interesting paper, which perhaps dealt more comprehensively than any other presentation with the central theme of this Symposium. I should like to clarify two points. First, one of the slides you showed in your oral presentation indicated the non-fatal days lost per million dollars of equipment. What is the purpose of that index? It would seem to produce a smaller value if equipment costs were increased. Secondly, you used a 60% confidence range, which would correspond to an unrealistically low range to represent the uncertainty. Why did you choose that figure?

L.J. HABEGGER: The slide in question illustrates the comparable hazard of component production on a unit cost basis. These values are multiplied by the capital costs of the respective technologies to obtain widely varying hazards on a unit electrical energy production basis (1000 MW(e), average). I should like to refer your second question to Mr. Morris of Brookhaven National Laboratory, as the 60% confidence range is based on work by that laboratory.

S.C. MORRIS: At Brookhaven, we have estimated the air pollution health risk as a probability distribution reflecting the level of existing knowledge. Using a 60% confidence range from this distribution is similar to stating the results as the mean ± slightly less than 1 standard error.

Current knowledge is inadequate for us to state with high confidence that exposure to low-level air pollution has no health effects whatsoever. This does not mean the best estimate of effects is zero, however. The role of the assessment is not to prove the existence of effects but to estimate their magnitude with some expression of uncertainty.

P.D. MOSKOWITZ: How did you calculate and compound the uncertainty for the estimates you presented?

L.J. HABEGGER: Most of the uncertainties were based on our interpretation of the range of estimates provided in the literature, with the exception of the uncertainty in effects from atmospheric emissions associated with coal technology which were obtained from studies by Brookhaven National Laboratory and have just been discussed by Mr. Morris. For the risks from direct and indirect technology component production, errors of ±20% were assumed for the most developed technologies and ±35% for developing technologies. The uncertainties were assumed to be additive, i.e. statistically independent.

P.D. MOSKOWITZ: Did you include a credit for the decentralized photovoltaic system, since it is not connected to the grid?

L.J. HABEGGER: The decentralized system characterized does not have sufficient capacity and storage to allow complete independence from the utility grid. Some credit for reduced dependence may be appropriate, but the nature and level of the appropriate credit requires extensive evaluation of the impact on the grid and was not included.

J. HIBBERT: Fatalities occurring in the various phases of the electrical power generation cycle can be classified as 'early' in the case of industrial accidents or 'late' if they are due to such causes as pneumoconiosis or radiation-induced cancer. The impact of a late death is regarded as less severe than that of an early death. Although no general agreement exists, as far as I am aware, on the equivalence of early and late deaths, I would like to know whether you have considered it appropriate to make allowance for such effects in the comparative analysis of health risks presented in your paper.

L.J. HABEGGER: The full study report given in Ref. [3] indicates whether the fatalities related to various technologies are early or late, but no allowance was made in the aggregations presented in the paper. Allowance should perhaps also be made for differences in ages of the groups which incur the increased risks.

RISKS IN U.S. ENERGY MATERIAL TRANSPORTATION*

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Abstract

RISKS IN U.S. ENERGY MATERIAL TRANSPORTATION.

For the past five years, the Pacific Northwest Laboratory has been conducting a programme to study the safety of transporting energy materials. The overall objectives of the programme are to develop information on the safety of transporting hazardous materials required to support the major energy cycles in the USA. This information was developed for use in making energy policy decisions; in designing and developing new or improved transportation systems for these materials; to help establish research priorities; and as an aid in developing effective transportation safety regulations. Risk analysis was selected as the methodology for performing these studies. This methodology has been applied to rail and highway shipments of nuclear fuel cycle materials and liquid and gaseous fossil fuels. Studies of the risks of transporting spent nuclear fuel by train and uranium ore concentrates (yellow cake) by truck were expected to be issued early in 1981. Analyses of the risks of transporting reactor waste and transuranic wastes are in progress. The work completed to date for nuclear material transportation makes it possible to estimate the transportation risks for the entire fuel cycle in the USA. Results of the assessment are presented in this paper. Because the risk analysis studies for the transportation of gasoline, propane and chlorine have been performed using a methodology, basic assumptions and data that are consistent with the studies that have been performed for nuclear materials, comparisons between the risks for nuclear materials and these materials can also be made. It should be noted that it is not the intention of these comparisons to judge the safety of one industry in comparison with another. These comparisons can, however, provide some insights into the regulatory philosophy for hazardous materials transportation. The remaining sections of the paper briefly review the risk-analysis methodology used in these studies, provide an overview of the systems analysed, summarize the results and present conclusions.

INTRODUCTION

Currently available transportation risk analysis tools have been developed over a number of years to meet the growing needs to understand and improve the safety of transporting hazardous materials. The first transportation risk analysis approaches developed were based on statistical analysis of accident case histories.

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Much useful information can be obtained with these statistical approaches, but their utility is limited. In many cases, there may be insufficient accident experience or the accident data may not have been collected in a way that permits accurate risk analysis. Historical data also tends to identify the most probable kinds of accidental releases, while accidents with lower probabilities but potentially larger consequences may not be identified. Analysis of hazardous materials transportation systems that do not have a good historical data base required the development of 'predictive' risk assessment techniques. These methods use general information on the frequency and severity of transportation accidents to predict the probability that accidents with various consequence levels will occur during transport of hazardous materials. These methods permit analyses to be made of the risk of transporting materials for which there is little or no actual accident experience. Predictive risk analysis techniques were first used in two Holmes and Narver studies to determine the risk of transporting bioweapons [1] and radioactive material [2]. These techniques have been further refined in the work being performed at PNL. Studies have been completed that analyze the risks of transporting uranium hexafluoride by truck and train [3]; spent nuclear fuel by truck [4]; plutonium by truck [5], train [6] and cargo aircraft [7]; gasoline by truck [8]; propane by truck and train [9]; and chlorine by train [10].

RISK ANALYSIS METHODOLOGY

The risk analysis methodology used in these studies evolved from a number of risk analysis models originally developed for use in analyzing safety aspects of fixed nuclear facilities. These facilities have a well-defined population distribution and the population in the immediate vicinity of the plant (the exclusions area) is controlled by the facility operator. The population distribution in the vicinity of a transportation accident, however, is highly variable, and a variety of geographic and meteorological conditions can be encountered. This adds a degree of complexity not found in risk analysis of fixed sites.

Four basic steps are followed in the transportation risk analysis methodology used in the PNL studies. These four basic steps include:

System Description. The basic information on the shipping system to be analyzed is collected in this step. The industry being studied is characterized by gathering data on facility locations, industry shipping requirements and shipping destinations. Shipping packages and vehicles used to transport the materials being considered in the study are described in detail. Information is collected on the physical and chemical properties of the materials transported. Population distribution and weather characteristics along the shipping routes are characterized.

Event Sequence Identification. Possible sequences of events are identified that could result in a release of hazardous materials during transportation. Fault tree analysis has been used to identify potential release sequences in all of the studies that have been performed to date using the PNL transportation risk analysis methodology.

Event Sequence Evaluation. This step includes estimation of event sequence probabilities and evaluation of the potential consequences of each event sequence. This requires information on the response of the shipping system to normal and transport accident forces, together with knowledge of the forces present in transportation accidents. Mathematical analysis or data available from testing programs is generally used to estimate system failure thresholds. This can be combined with accident environment data such as that developed at Sandia National Laboratories [11, 12, 13] or statistical analysis of other accident data to calculate failure probabilities.

The consequences of a release of hazardous materials during transportation depend on the system characteristics, the nature of the failure that produced the release, the location of the release along the shipping route, and the weather and population conditions at the time of the release. These consequences may be determined from historical accident data, from information on tests that have been conducted with the material being shipped, or by engineering analysis. In general, consequences must be evaluated for each type of failure that can occur and for each combination of weather conditions and population distribution that can be encountered along the shipping routes that are used. The probabilities of encountering the various population distributions and weather conditions along the route are used in estimating the frequency of the various consequence levels that can result from a given release.

Risk Calculation. The information developed in the previous steps is used to calculate the risk for the shipping system being studied. The results are then analyzed to determine the primary contributors to the risk and to identify alternatives that could reduce the system risk. Since the information to perform the risk analysis has been developed in discrete data blocks, sensitivity studies can also be carried out to test the effect on the system risk of assumptions and approximations that were made to develop key pieces of information. This may identify areas where further analysis is required or delineate the limitations of the analysis. Sensitivity studies can also determine the effectiveness of design or regulatory changes in reducing the system risk.

SYSTEMS ANALYZED

Transportation systems discussed in this paper include materials shipped in the U.S. nuclear fuel cycle, the propane distribution system, the gasoline distribution system, and the

TABLE I. SUMMARY OF TRANSPORTATION SYSTEMS DESCRIPTIONS

Material shipped	Mode of transport	Shipping container	Container capacity	Containers/ shipment
Nuclear fuel cycle				
Yellow cake (U_3O_8)	Truck	Steel drums	360 kg	50
Natural UF_6 [a]	Truck	Steel cylinder	9.5 or 12.5 t	2 or 1
	Rail	Steel cylinder	12.5 t	4
Enriched UF_6 [b]	Truck	Steel cylinder with overpack	9.5 or 2.3 t	1 or 5
Fresh fuel	Truck	Overpack	1 PWR or 2 BWR assemblies	6 PWR or 16 BWR
Spent fuel[c]	Truck	Cask	1 PWR or 2 BWR assemblies	1
	Rail	Cask	7 PWR or 13 BWR assemblies	1
Reactor wastes	Truck	Drums Overpacks	210 ltr or 1.4 m^3 or 2.8 m^3	40 or 150 1
Spent fuel[d]	Rail	Cask	15 PWR or 41 BWR assemblies	1
Other wastes	Truck	Drums	210 ltr	150
Totals				
Chlorine	Rail	Steel tank	82 t	1
Propane	Truck	Steel tank	9 or 39 m^3	1
	Rail	Steel tank	130 m^3	1
Gasoline	Truck	Aluminum tank	32 m^3	1

[a] Expected about 1985.
[b] Includes shipments between enrichment plants.
[c] Reactors to interim storage.
[d] Interim storage to disposal site.

FOR SELECTED ENERGY MATERIALS IN THE USA

Quantity shipped (t/a)	No. of shipments	Avg. shipment distance (km)	Total distance (km/a)	t·km/a
2.4×10^4	1 350	2 600	3.5×10^6	6.3×10^7
2.5×10^4	1 700	650	1.1×10^6	1.7×10^7
3 700	75	540	4.0×10^4	2.0×10^5
5 300	490	860	4.2×10^5	4.5×10^6
3 890	840	1 000	8.4×10^5	4.7×10^6
1 170	1 830	930	1.7×10^6	1.1×10^6
2 720	630	930	5.8×10^5	2.5×10^6
3.4×10^4	1 800	1 200	4.4×10^5	7.3×10^6
5.3×10^4	8 400	1 200	1.0×10^7	4.9×10^7
3 890	375	4 600	1.7×10^6	1.8×10^7
6 800	450	1 000	4.5×10^5	6.7×10^6
1.6×10^5	1.8×10^4	1 150	2.1×10^7	1.7×10^8
4.5×10^6	5.5×10^4	450	2.5×10^7	2.0×10^9
3.2×10^7	3.0×10^6	120	3.6×10^8	5.3×10^9
1.5×10^6	2.5×10^4	400	1.0×10^8	6.4×10^9
3.2×10^8	1.4×10^7	80	1.1×10^9	2.5×10^{10}

chlorine distribution system. An overview of the characteristics of each of these systems is presented in this section. A summary of the key systems description information used in the risk assessment studies is presented in Table I. All systems are based on industry projections for the mid-1980s in the U.S.A.

Nuclear Fuel Cycle. Current plans in the U.S. are to indefinitely defer reprocessing of spent fuel. This policy results in a 'once-through' fuel cycle; with spent fuel being stored for a number of years and eventually disposed of as waste. Table I contains estimates of the materials required to be transported to support the operation for one year of a once-through fuel cycle with 100 GWe of installed generating capacity. Steady-state operation of a complete fuel cycle is assumed. The materials shipped include uranium ore concentrates (yellowcake) from uranium mills to UF_6 conversion facilities; natural UF_6 from the conversion facilities to enrichment plants; enriched UF_6 from enrichment plants to fuel fabrication plants; fresh nuclear fuel to reactors; spent fuel to interim storage facilities and from the storage facilities to a geologic repository; and low-level wastes from all the facilities to shallow-land burial grounds. Shipping distances are actual distances for existing facilities that are not currently in existence.

Chlorine Distribution. Chlorine is used as a chemical feedstock, as a bleaching agent, and in purification of domestic water supplies. In the U.S., about 70% of the chlorine is shipped from the manufacturing plants to the user in railroad cars. This is the portion of the chlorine distribution system considered in the risk analysis. The railroad tank cars used for chlorine transport have a capacity of 82 t of chlorine. The tanks are insulated and equipped with pressure relief valves.

Propane Distribution. In the U.S., propane is used as a domestic fuel, primarily in sparsely populated rural areas where the cost of providing piped natural gas service would be prohibitive. Propane is separated from natural gas at processing plants near the gas fields and also produced as a refinery by-product. It is transported from these facilities to retail distributors by pipeline, barge or railroad tank car and distributed to customers primarily in large- or medium-sized tank trucks. The rail and truck portions of the distributions systems were analyzed in this study. They account for most of the transportation risks that result from operation of the system. The rail tank cars used for propane transportation are large steel pressure cylinders with a liquid volume capacity of 130 m^3. They are equipped with thermal insulation, pressure relief valves, and head shields and shelf couplers to prevent coupler puncture of the tank ends in accidents. The tank trucks used in the system are also large steel pressure cylinders. Two typical trucks have capacities of 9 m^3 and 39 m^3 of propane. The tanks are equipped with pressure relief valves but are not insulated.

Gasoline Distribution. In the mid-1980s, gasoline is expected to provide energy for about 10^{17} vehicle miles of transportation in the U.S. Gasoline is shipped in bulk quantities from refineries to wholesale distribution terminals by pipeline or barge. Tank trucks are used to distribute the gasoline to the retail outlets. Truck transportation of gasoline was analyzed in this study. It represents most of the transportation risk in the gasoline distribution system. The truck tanks used in this system have aluminum walls and a capacity of 32 m^3. Both truck-trailer and tractor-semitrailer vehicles are used.

RESULTS AND CONCLUSIONS

Estimated risks associated with accidental releases of materials transported for each step of the nuclear fuel cycle are presented in Table II. These estimates are based on the work presented in References 4, 10, 11, 12, and 13 and preliminary results from studies that are nearing completion. The risk numbers were calculated for the reference fuel cycle described in Table I. The risk estimates include both immediate and latent fatalities caused by releases of these materials in transportation accidents. Studies of the risk of transporting yellowcake, fresh nuclear fuel and low-level wastes from the front end of the fuel cycle have not been completed. Existing information permits estimates of the risks to be made. The estimates presented in the table result from the very low hazards associated with release of these materials. These estimates are consistent with the results of other studies.

The results presented in Table II show that risks from all the fuel cycle transportation steps are low. The results also indicate that the total transportation risks associated with the nuclear fuel cycle are distributed about evenly between the front end (fuel supply) and the back end (waste management) of the cycle. Risks in the front end of the cycle result primarily from the chemical toxicity of the materials transported (even though transportation of these materials is regulated on the basis of their radioactivity). For example, public health risks from transportation accidents that release UF_6 are due almost to the hydrogen fluoride (HF) gas that is formed when UF_6 reacts with water vapor in the air. This is the only mechanism in any of the fuel cycle steps with potential to cause immediate fatalities from accidental releases during transportation. All other fatalities associated with accidental releases of fuel cycle materials during transportation are latent effects associated with the toxicity of heavy metals or the increase in cancer rates in populations exposed to increased levels of radioactivity.

TABLE II. ESTIMATED TRANPORTATION ACCIDENT RISKS FROM THE ONCE-THROUGH NUCLEAR FUEL CYCLE

Fuel cycle step	Material shipped	Transportation accident risks (fatalities/a)
Mining/milling	U_3O_8 (yellow cake)	$<10^{-6}$
Conversion/enrichment	UF_6	6×10^{-5}
Fuel fabrication	Fresh fuel	$<10^{-6}$
Power production	Spent fuel	10^{-5}
	Reactor wastes	2×10^{-6}
Fuel storage/disposal	Spent fuel	2×10^{-5}
Waste management	Other fuel cycle wastes	$<10^{-6}$
Total		10^{-4}

TABLE III. COMPARISON OF RISK-ANALYSIS RESULTS FOR TRANSPORTATION OF SELECTED ENERGY MATERIALS IN THE USA

Transportation system	Risk measure					
	Fatalities/ year	Fatalities/ shipment	Fatalities/ km	Fatalities/ t·km	Fatalities/ GW·h	Maximum fatalities
Nuclear fuel cycle shipments	1×10^{-4}	6×10^{-9}	5×10^{-12}	6×10^{-13}	2×10^{-11}	100
Chlorine	9	2×10^{-4}	4×10^{-7}	4×10^{-9}	–	600
Propane	15	5×10^{-6}	3×10^{-8}	1×10^{-9}	4×10^{-5}	800
Gasoline	29	2×10^{-6}	3×10^{-8}	1×10^{-9}	2×10^{-5}	40

The results of the risk analysis studies for transportation of nuclear fuel cycle materials are compared with the results for the three studies that have been completed for non-nuclear systems in Table III. A number of risk measures are presented to facilitate comparisons among the various systems. These risk measures include the usual risk number (expected fatalities/a) and the expected number of fatalities per shipment, per km and per t-km. Conversion factors for the latter quantities are taken from the system description information in Table I. An estimate of the risk normalized to the energy content of the material transported is also presented. These numbers are based on the electricity generated by a 100-GWe installed nuclear fuel cycle and the estimated amount of electricity that would be required to replace the energy supplied by the propane and gasoline. The risk analysis methodology used in these studies identifies the complete spectrum of potential accident consequences and estimates the probability of events producing that level of consequence. The maximum number of fatalities predicted for each material is present in Table III. A variety of risk measures have been used because of the inherent difficulties in making risk comparisons. Examination of a number of risk measures can provide additional insights and help guard against conclusions that are dependent on the way the risk information has been developed and displayed.

The results summarized in Table III indicate that the risks from transporting these materials are all relatively low in comparison to other risks in society. For example, in the U.S., highway fatalities are about 55 000/a; about 6500 people die in building fires each year; 160 are killed by lightning; and 90 die in tornadoes. The results also show that for all the risk measures, gasoline, propane and chlorine transportation risks are very similar. The risks from shipments in the nuclear fuel cycle are, in general, several orders of magnitude less that the risks for the other hazardous materials. This is especially true when comparing risks normalized by the energy content of the material transported. This results primarily because nuclear materials have a high energy density, but this energy cannot be released in a transportation accident. Effects of release of fuel cycle materials are secondary ones caused by the radioactivity or chemical toxicity of the material transported. The concern in releases of gasoline or propane is the uncontrolled release of the stored chemical energy in these materials, resulting in a fire or explosion.

The relatively low risks associated with nuclear fuel cycle shipments in comparison to the other materials can be attributed to a number of factors. Foremost is probably the traditional conservatism built into standards and regulations in the nuclear industry. The transportation regulations for radioactive materials require increased levels of protection as the potential hazard from the material shipped is increased. Significant quantities of radioactive materials are provided protective packaging designed to prevent releases in severe transportation accidents. Packagings for other hazardous materials are generally designed to provide containment during normal transportation with limited protection during accidents. Package resistance to accident stresses is reflected most directly in the fatalities/km risk measure. The value of this risk measure for fuel cycle materials differs from the value for the other hazardous materials by about four orders of magnitude. This is approximately the difference in risk that would result from transporting a hazardous material in a package designed to withstand severe accident conditions instead of a package designed to withstand only normal transport forces.

These calculated risk numbers provide a basis for examining the current hazardous material transportation regulatory philosophy. There seems to be consistency in the level of protection required for hazardous materials such as chlorine, propane and gasoline. However, there seems to be an inconsistency when these are compared to nuclear shipments. For all the risk measures except maximum credible consequences, nuclear materials transportation risks are much lower than those for other materials. Even for maximum consequences, risks from propane and chlorine are 6 to 8 times higher. This would indicate that risks from nuclear material shipment are perceived to be more hazardous than they really are. There also seems to be a different risk/benefit philosophy applied to nuclear materials than to other hazardous materials. The risk/GW-h risk measure indicates that nuclear material transportation has by far the lowest risk/benefit ratio of the materials analyzed. Nevertheless, significant efforts are still under way in the U.S. to provide even more stringent controls over these shipments. Based on the figures presented in Table III, it is the opinion of the authors that there appears to be little real need to attempt to substantially increase the level of protection provided nuclear materials in transport.

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DISCUSSION

S.I. IMAGAWA: Do you consider transport risks to be dependent on population density?

A.L. FRANKLIN: Transportation risks are directly related to the population density of the region through which the shipments are made. Our methodology uses population density as one of the variables in calculations of consequences. When using population as a variable it is important to take into account the probability of encountering each of the different population densities.

M. EL DESOUKY: Are there any regulations regarding the transport of chlorine gas by truck through populated areas, and would it be possible to evaluate the risk associated with such transport?

A.L. FRANKLIN: I am not aware of any restrictions on transporting chlorine through populated areas, although there are specifications for the construction of the shipping containers and regulations regarding their labelling. These specifications and regulations are intended to reduce the probability of accidental release and to inform emergency response teams of the nature of the cargo. The risk involved could be evaluated. Much of the analysis would be similar to the 'propane by truck' analysis which has already been carried out. There would, on the other hand, be differences in the methods of estimating the consequences.

L.D. HAMILTON: I should simply like to point out that your conclusion that current protection levels for nuclear materials are adequate is beyond the scope of this Symposium. We are concerned with the quantitation of health impacts. We should provide this information to policy analysts and decision-makers and leave to their judgement the question whether protection levels are adequate.

RADIOLOGICAL RISK ANALYSIS OF AN OPERATING HIGH-LEVEL WASTE REPOSITORY

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Abstract

RADIOLOGICAL RISK ANALYSIS OF AN OPERATING HIGH-LEVEL WASTE REPOSITORY.

Many dose assessments of generic facilities have been undertaken since the decision was made to pursue a deep geologic repository as the primary option for the disposal of high-level nuclear waste. As the siting and design efforts continue, it seems appropriate to review the various assessments and identify aspects of the repository operation which appear capable of contributing major portions of the potential doses. Representative samples of dose assessment results are exhibited for the important phases of repository development and operation: construction, operation, transport, decommissioning and retrieval. Operational aspects are further subdivided into routine and accidental considerations. These results are analysed to ascertain: (1) the orders of magnitude of the doses from each phase; (2) the relative importance of occupational versus population doses for each phase; and (3) the major facility operations and systems contributing to the doses. Converting all of the whole-body doses discussed to 50-year dose commitments in person-rem and ranking from high consequences to low, one obtains the following population and occupational lists: (a) Population: hoist-drop accident; transport; routine operation; and construction. (b) Occupational: hoist-drop accident; routine operation; transport; and construction. Except for the occupational result from the hoist-drop accident, all the postulated doses represented in the above list are orders of magnitude less than the person-rem received by the same groups from natural radiation in the same time period.

INTRODUCTION

One of the National Waste Terminal Storage Program's performance objectives is that "Risks during the operating phase of waste disposal systems should not be greater than those allowed for other nuclear fuel cycle facilities". Several assessments of the radiological performance of operating deep geologic repositories have been made by a number of groups, with relatively consistent results. This presentation summarizes these assessments for both normal and accidental conditions, and for occupational, maximum individual, and public exposure.

The context of the analyses is established by means of brief descriptions of site, facility, and facility operational characteristics. Representative samples of dose assessment results are reviewed to identify aspects of the repository operation which appear to be capable of contributing major portions of potential doses.

FACILITY DESCRIPTION

The reference repository is located on a site which contains an inner controlled area that occupies some 850 ha (2 100 acres) and a central exclusion area that covers 162 ha (400 acres) on the surface. Surrounding the site is an outer controlled area which extends 3.2 km (2 miles) from the inner controlled area.

The repository is designed to receive, package and dispose of one-third of all spent fuel generated by commercial nuclear power plants in the USA , assuming a 380-GWe nuclear generating capacity (as compared with the present capacity of 56 GWe). It will also receive transuranic (TRU) waste from commercial facilities and all low-level radioactive waste generated on the site. However, because of the greater potential radiological hazard, only spent fuel will be considered in the dose assessment results here. Surface facilities consist of radioactive waste receiving and handling facilities, and excavated waste rock handling and storage facilities. The underground workings consists of a 556-ha (1 375-acre) area excavated in a room-and-pillar arrangement. The underground, including an outer buffer zone, occupies approximately 850 ha (2 100 acres). Access to the underground is through one men-and-materials shaft, two exhaust shafts, one waste shaft, and one ventilation-supply and emergency-egress shaft.

Before the radioactive waste is emplaced in the underground disposal rooms, a progression of receiving, storage, preparation and transfer operations takes place. These operations are performed in the waste-handling building, the principal surface facility of the reference repository. The waste-handling facility has the following major components:

- Shipping-cask receipt and unloading
- Cask wash and cooldown facility
- Water-filled unloading pools
- Presentation pool, lag storage pools, and connecting canals
- Transfer and drying facility
- New canister receipt and storage
- Weld-and-test cells for canistering spent-fuel assemblies

- Special function cell for canistering defective or damaged spent-fuel assemblies
- Empty shipping-cask decontamination

The underground facilities consist of shafts, corridors, disposal rooms and buffer rooms 640 m (2 100 ft) below the surface. The main underground corridors are parallel openings extending along the length of the repository. For long-term stability, these corridors are separated by massive pillars of rock. The corridors will be used for the transportation of personnel, materials and equipment; the transportation of waste containers; providing a belt-conveyor route for moving waste rock; disposal area exhaust; and development area ventilation supply and exhaust. Branch corridors extending perpendicularly from the main corridors will provide access for personnel, materials, waste and ventilation to one panel of disposal rooms. They are connected with crosscut corridors that can also be used as egress routes in emergencies.

Approximately 700 disposal rooms will be excavated for the disposal of spent fuel. The rooms are closed at one end and open to a branch corridor at the other end. Each room contains 264 predrilled burial holes in two rows. Buffer rooms, which separate the disposal rooms from the buffer zone, are provided to accommodate the TRU waste and low-level waste.

WASTE-HANDLING OPERATIONS

Spent fuel will be received in standard shipping casks. After its auxiliary equipment is removed, the cask will be unloaded from the carrier and moved to the cooldown area. From here the cask will be transferred to the spent-fuel unloading pool.

The fuel assemblies will be removed under water and loaded into transfer baskets. The empty cask will then be washed down, reassembled in the preparation area, and shipped out of the repository for reuse. The transfer basket will be moved to storage racks in a lag storage pool for temporary surge storage or to the presentation pool. In the next step a buggy will transfer the basket to a weld-and-test cell or to the special function cell.

Each fuel assembly will be lifted out of the buggy and dried. It will then be inserted into a canister in the weld-and-test cell. These operations will be remotely controlled. Damaged or broken fuel assemblies will be canistered in the special function cell.

Each canister will be identified as to its contents and transferred underground in a transfer cask. A special transporter will move this cask to the waste shaft for loading into the hoist cage. Loading and unloading will be carried out simultaneously on the surface and underground from the double-cage waste-hoisting system. After descending underground, the transfer cask will be loaded onto an underground transporter that will move it to a disposal room.

On arrival in the disposal room, the canisters will be emplaced in the predrilled burial holes by the burial crane. The empty transfer cask will be returned to the surface for reuse.

During its peak year of operation, the repository will dispose of about 12 100 boiling water reactor (BWR) assemblies and 8 600 pressurized water reactor (PWR) assemblies. The average emplacement rate will be 43 canisters per day.

RISK ASSESSMENT METHODOLOGIES FOR NORMAL OPERATIONS

The strategy for assessment of the risk of normal repository operations consists of two steps: (a) the identification and classification into separate 'Unit Operations' of all operations involving potential radiation exposures, and (b) the determination of doses anticipated to be associated with each Unit Operation. Assuming probabilities of one for all routine unit operations, the risks (consequence x probability) and doses are numerically equal.

The strategy has been applied to all operations related to waste handling in the facility, including transportation, retrieval, decommissioning operations, as well as normal maintenance and construction activities. Doses to workers and the public resulting from various Unit Operations have been determined and analyzed individually and collectively. These results are summarized following a discussion of the assessment methodologies.

The first step in the safety assessment of normal operations is the identification and classification of all operations into 'Unit Operations' involving potential radiation exposures. The 'Unit Operations' list is of sufficient detail to permit an account of all waste material transfers from receipt of the waste at the shippers facility to final emplacement, and all other tasks recognized as involving radiation or radioactivity.

Each 'Unit Operation' has distinct boundary conditions; that is, each has a unique radiological condition, a specific number of population and/or workers involved, a defined duration, and a defined physical boundary.

The potential of radiological hazards to repository personnel from normal operations is due to the presence of large quantities of radiation or radioactivity and work operations requiring long residence times in proximity to radiation sources. Internal exposures will be precluded in normal operations by measures such as protective clothing and respiratory protection equipment. On the other hand, in general, the potential public risks from normal operations are associated with the accumulation of radionuclides chronically released from the facility at low levels and recurring exposure to low-level radiation fields associated with transportation of waste.

The methodology for the assessment of occupational exposures from normal operations for each unit operation is based on the quantification and multiplication of (a) the number (by classification) of workers involved in each task, (b) the duration of each task, and (c) the radiological conditions associated with each task. The total exposure is determined by forming the product of these three terms.

The methodology for the assessment of potential public exposure to radiation is based on an evaluation of the release of effluent materials from the repository during a given Unit Operation, and possible external exposures during transportation of radioactive wastes en route.

The effluents released routinely may be particulates, gases or liquids; their quantities can be determined from operational aspects of the repository such as throughput rate. Following identification of source-terms, the critical pathways are analyzed to evaluate the proportion of the source terms which reaches a maximum individual, and the radiological dose consequences are then calculated. For transportation-related exposures, external dose rates and population exposure configurations are considered in consequence calculations [1].

RISK ASSESSMENT RESULTS FOR NORMAL OPERATIONS

Construction

Essentially all radiological exposures to personnel or population during construction result from the presence and liberation of Rn-222 and its daughters during underground excavation. Assuming the presence of an excavation work force in the repository for 7 years, 2 040 hours per year, a 50-year whole-body dose commitment of 0.13 person-rem is calculated [1] for a salt repository. Dispersion of these radionuclides to the environment from the excavated caverns results in a 50-year whole-body dose commitment to the population within 50 miles of the site of 0.005 person-rem [1]. For purposes of comparison, the

TABLE I

EXPECTED DOSES DURING ROUTINE CANISTER HANDLING AND STORAGE OPERATIONS

Operation Performed	Expected Occupancy Time (Min.)	Expected Max. Dose Rate (mrem/hr)	No. of Personnel	Integrated Dose Person-mrem Per Canister
Cask arrival, inspection and documentation				~1
Preparation of cask for unloading, washdown, and sampling				~2
Attach and seal cask to hot cell floor				~4
Move canister from holding station to transfer cask and to shaft loading station	6/canister	2	1	0.2
Load and unload waste cage at top and bottom of shaft and load transporter	6/canister	10	1	1
Transport cask to storage area, locate hole, and spot transporter	18/canister	2.5	2	1.5
Remove plug, lower canister, replace plug	6/canister	20	2	4.0
				13.7 or ~14

radon contents of other potential repository media such as basalt and granite are estimated to be greater than for salt.

If there are radioactive 'impurities' such as uranium or thorium present in the salt, there are two pathways by which the mine tailings could present a potential for population exposures. The first is the release of radon directly into the air. The second is the resuspension of salt particulates containing the radionuclides by the wind.

Since there is not expected to be any significant amount of uranium or thorium in the salt mine, the salt-tailings pile should not constitute a major radiological problem. In case there is an accumulation of radon near the tailings pile, a few feet of earth cover, or layers of plastic cover, would reduce the radon escape to essentially zero.

Routine Operation

Routine radiological releases from geologic repositories during normal operation will consist principally of radon emanating from exposed rock faces and radon's decay products. These releases will also occur from backfilling operations but are negligible compared to radon releases during repository construction. Occasionally, external contamination may occur on canisters as a result of some minor accident. The population dose from decontamination activities would be much less than that from operation at a spent-fuel packaging and storing facility, for which the 50-year whole-body population dose was determined to be about 0.7 person-rem [1].

Routine occupational external exposures have been assessed by (a) identifying each task associated with operation of the repository, (b) estimating the duration of each task, (c) characterizing the radiological conditions surrounding the execution of each task, and (d) establishing the number of workers required for each task. Table I gives the expected doses during routine canister handling and storage activities. Adjusting the 14 person-mrem per canister for the 20 700 canisters expected in the busiest year of repository operation, the occupational external dose per year is calculated to be 290 person-rem [2]. Members of work classifications other than operators are expected to receive less dose. The average maintenance person is expected to receive approximately 90 percent of the average operator dose; health physicists - 60 percent; and security personnel - 20 percent [3].

Retrieval and Decommissioning Operations

Retrieval of waste canisters, if required, will be accomplished with the same relative effort as that required

TABLE II

ROUTINE TRANSPORTATION-RELATED DOSES

Siting Option	Population Dose (Person-rem)
Single repository location	47
Three sites based on equal capacity regions	21
Five sites based on NRC regions	15
Five sites based on equal capacity regions	14
Nine sites based on electric reliability council regions	13
Total truck drivers' dose	16x*
Total rail workers' dose	7x*

*16 and 7 times the corresponding population doses.

originally for emplacement [2]. Dose rates for retrieval, therefore, are not expected to exceed those for emplacement.

During decommissioning, the potential exists for both external and internal exposures to personnel from residual contamination. The shipping-cask handling area, the transfer-cask handling area and the below-ground areas will have been maintained in fairly clean conditions during normal operation, or will have been sealed off because of permanent burial of the canisters. These areas will not pose an external or internal radiation problem to decommissioning personnel. The hot cell is anticipated to be the major contributor to occupational dose (15 person-rem, 50-year dose commitment) during this phase.

Transportation

One important aspect of siting a repository is transporting the waste to the repository from its point of origin. A study has been made of the population doses which would be expected to occur from the transportation of spent fuel to sites selected on the basis of various alternatives. The results are shown in Table II [4]. These doses reflect the dose accumulated during the complete operation lifetime of the repository(ies). It can be noted that the occupational doses vary from 7 to 16 times the population doses shown in the table.

RISK ASSESSMENT METHODOLOGIES FOR ACCIDENT CONDITIONS

The strategy for assessing the safety of accidental conditions includes: (a) identification of credible events which might initiate accidental conditions, (b) development of detailed scenarios for each event, and (c) evaluation of the consequences of each selected scenario.

The scenarios span the range of potential consequences and occurrence probabilities, address all potential initiating mechanisms, and affect all components of the waste-management and transportation system. Doses to both workers and the public resulting from various accident scenarios are determined and analyzed both individually and collectively.

The scenario approach to consequence assessment uses the sequential application of known or postulated characteristics of the facility, site, and radioactive contamination. The sequence in which the characteristics are applied to the analysis is:

- Contaminant release characteristics (source term)
- Contaminant transport characteristics
- Deposition characteristics
- Radiological dose consequence

Based on the description of the initiating event and the inventory involved, the amount of contaminant released is calculated. This quantity may be based on a percent of inventory exposed to the initiating event, on the ability of moving air to keep particulates suspended, or on the rate of contaminant released and the duration of the incident.

The result of applying the release fraction in the accident assessment sequence is an estimate of the quantity of radionuclides available for transport either to the occupationally exposed person or to the public. How the contamination is assumed to be transported depends upon the characteristics of the contaminant and the facility.

In many analyses, the quantity of radioactive material deposited at any point in the facility or in the environment is treated as an end point. Here, the deposition characteristics are the precursors of dose calculations. This analysis includes considerations such as personnel exposure time to the contaminant based on anticipated emergency response of employees. For release to the environment, parameters of importance in the determination of deposition include the critical environmental pathways for the nuclides released, the meteorological characteristics of the area, and other environmental characteristics.

For accidents occurring during the operations phase, the principal radioactivity-release pathway is airborne. Inhalation and whole-body doses delivered to an individual by exposure to airborne radioactivity are calculated. Other dose pathways, such as the ingestion of contaminated food or water, are considered for severe and perhaps incredible accidents where particulates are assumed to be released to the atmosphere as in the case of the hoist-drop accident. Particulates would not be released from normal or expected abnormal conditions.

RISK ASSESSMENT RESULTS FOR ACCIDENT CONDITIONS

Many different accident scenarios can be postulated for an operating repository. Most comprehensive accident assessments have been based on the need to indicate the consequences from a broad range of potential scenarios.

One example of a higher probability scenario is the accidental venting of a leaky fuel assembly inside the waste-handling building. In analyzing this accident it is assumed that operator error or equipment failure results in a poor connection, or no connection at all, between the cask and the off-gas system and a release of gaseous radionuclide into the waste-handling building. This occurrence would subject the staff within the building to a radioactivity inhalation hazard for 15 minutes, after which administrative controls would limit access inside and require personnel to wear respirators. During the entire period of the gaseous release, the building HVAC system will be operational. The resulting doses are shown in Table III [2]. The doses are small, but demonstrate the value of personnel response to an occurrence. It can be noted that doses at the exclusion area boundary are more than those for the worker. This type of result is observed when the worker reduced his exposure time to the radiation cloud by leaving the site of the occurrence, whereas the individual at the boundary is assumed to be exposed during the complete passage of the contaminant cloud.

The most significant operational accident yet identified is the hoist-drop accident. The sequence of events for this hypothetical accident is as follows: all four cables supporting the hoist cage and the loaded transfer cask break, allowing the cask to fall down the entire length of the hoist shaft, with the four hoist cables suspended below it. The velocity near the bottom of the shaft is sufficient for the cask to break through the structural members at the lower work station and to carry them and the cables down to the bottom of the shaft, more than 30 m (100 ft) below. Although the materials preceding the cask to the bottom of the shaft will tend to cushion it, and the cask provides some protection for the fuel

TABLE III

DOSES FROM ACCIDENTAL VENTING OF A LEAKY FUEL ASSEMBLY INSIDE THE WASTE-HANDLING BUILDING

	External Dose (mrem)		Dose Due to Inhalation (mrem)	
	Beta	Gamma	Whole Body	Lung
Worker	2.9×10^{0}	3.5×10^{-2}	9.5×10^{-2}	1.1×10^{-1}
At Exclusion Area Boundary	6.5×10^{0}	7.8×10^{-2}	6.2×10^{-1}	7.1×10^{-1}
At 2.0-Mile Distance	2.0×10^{-1}	2.3×10^{-3}	1.9×10^{-2}	2.1×10^{-2}

assembly inside, it was conservatively assumed that the spent fuel assemblies break, releasing part of the contents.

A hoist drop is conservatively estimated to result in a maximum individual bone dose of 0.01 rem. Organ doses are considerably less. External beta and gamma doses are about 0.1 rem and 5×10^{-4} rem, respectively. The dose calculations have been performed for one PWR assembly in the accident. Dropping two BWR assemblies will result in smaller doses because of the smaller radioactive inventory in two BWR assemblies. The population doses resulting from this accident are several orders of magnitude higher than for the gas leak accident discussed earlier. However, the probability of occurrence (10^{-5} per year) is several orders of magnitude less. The mix of nuclides contained in the fuel assembly assumed to be involved includes fission products, long-lived halogens, heavy metals and transuranics. Therefore, a fairly uniform distribution of dose among the exposed organs is observed.

The WIPP Environmental Evaluation Group of the State of New Mexico has evaluated potential dose commitments to a Maximum Exposed Individual following a railroad accident involving high-level waste [5]. Example postaccident protective actions are assumed in the calculations. First-year ingestion doses are precluded via the protective actions. First-year external dose to the maximum exposed individual is calculated to be 0.16 rem. Fifty-year dose commitments to liver, bone and whole body to this individual are reported to be 3.4, 3.2 and 1.3 rem, respectively. The first-year maximum individual appears to be an adult whereas the 50-year is an infant.

CONCLUSIONS

Converting previously discussed whole-body doses to 50-year dose commitments in person-rem and ranking from high consequences to low ones gets the following list for population and occupational:

Population	Occupational
Hoist-drop accident	Hoist-drop accident
Transportation	Retrieval
Retrieval	Routine operation
Routine operation	Transportation
Accidental venting	Decommissioning
Construction	Accidental venting
Decommissioning	Construction

Except for the occupational result from the hoist drop accidents, all of the postulated doses represented in the above list are significantly less than the person-rem received by the same groups from natural radiation in the same time period.

Examination of the population list leads one to the conclusion that accidents present the greatest potential risk. It should be noted, however, that the hoist-drop accident results in about the same population dose commitment as does routine operations to the workers.

Transportation is observed to fall in the middle of the consequence listing. It might be noted, however, that occupational doses from transportation only are about 0.1 of occupational doses received from operation of the facility.

The fact that routine operation falls toward the bottom of the population list leads one to conclude that the facilities assumed in these assessments are properly designed to minimize environmental releases. One might conclude therefore that routine operation attention should be focused on the occupational aspects. This conclusion is reinforced by the high rank of routine operations in the occupational list. A further source of incentive to focus on occupational aspects of routine operation is a potential reduction in occupational dose limits which might affect some of the higher exposed members of the repository work force.

Decommissioning of the facility does not appear to present a significant risk to workers and can justifiably be included in preclosure safety assessments as just another set of tasks in the preclosure time phase. Preliminary examination of retrieval operations indicates that the dose rates are not anticipated to exceed those of emplacement. However, retrieval will impact dose commitment by possibly extending the operational lifetime of the repository.

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DISCUSSION

R. WILSON: Have you any information on the person-rem per MW(e)·a?

D.A. WAITE: All the results discussed in my presentation were calculated on the basis of a repository designed to receive, package and dispose of one third of all spent fuel generated by commercial nuclear power in the USA, assuming a 380-GW(e) nuclear generating capacity.

F. GIRARDI: If I understood you correctly, you assumed in your study the complete retrievability of all waste for the entire operational period. Did you also evaluate alternative management policies for the repositories, such as backfilling individual galleries after a certain control period? If so, what is the effect on the overall operational risk?

D.A. WAITE: Complete retrievability during the operational phase was in fact assumed. Alternative management policies such as backfilling individual galleries were also analysed in the context of retrievability. For reasonable alternative design and management policies, the retrieval risks tend to be in the same range as those stated in the paper for emplacement. Given that the canisters are designed to perform adequately for this period and well beyond, the most important factor in deciding which alternative management strategy to use is the amount of waste rock which can reasonably be accommodated on the surface at one time. In other words, what we must ask is whether waste rock needs to be moved back into the repository before the operational period is completed because of waste rock storage considerations, or whether it makes sense to move all waste rock to the surface when some newly excavated material could be moved directly to a completed gallery for use as backfill without ever being transported to the surface.

S.I. IMAGAWA: Your paper concludes that expected doses during routine canister handling and storage operations amount to about 13.7 (mrem-person) × 20 700, or between 274 and 290 person-rem. Isn't this rather a high figure?

D.A. WAITE: Whether the 274 to 290 person-rem per year number is the absolute minimum for operating such a facility will only be seen when an ALARA analysis of facility operations is completed. However, the intention at this stage is to determine whether the risks during the operational phase of waste disposal systems are lower than those permitted for other nuclear fuel cycle facilities. In view of the fact that three such facilities could handle the entire waste output of the United States commercial nuclear power industry and that the total risk is about 0.1 projected health effects per year, I would conclude that the numbers presented are not unreasonably high.

Session VII

RISK ANALYSIS STUDIES
COMPARING DIFFERENT ENERGY SOURCES

Chairman
K.G. VOHRA
India

Invited Paper

HEALTH IMPLICATIONS OF HYDROPOWER DEVELOPMENT

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Abstract

HEALTH IMPLICATIONS OF HYDROPOWER DEVELOPMENT.

Hydropower development had been neglected in many countries during the past few decades, but the situation dramatically changed during the 1970s owing to the constantly increasing costs of electricity generation by fossil-fuel and nuclear power plants. Currently, hydroelectric generation accounts for approximately 23% of total global electricity supply. Much of the hydropower potential in developing countries of Africa, Asia and Latin America still remains to be exploited. Like any other source of energy, hydropower development has several health impacts. Conceptually, health implications of hydropower development can be divided into two broad categories: short-term and long-term problems. Short-term health impacts occur during the planning, construction and immediate post-construction phases, whereas long-term impacts stem from the presence of large man-made lakes, development of extensive canal systems, alteration of the ecosystem of the area, and changing socio-economic conditions. Longer-term impacts are further classified into two categories: introduction of new diseases and/or intensification of existing ones due to the improvements of the habitats of disease-carrying vectors, and health problems arising from resettlement of the people whose homes and land-holdings are inundated by the reservoirs. All these impacts are discussed in detail. Health impacts of hydropower developments have not yet been studied extensively. It is often implicitly assumed that health impacts of major dams are minor compared with other social and environmental impacts. Future studies could possibly reverse this assumption.

1. INTRODUCTION

Use of water power is not a new phenomenon: it has been used in one form or another for some two thousand years. Earlier attempts to harness energy from water were small-scale and for specific purposes only. The situation has dramatically changed during the past hundred years owing to major scientific and technological innovations.

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* President, International Society for Ecological Modelling; Vice-President, International Water Resources Association.

Hydropower development was somewhat neglected in many countries during the 1960s, mainly because it was economically more advantageous to generate electricity by using oil, which was cheap and plentiful. Although operating costs of hydropower plants are low, capital costs are high, and accordingly more emphasis was placed on power plants operating with fossil fuels. The events of October 1973 started to change this situation, when constantly escalating oil prices ended the era of cheap energy and forced most countries to explore alternative ways of generating cheaper electricity. Since hydropower is the only renewable form of energy that can be used extensively in most countries for large-scale generation of electricity and can be developed with indigenous expertise, it started to attract increasing attention in the 1970s. Since the dams and various structures built are also invariably used for other purposes like irrigation, flood control, navigation, recreation, aquaculture, etc., such comprehensive developments have become increasingly attractive for many countries. For developing countries in particular, water resource developments could successfully contribute to the solution of two major crises facing them today: energy and food. Furthermore, if the construction processes are planned carefully so as to use extensive labour-intensive technology, employment could be provided for a large number of unskilled labourers -- the main type of labour force unemployed – for several years [1].

According to the survey of energy resources presented at the World Energy Conference, Munich, in September 1980, hydroelectric generation currently accounts for approximately 23% of total global electricity supply. This is an average figure: for many countries hydropower is the principal means of electricity generation. For countries like Brazil, Canada, Morocco, Norway and Sri Lanka, hydropower currently accounts for 70 100% of all electricity generated [2].

While potential for hydropower development has been exploited to a considerable extent in the advanced industrialized countries of North America and Europe, including the USSR, a vast potential in many countries of Africa, Asia and Latin America still remains to be developed. Annual hydroelectric potentials for different regions of the world, as provided during the World Energy Conference of 1980, are shown in Table I. It should be pointed out that during the preparation for the UN Conference on New and Renewable Sources of Energy at Nairobi in August 1981 it was felt that the estimates provided for Asia may not include data from the People's Republic of China. The theoretical potential for the People's Republic of China is currently estimated at 6×10^{12} kW·h, with technically usable potential of 1.9×10^{12} kW·h. At the end of 1979, the total operating potential was 0.05×10^{12} kW·h and potential under construction was 0.0517×10^{12} kW·h.

With the present constantly changing energy price scenarios, and since hydropower is a good source of indigenous energy, it is highly likely that an increasing

TABLE I. ANNUAL HYDROELECTRIC POTENTIAL OF DIFFERENT REGIONS

Region	Theoretical potential	Technical usable potential	Operating potential	Potential under construction	Planned potential
	←		(10^{12} kW·h)		→
Africa	10.118	3.14	0.151	0.047	0.201
North America	6.15	3.12	1.129	0.303	0.342
Latin America	5.67	3.78	0.299	0.355	0.809
Asia (excluding USSR)	16.486	5.34	0.465	0.080	0.368
Oceania	1.5	0.39	0.059	0.020	0.032
Europe	4.36	1.43	0.842	0.094	0.197
USSR	3.94	2.19	0.265	0.191	0.17[a]
Total	44.28	19.39	3.207	1.090	2.12

[a] Estimated.

number of countries will encourage intensive development of their hydropower potentials. Estimates of probable hydroelectric development during the period 1976–2020, as provided during the 1980 World Energy Conference, are shown in Table II, which indicates that the total annual energy from installed hydroelectric facilities from all regions of the world is likely to increase five-fold during these 44 years.

2. HEALTH IMPLICATIONS

Like any other form of energy, hydropower development has several health implications. It is important, however, to note a major distinction between hydropower and fossil-fuel or nuclear power plants that generate electricity. In the case of fossil-fuel or nuclear power plants, health impacts can be considered exclusively as a direct consequence of energy development, but for hydropower the situation is somewhat different. Since very few water development projects have been constructed solely for hydroelectric generation, it can be argued that the health impacts of hydropower are actually due to the water resources development process itself, and thus it is difficult to attribute them to hydropower, per se. In other words, if a dam is constructed, its health implications will remain very

TABLE II. ESTIMATED PROBABLE HYDROPOWER DEVELOPMENT, 1976–2020

Country groupings	Potential energy in exajoules (1 EJ = 10^{18} J)			
	1976	1985	2000	2020
OECD countries	3.78	4.49	5.37	7.80
Countries with centrally planned economies	0.72	1.20	2.88	8.70
Developing countries	1.17	1.97	4.49	11.80
World total	5.67	7.66	12.74	28.30

similar irrespective of whether hydropower is or is not a reason for the development. It is not always easy, and at certain times it is impossible, to ascribe the different health implications to specific purposes of water development, i.e. irrigation, power generation, navigation, etc.

Conceptually, health implications of hydropower developments can be divided into two broad categories: short-term and long-term problems. Short-term health impacts occur during the planning, construction and immediate post-construction phases during the filling of the reservoirs [3–6]. Longer-term health impacts stem from the presence of large man-made lakes, development of perennial irrigation instead of seasonal irrigation, alteration in the ecosystem of the area, and the changing socio-economic situation.

2.1. Short-term health impacts

Since water resources development projects require major constructions such as dams, spillways, diversion works, etc., they create new employment opportunities, and new workers of different categories move into the areas near construction sites in large numbers. Since dams are often built in remote, undeveloped regions, they lack suitable housing, sanitation and other infrastructural facilities. Even when they are not in remote regions, host communities are seldom able to absorb large numbers of immigrant workers without encountering serious social problems.

In developing countries, labour-intensive technology is often used to construct such large structures, and this invariably means that a large number of unskilled, uneducated labourers arrive at construction sites in search of employment. It is often not possible to satisfy labour requirements locally. For example, during the construction of the Aswan Dam, contractors were forced to transport labourers from neighbouring governorates to the construction sites. Similarly, for the Bhakra-Nangal Project in India, as many as 60% of the workers had to be imported from other provinces [2].

The construction of large water development projects often stretches over a decade, and the daily labour force engaged fluctuates tremendously not only from one year to another but also within any year. For example, the peak labour strength for the Sarda Sahayak Project in India varied from a low of 4000 in 1969–1970 to over 140 000 during 1975–1977. Minimum labour engaged in any year could vary from only 5% to 20% of the peak strength of that year. There is also a high staff turnover.

In these difficult circumstances, and in view of the lack of adequate medical assistance in most developing countries, it is impossible to provide appropriate medical and sanitation facilities to the majority of the workers who, as mentioned earlier, are poor, unskilled and illiterate, and thus lack any political power base. While a certain amount of medical facilities are available for skilled, salaried and educated workers, the unskilled workers are employed primarily as daily labourers without job security and are mostly on their own so far as health services are concerned. In addition, those who live in the new settlement but are not directly employed in the project often do not have access to medical facilities. The presence of large numbers of workers within a limited area, living in unsanitary conditions and without adequate medical facilities, is conducive to the prevalence of diseases like malaria, filariasis, poliomyelitis, venereal and skin diseases, tuberculosis, leishmaniasis (*espundia* of Kala-azar), etc. Furthermore, in some countries, for example Pakistan, groups of labourers exist who travel all over the country working from one construction site to another [2]. Such movements have one important and undesirable side-effect: they tend to increase disease transmission rates.

2.2. Longer-term health implications

As is to be expected, longer-term health implications of hydropower development are more serious than short-term impacts. The longer-term impacts could be broadly classified into two types: (a) introduction of new diseases and/or intensification of existing ones due to the improvement of the habitats of disease-carrying vectors, and (b) health problems arising from resettlement of the people whose homes and land-holdings are inundated by the reservoirs. These two are somewhat interrelated.

2.2.1. Diseases due to improvement of vector habitats

So far as diseases are concerned, probably the most widespread and important one is schistosomiasis. This disease, however, occurs only in tropical and semi-tropical countries and not in temperate regions. Even in countries like China, with a wide variety of climatic regions, schistosomiasis can be observed only in areas where warmer climates are prevalent. While the spread of schistosomiasis is common in most tropical climates owing to dam constructions, the situation becomes far worse if extensive canal systems are developed for irrigation. In other words, increases in schistosomiasis depend more on whether the water development project had irrigation as a primary focus rather than on hydropower per se.

Schistosomiasis is not new: it was known during Pharaonic times. The unprecedented expansion of water resources development, especially the introduction of perennial irrigation systems, has introduced this disease into previously uncontaminated areas [7, 8]. The disease is contracted percutaneously from water infected by cercaria released by snails, and the victims are debilitated by progressive urinary or intestinal infections, reducing labour productivity by some 30–50% [9–10]. The victims also become progressively more vulnerable to other diseases; they face difficult and unpleasant treatments, which are often not available, especially in rural areas of developing countries. The disease is currently endemic in over 70 countries and affects over 200 million people. Prior to development of the present extensive irrigation networks, and when agriculture depended primarily on seasonal rainfall, the relationship between snail host, schistosome parasite and human host was somewhat stabilized, and infection rates were low. Snail populations increased during the rainy season, when agriculture was possible, and this provided the contact between man and parasites. During dry periods, however, there was a lull in infection. With the stabilization of water resource systems due to the development of reservoirs and extensive canal systems, the habitats for snails were vastly improved and extended, and they also had a prolonged breeding phase which substantially increased their population. More human contacts with parasites were provided, which not only raised infection rates but also greatly increased worm load per person. The incidence and extension of these diseases can be directly related to the proliferation of water development schemes, the stabilization of the aquatic biotope, and subsequent ecological changes.

The characteristics of snail habitats, as described by Malek [11], are as follows: *"They breed in many different sites, the essential conditions being the presence of water, relatively solid surfaces for egg deposition, and some source of food. These conditions are met by a large variety of habitats: streams, irrigation canals, ponds, borrow-pits, flooded areas, lakes, water-cress fields, and rice fields. Thus in general they inhabit shallow waters with organic content, moderate light penetration, little turbidity, a muddy substratum rich in organic matter, submergent or emergent aquatic vegetation, and abundant micro-flora."* Thus, water resource developments, especially improvements for hydropower, irrigation or fishing industry, are most likely to favour increased propagation and the spread of these snails [12].

Infection rates of 70% or more can often be observed in certain regions of countries with large irrigation development (e.g. Egypt, Kenya and Sudan). The Lake Victoria area of Kenya is hyperendemic, and the infection rate in schools is up to 100% in certain areas associated with irrigation schemes.

Experience with malaria has been somewhat mixed. There does not appear to be any resurgence of malaria directly associated with the construction of large dams in Africa, but in India, Pakistan and Sri Lanka the incidence of malaria has increased owing to impoundments. Very few in-depth studies are available on the relationship between water development projects and malaria, and this could possibly explain the anomaly. For example, the environmental impact assessments of the Kamburu-Gtaru Dam in Kenya indicate that *"increase in transmission in Kamburu will move malaria from the presently low mesoendemic towards hyperendemic level"* [13].

Malaria was a problem during the early days of the dams operated by the Tennessee Valley Authority in the USA. By fluctuating reservoir water levels by means of carefully controlled draw-downs, mosquito populations were successfully controlled. However, on a global basis, there are over 100 species of mosquitoes capable of carrying malarial or filarial infections like Bancroftian filariasis (elephantiasis) or arboviruses like dengue, yellow fever, viral encephalitis, etc. The different species of mosquitoes often have different behavioural patterns and prefer different types of habitat, so it is not easy to control all disease-carrying mosquito populations in a specific area by any one technique which may have been used successfully elsewhere. Physical, chemical and biological techniques must often be combined with public education and hygiene for any chance of long-term success. In the final analysis, success will depend very much on the availability and use of knowledge of local conditions.

Gambian trypanosomiasis (sleeping sickness), transmitted by tsetse fly, is prevalent in West Africa. The tsetse fly prefers light forests, which often tend to lie along watercourses. New reservoirs, especially those with irregular shorelines, often increase forest areas and thus provide suitable habitats for tsetse fly to flourish. African trypanosomiasis is a debilitating disease which often proves fatal to both humans and animals. Control of trypanosomiasis is not easy, and it is made more difficult by free movement of people and animals from contaminated to uncontaminated areas.

In contrast to diseases discussed earlier, hydropower developments tend to reduce the incidence of onchocerciasis (river blindness). The intermediate host, *simulium* spp. (blackfly), tends to breed in fast-flowing reaches of rivers with turbulent, and thus well-oxygenated, flow. These areas are often drowned by the construction of dams. However, special care must be taken to ensure that new breeding grounds do not develop, especially immediately downstream of the spillways with fast-flowing waters. Construction of the Volta Dam destroyed the blackfly breeding ground that existed upstream. However, blackfly vectors continue to infest many tributaries, and the benefits initially expected from the construction of the Volta Dam have not yet materialized. Furthermore, the vector can travel as far as 300 km, whereas it was formerly thought to be able only to travel 50 km. This naturally increases their sphere of influence.

2.2.2. Health implications of resettlement

One of the major social problems created by large-scale hydropower developments is the displacement of local inhabitants owing to extensive inundations. The Volta Dam, for example, inundated an area of about 3275 square miles and the resulting lake has a shoreline of over 4000 miles. Consequently some 78 000 people and more than 170 000 animals had to be evacuated from over 700 towns and villages of varying population. Eventually 52 new settlements were developed to house 69 149 people from 12 789 families [1].

Resettlement planning for large dams and, what is more important, the implementation, have seldom been successful in developing countries. Most of the sites selected for resettlements are not ready when the settlers arrive, and lack of potable water and sanitary facilities force people to use lake or river water which could be contaminated. People often store water near dwellings for convenience, and this could become potential breeding grounds for mosquitoes which are carriers of numerous diseases. Medical facilities are often non-existent, and people, mostly illiterate and often nomadic, are unaware of the basic precautions necessary for health. Theoretically, the health of settlers in the new environments should be better than before they were evacuated, but in reality conditions generally turn out to be worse than before. There are several reasons for this:

(a) People are frequently moved from rich, alluvial farming land to areas with less desirable soil and sources of water. Social and institutional infrastructures often do not exist and it is therefore difficult to obtain fundamental agricultural requirements such as seeds, fertilizers and pesticides. (For example, ten years after the resettlement of the refugees from the Volta Dam, their agricultural productivity did not equal the pre-impoundment level.) Agricultural yields diminish and become inadequate. Whereas in earlier locations diet could be supplemented with fish, a common and important source of protein, new areas are often far from water bodies and this source disappears. The settlers are sometimes not familiar with local wild food, and when faced with food shortages they experiment with new types. This was probably what caused the 'Lusitu tragedy' of 1959, when 53 women and children who were resettled owing to the construction of the Kariba Dam were poisoned. World Food Programme had to step in at the Volta Dam and the Aswan Dam to alleviate widespread sufferings.

(b) Relocated people may have no resistance to diseases prevalent in the areas where they are being resettled, and may become more susceptible to certain new forms of diseases.

(c) When people are relocated to totally unfamiliar environments they suffer considerable psychological stress. For example, the 57 000 members of the Tonga tribe who were moved owing to the construction of the Kariba Dam suffered great cultural shock when thrust into communities as different from their own as theirs from Great Britain [14]. Two years were required to clear sufficient land to meet even their subsistence needs. The government had to intervene and establish grain stores to avert famine and very serious hardships. Ironically, this well-intentioned step became one of the most destructive parts of the process: the grain distribution centres became transmission sites of the dreaded sleeping sickness disease.

(d) The displaced population is often neglected by both local governments and international organizations. 'Operation Noah', a much publicized and expensive scheme to resettle only some of the animals threatened with inundation from the rising water levels of Lake Kariba, received more international attention and assistance than the refugees,whose sad plights went unnoticed. Planning and constructing adequate and appropriate housing, provision of basic services, including health, and preparation of land for agricultural development, have seldom been completed in any developing country prior to the arrival of the refugees. In most cases, at least during the initial years, the settlers faced worse facilities than they had enjoyed earlier. In the case of the major African dams, the initial emphasis has so far been placed mainly on constructing 'improved' housing, which has too often turned out to be unsuitable for the people being resettled. Rarely had the infrastructure been developed to provide facilities like health services or means of earning a living. The psychological trauma of enforced resettlement, lack of the type of food to which the evacuees had been accustomed for generations, local resentment of the newcomers, breakdown of social order, and exploitation of the settlers by government officials and local people, all ensured that the refugees were under multidimensional stress for a prolonged period, which in certain cases has extended beyond a decade. In such situations, the people were more susceptible to disease, and consequently their health suffered.

In some cases the situation was better. In Sudan, for example, soon after the relocation of the inhabitants from old Wadi Halfa town to New Halfa owing to the rising water level of Lake Nasser, their general state of health improved. One side-effect was an increase of approximately 30% in the birth rate, which was due partly to better living conditions, more food, and improved medical care, and partly to the return of many absentee husbands to settle in the area [15]. Such improvements, especially during the early years, are somewhat uncommon in most developing countries.

3. CONCLUSIONS

Health implications of hydropower development have thus far been a neglected subject of research. While several studies are available on the overall environmental impacts of hydro-dams, not much in-depth work has been carried out on their impacts on the health of both humans and animals. The general assumption so far has been that the health impacts of major dams are small compared to other social and environmental impacts. However, future studies may very well reverse this assumption.

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DISCUSSION

K.G. VOHRA *(Chairman)*: Have you made any quantitative estimates of the health detriment associated with hydroelectric power generation in terms of, for example, fatalities per 1000 MW(e)? This has been done for other methods of generating electricity.

A.K. BISWAS: Not much research has been conducted into the health risks associated with hydropower development, and this is partly due to an erroneous belief that such health risks are minimal and hence can be neglected. Most studies deal with health risks arising from the development of nuclear or fossil fuel plants.

The information available world-wide on the question of health hazards due to the failure of hydroelectric dams is very sketchy.[1] While we are aware of most of the problems created by spectacular failures and/or overtoppings of hydroelectric dams, we have very little information on minor failures, which, as one might expect, are far more numerous than the former. In many cases it is not even reported whether the dams concerned generated hydroelectric power! There has been a tendency in the past to aggregate all the available information on dam failures and then determine the failure risks as if they were all statistically independent events. This is mainly because some of these studies have been conducted by people who have very little knowledge of the design of dams and their behaviour after construction. Such estimates are unreliable and would not even provide 'ball-park' estimates of risks.

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CARCINOGENIC POTENTIAL OF VARIOUS ENERGY SOURCES

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Abstract

CARCINOGENIC POTENTIAL OF VARIOUS ENERGY SOURCES.

Evaluation of the health impacts of different sources of energy should include a comparison of the potential carcinogenic effects of the radioactive and chemical substances produced by various sources. In general, these potential health effects are too small to be measured directly and are therefore estimated by extrapolation, on the basis of a linear dose-response model, from measurable effects at high dose levels. Estimates of the carcinogenic potential of various energy sources available in North America are given in this paper. For most if not all of the energy sources for which data are currently available, it would appear that the known biological benefits in terms of life expectancy greatly outweigh all the potential harm due to carcinogenic (and genetic) effects on human beings, when expressed in the same terms, i.e. life expectancy.

INTRODUCTION

Comparisons of health effects of various energy sources (cf.1,2) have in the past included potential carcinogenic and genetic effects of exposures to ionizing radiation associated with nuclear power but have usually failed to include an assessment of carcinogenic effects associated with other energy sources. This paper provides calculated estimates of the potential effects of some of these other sources.

1. HEALTH RISKS OF ENERGY-DERIVED POLLUTANTS

(a) Ionizing Radiation. Potential health effects of ionizing radiation have been thoroughly reviewed by many scientists and scientific committees. It is generally agreed (3-8), first, that a linear dose-response model should be adopted for health protection purposes; second, that the potential risk of fatal cancer is probably in the region of 100-200 cases per 10^6 persons exposed to one rem of whole-body radiation or in the region of 100-500 cases per 10^6 persons for inhalation of radon daughters

TABLE I. CARCINOGENIC RISKS OF POLLUTANTS ASSOCIATED WITH ENERGY PRODUCTION

Carcinogenic source	Risk estimate for fatal cancers	Average current public exposure	Estimated deaths per 10^6 persons per year	% of normal cancer mortality[a]
Whole-body irradiation	100-200 cases per 10^6 persons per rem	100 millirem per year	10-20	0.5-1
Inhalation of radon daughters	100-500 cases per 10^6 persons per WLM	0.1 WLM per year	10-50	0.5-3
Inhalation of chemical combustion products	10-40 cases per 10^6 persons per year per 1 ng/m^3 BaP[b]	2 ng BaP per m^3	20-80	1-5

[a]Normal cancer mortality 1750 per 10^6 persons per year. [b]For continuous inhalation of air with a mixture of combustion products containing 1 ng of benz-a-pyrene (BaP) per m^3.

equivalent to one working level month (WLM) (Table I). The average background exposure to whole-body radiation from natural sources is generally taken to be about 100 millirem per year (3-5). Average doses received from exposure to radon daughters are less certain; with correction for breathing rates applied, the average exposures of members of the general public to radon daughters would appear to be in the region of 0.1 WLM per year (3,9,10). Using the above values and assumptions, potential cancer deaths due to these sources can be calculated (Table I). Estimates of genetic risk are also available (3-6; cf.11).

Two cautionary notes apply. First, a number of scientists believe that the health hazards resulting from inhalation of low levels of radon daughters by the general public do not exceed the lower limit of the range of risk estimates given above (12). Second, the above values are potential risks calculated by linear extrapolation from known health effects produced by much higher exposure levels. Low levels of radiation have not been shown to produce any demonstrable health effects (5, cf.13,14).

(b) Combustion Products. Less attention has been paid to quantitative assessment of the potential carcinogenic effects of low levels of chemical combustion products, although these may be appreciable (15-17). The chemical products of concern include well-known carcinogenic agents such as benz-a-pyrene (BaP) and other polycyclic aromatic hydrocarbons. In addition, they include a number of other agents which are known to produce genetic changes in lower organisms and are therefore suspected to be potentially carcinogenic in mammals; nitrosamines, dimethylsulphoxide, nitrite (from NO and NO_2), and bisulphite (from SO_2) fall in this category. Combustion products further include assorted hydrocarbons and other agents which, although not necessarily carcinogenic by themselves, can act very effectively as tumor promoters (18,19).

Despite this knowledge, a problem remains which has been concisely summarized by Pike (20): "The air of our cities contains varying amounts of substances known to cause cancer in experimental animals. Tars collected from the air are carcinogenic to animals and transform cells in culture. It appears reasonable, therefore, to assume that this air is carcinogenic to us, in particular to our lungs. The problem arises when we want to quantitate the effect...."

One approach to this problem relies upon the use of BaP as a relative index of the total carcinogenic potential of the products in question. Since the relative concentrations of other carcinogenic and promoting agents in combustion products from different sources may vary widely, it is to be expected that risk estimates based solely upon concentrations of BaP could show considerable variation. However, the values derived by this method generally suggest that continuous inhalation of all the combustion products in air that contains 1 ng BaP per m^3 would potentially produce about 10-40 fatal cancers per 10^6 persons per year (15,20,21).

There is no convincing evidence that low levels of combustion products cause a measurable excess of fatal cancers (22). The relationship between exposure to cigarette smoke and number of induced lung cancers (15,23) is compatible with a linear dose-response model. Moreover, the evidence for or against the linear hypothesis for carcinogenic potential does not appear to be any stronger in the case of combustion products (cf.24) than it is in the case of ionizing radiation. The committees of the U.S. National Academy of Sciences that considered these two sources of potential cancers in 1972 (4,21) adopted the linear dose-response model in both cases for health protection purposes.

The combustion products in the average urban atmosphere (21) would thus potentially be responsible for 20-80 fatal cancers per 10^6 persons per year in North America (Table I). Similar values can be derived using a more general model for comparison of the health effects of cigarette smoke and of combustion products in the urban atmosphere (25) on the assumption that 27% of total premature deaths are due to fatal cancers (26). (These atmospheric combustion products result from a total energy consumption of about 10 kilowatts equivalent per person in the U.S.A. (27); thus the above value corresponds to a cancer death increment of 2–8 per 10^6 persons per year per continuous kilowatt energy used.)

2. POTENTIAL CANCER RISKS ATTRIBUTABLE TO VARIOUS ENERGY SOURCES

(a) Nuclear Power. The average radiation dose commitment to the general public living within 80 km of a light-water reactor under normal operating conditions is reported to be about 10 man·rem per GW(e)·year (29,30). On the basis of a linear dose-response relationship, this dose increment could potentially cause 0.001–0.002 additional fatal cancers.

A different result is obtained when we consider the worldwide effects of radioactive materials from a nuclear power plant, allow for occupational exposure and for exposure increments due to the mining and milling of uranium, to transportation and manufacture of fuel, to reprocessing of fuel, to accidents and to disposal of used fuel. Under these circumstances, the estimated total population dose commitments are in the region of 5000 man·rem per GW(e)·year (3,28). In other words, in a situation where each person in the world received 1 kilowatt of electricity continuously from nuclear sources, and this situation had existed for about 500 years so that the short-lived (but not long-lived) radionuclides from this source had reached an equilibrium level, the average increment in radiation dose to each person would be about 5 millirem per year. This increment could potentially increase the normal incidence of fatal cancers by 0.5–1 fatal cancers per 10^6 persons per year.

(b) Coal-fired Power Stations. The maximum individual radiation dose to a person living in the neighbourhood of a coal-fired power station is similar to that for persons living in the neighbourhood of a nuclear power plant with the same power output (29,30). Previous calculations of the total population dose commitments from air-borne releases of uranium and thorium decay products suggest values in the region of 10–10 000 man·rem per GW(e)·year (3,29,30) depending upon the site of the station and the amount of radioactivity released with the fly-ash. A recent detailed analysis of collective effective dose equivalent commitments resulting from the operation of a coal-fired power station in England suggests a value of about 1000 man·rem per GW(e)·year when the dose commitment (from air-borne sources only, ignoring potential leaching from ash-piles) is summed up to 500 years later (31). In that analysis, station site had little influence upon the result. This latter value suggests that the average increment in radiation dose in a situation where each person in England received 1 kilowatt of electricity continuously from coal-fired power stations would be about 1 millirem per year, which might cause 0.1-0.2 additional cancer fatalities per 10^6 persons per year. Most of this increment in radiation dose is considered to be due to immediate inhalation of radionuclides in the plume from the power station (31) and would therefore not alter appreciably over the 500-year period.

Chemical combustion products produced by a coal-fired power plant contribute greater carcinogenic potential. We estimated the magnitude of this potential by determining the contribution made by coal-fired plants to the total production of various atmospheric pollutants with a potential carcinogenic or tumor-promoting action (Table II). This approach would suggest that coal-fired power plants are responsible for approximately 3% (range 0.14-65%) of all fatal cancers caused by air pollution with chemical combustion products (Table I). Since coal-fired power plants produced about 0.45 kilowatt of electricity per person in the U.S.A. in 1976 (35), the chemical combustion products from coal-fired power plants producing 1 kilowatt of electricity per person continuously would then be potentially responsible for 1-5 additional fatal cancers per 10^6 persons per year. The wide range of uncertainty in this estimate (20-fold in both directions) should however be emphasized. The health effects of various coal-conversion technologies would appear to be roughly similar to those of coal-fired power plants (42).

(c) Other Fossil Fuels. The amounts of radionuclides released during the combustion of oil are low, probably about 1% of those produced by combustion of coal, and are offset by the dilution of carbon-14 (3,31) in the biosphere that accompanies the combustion of any fossil fuel. Natural gas contains varying concentrations of radon which depend upon the particular gas-field of origin and upon the time in transit to the site of utilization (3). Assuming that an electrical power station utilized natural gas containing 10 pCi radon-222 per litre, the estimated release to the atmosphere would be about 25 Ci per GW(e)·year which would result (cf.30,31) in an effective dose equivalent of about 10 man·rem to the surrounding population; this small value would again be nullified by the concomitant dilution of carbon-14 in the biosphere.

The potential effects of the chemical combustion products resulting from the utilization of organic fuels are of more immediate concern. The amounts of BaP and sulfur oxides produced per GW·year of heat under various circumstances are summarized in Table III. The values suggest

TABLE II. CONTRIBUTION OF COAL-FIRED ELECTRICITY GENERATION TO TOTAL ESTIMATED PRODUCTION OF ATMOSPHERIC POLLUTANTS IN THE U.S.A., 1976

	Total suspended particulates[a]	SO_x[a]	NO_x[a]	Hydro-carbons[a]	BaP[b]
Total emissions, tonnes/year	13.4×10^6	26.9×10^6	23.0×10^6	27.9×10^6	482
From coal-fired stations:					
tonnes/year	3.2×10^6	17.6×10^6	6.6×10^6	0.1×10^6	0.7
% of total[c]	23.9	65.4[d]	28.7	0.36	0.14[e]

[a]From (32). [b]From (33). [c]The geometric mean of the contribution of coal-fired electricity generation to total SO_x and total benz-a-pyrene emissions is 3%. A similar value, 4.7%, is obtained for all five pollutants listed. The use of the geometric mean of 3% as an index of carcinogenic potential may be an underestimate, since excess lung cancers due to air pollution appear to be most highly correlated with SO_2 levels (40, 41) and since SO_2 is the pollutant to which coal-fired power stations contribute the largest percentage. [d]Assuming that the major contributor to the 17.6×10^6 tonnes SO_x emissions (32) for electric utilities is coal. We independently calculated from references (34-36) and (39) that coal-fired power stations accounted for 50.3% of all potential SO_2 emissions. [e]For benz-a-pyrene, we independently calculated that coal-fired electrical utilities accounted for 0.028-0.56% (geometric mean, 0.13%) out of a total estimated emission of 736 tonnes (1976) from data in references (21) and (35-37). A similar calculation based on relative polycyclic organic matter (POM) emissions (38) yielded a figure of 0.4% for the contribution of coal-fired power stations to total POM emissions.

that the potential carcinogenic effects resulting from combustion of oil for various customary purposes (production of electricity, heating, transportation) would be similar to those suggested to result from the chemical combustion products from a coal-fired power station (see section 2b). Natural gas appears to be less carcinogenic than either of these two energy sources per unit heat produced. However, the carcinogenic potential from the combustion of coal in hand-stoked stoves and furnaces is probably much greater per unit of heat produced than that from combustion of coal in a power station (Table III).

TABLE III. AIR-BORNE EMISSIONS OF CHEMICAL COMBUSTION PRODUCTS FROM VARIOUS SOURCES

Source	Emissions per GW·year of heat[a]		Approximate index of relative potential carcinogenicity[b]
	BaP in tonnes	SO_2 in 10^3 tonnes	
Natural gas	0.002 (0.0006-0.006)	≦ 0.1	0.03
Coal-fired power plant	0.003[c] (0.006-0.012)	30-60	1
Oil for heating and for power	0.009 (0.003-0.027)	18-36	1.3
Motor vehicles	0.015 (0.002-0.12)	4 (0.6-10)	0.7
Wood stoves	1-2	0.5-2	3
Hand-stoked residential coal furnace	≦ 70	60	150

[a] BaP emissions calculated from data in (21) and (37) and SO_2 emissions from data in (34-36) and (39). Values based on total heat content of the fuels (about 20-60% of the total heat content is useable). [b] Derived from the geometric means for emissions of BaP and SO_2, with emissions from coal-fired power stations being assigned a value of one. [c] Might be as high as 0.1-0.9 if BaP emissions attributed to coal-refuse burning (21, 37) were included; no appreciable increase in SO_2 emissions from this latter source (32).

(d) Reduced Ventilation. Increased insulation and reduced ventilation of buildings are currently under active consideration as important means of energy conservation in many parts of North America. Although there may be minor increments in carcinogenic potential associated with the manufacture of additional insulation, and with formaldehyde or other chemicals diffusing out of the insulating materials, the major concern would appear to be the increase in radon daughter concentrations in the air inside buildings due to decreased ventilation rates (43,44).

TABLE IV. HEALTH EFFECTS OF CHANGES IN RADON DAUGHTER CONCENTRATIONS INSIDE BUILDINGS DUE TO REDUCED VENTILATION

Decreased ventilation air changes per hour	Increase in WLM for 6-month period[a]	Potential increase in fatal cancers per 10^6 persons[b]	kW(e)•year energy saved per person[c]	Potential fatal cancers per 10^6 persons per GW(e)•year saved[d]
2.0 to 1.0	0.038	3.8	0.6	6
1.0 to 0.5	0.09	9	0.3	30
0.5 to 0.2	0.30	30	0.18	170
0.2 to 0.1	0.48	48	0.06	800

[a]From (43). [b]Data calculated assuming 10 fatal lung cancers per 10^6 persons per 0.1 WLM. [c]Assumptions: 100 m^3 per person, 5000°C•days of heating over 6 months (typical of Ottawa, Canada), and maintenance of 40% relative humidity. Under these conditions, 0.6 kW(e)•year per person is required to provide a ventilation rate of one air change per hour during the winter season. Calculations based on energy consumption suggest that the average home ventilation rates in the Ottawa area during the winter months may currently be closer to 0.3 than to 1 air change per hour. [d]Since increased WLM depends only on ventilation rates, these values would be five times greater in an area that required only 1000°C•days of heating in the winter season. Assuming that the efficiency of a heat pump for air-conditioning is twice that of resistive heating, the values in the last column would be two times greater for 5000°C•days of cooling and ten times greater for 1000°C•days of cooling during summer months.

Energy saved by reduced ventilation was calculated as shown in footnote "c" of Table IV. Data on radon daughter concentrations in buildings at different ventilation rates are rather scarce. We have assumed, first, that the average annual exposure at a ventilation rate of one air change per hour is about 0.14 WLM after correction for occupancy time and breathing rates, as reported for buildings in England (44); second, that the risk of fatal lung cancer is 100 cases per 10^6 persons per WLM. Calculations based on the above assumptions are summarized in Table IV. In a climate requiring 5000°C•days of heating during winter months, a decrease in ventilation rates from 1 to 0.2 air changes per hour during the winter season would be responsible for approximately 80 potential lung cancers per 10^6 persons per GW•year saved. These values would be five times greater if the risk of fatal lung cancer were 500 rather than 100 cases per 10^6 persons per WLM. On the other hand, the values would be about five times smaller if the average exposure to radon daughters in wooden frame buildings such as those commonly constructed in the colder areas of North America were about 0.1 WLM at a ventilation rate of 0.3-0.5 air changes per hour (cf.(10), (45), and footnote "c" of Table IV). The potential carcinogenic consequences of reduced ventilation also apply to a building sealed to achieve lower energy consumption for air-conditioning purposes (see footnote "d" of Table IV).

(e) Renewable and Other Energy Sources. All sources of energy are associated with some risk in terms of probability of accidents and disease resulting in injury or death; preliminary attempts to quantitate these risks have been made (cf.1,2). However, data on the potential carcinogenic effects of renewable and other energy sources are unreliable at present. An indication of potential effects can be derived from data on SO_2 emissions associated with the manufacture of steel and other components required for these particular energy sources and with the backup energy supplies that would be required in many cases(2). Since these SO_2 emissions are mainly associated with the combustion of coal, the values can be directly related to those derived in section 2b. This comparison suggests that the potential carcinogenic risk caused by the combustion products associated with the use of sunlight, wind or methanol as energy sources would be roughly 30 times lower than those associated with coal-fired power stations per unit of useable energy derived (Table V). The potential carcinogenic effects would thus be a minor fraction of the other predicted health effects of solar, wind and methanol energy sources (2).

Two other sources of potential carcinogenic hazard might be noted. The production of electricity from geothermal steam in northern California appears to liberate appreciable amounts of radon and of sulfur-containing gasses (30); the potential carcinogenic hazards of these emissions appear to be high per GW·year compared to those from coal-fired or nuclear power stations. The use of masonry or rocks to serve as heat reservoirs inside houses has been advocated; this course of action would generally tend to raise the concentration of radon daughters in these buildings unless special precautions were taken to avoid this effect.

3. SUMMARY OF RISKS AND COMPARISON WITH BENEFITS

(a) Risks. Table V provides estimates of carcinogenic risks per GW·year (i.e. the amount of energy consumed in one year by one million persons each using one kilowatt continuously). For simplicity in comparison, each of the various sources was regarded

TABLE V. CARCINOGENIC RISKS PER GW•YEAR OF ENERGY

Source of energy	SO_2 emissions in 10^3 tonnes[a]	Man•rem[b]	Potential cancer deaths
Energy saved by reduced ventilation in cold climates	-	-	30 (6-170)[c]
Coal-fired power stations	100	1000	2 (0.2-40)[d]
Oil-fired power stations	37	0	0.8 (0.04-16)
Nuclear (LWR) power stations	∿ 0.1[e]	5000	0.8 (0.5-1)
Solar, wind or methanol	1-7	10-70	0.1 (0.004-1.4)
Natural gas power station	∿ 0.1[e]	0	∿ 0.06[f]

[a]From (2). [b]From (29-31) as discussed in the text. [c]From Table IV. [d]This value includes 0.1-0.2 potential cancer deaths from radionuclides plus about 2 potential cancer deaths from chemical combustion products (see section 2b). [e]Associated with the manufacture of transmission lines (2). [f]An approximate value derived from Table III.

as the sole source of energy for each person over a prolonged period of time.

The data given in Table V represent rough approximations. We emphasize that they are theoretical potential risks calculated on the basis of a linear dose-response model using the particular assumptions stated, all of which refer to current conditions. The levels of radiological and chemical carcinogens in effluents from power plants can be reduced, at considerable cost. The relatively high risk per GW•year due to accumulation of radon daughters inside buildings with reduced ventilation can be decreased in a number of ways (e.g. by the use of heat exchangers, air filtration systems, selection of building materials with low radium content, etc.). If such activities were undertaken after careful analysis of relative costs and benefits, an appreciable change in the data base would be anticipated.

(b) Benefits. The preceding discussion has dealt with estimates of carcinogenic risks associated with the by-products of energy production. To place these risks in perspective, it is essential to consider some of the health benefits due to energy production.

The average life expectancy in technologically advanced countries (∿ 73 years) is about twice that observed in many areas of the world that have not yet developed abundant energy supplies and hence cannot afford the concomitant standards of health care, nutrition, housing, transportation and education (27). Much of this increase, i.e. 35 years in life-span, has occurred within the last 150 years (46) and can probably be attributed to accumulated technological development during this time. Even within a given country (such as Canada, the U.S.A. or UK), the average life-span is still dependent upon socio-economic status (47,48), which is presumably related, at least in part, to level of energy consumption. There is thus little doubt that cheap and safe supplies of energy play an essential and beneficial role in longevity.

Increases in life expectancy comparable to those in North America have, however, been achieved in other technologically-advanced countries where energy consumption per person is one-quarter of that in North America (27); below this level, adverse health effects (e.g. increased infant mortality) are observed (Fig. 1). If one ascribed 10% (about 3.5 years) of recent increases in life-span solely to energy availability, this value could be compared with the loss of life expectancy caused by power production (cf.49).

Calculations of loss of life expectancy due to cancers induced by an increment of 5 millirem in average annual radiation exposures suggest an average potential decrease of about 0.001 years (50). This decrease might be approximately doubled if genetic diseases in subsequent generations were included (3,6). Assuming that this increment would be a result of providing each person in the world with 1 kilowatt continuous power from nuclear power, the total potential decrease in life-span under these circumstances would then be about 0.002 years. If nuclear power could supply all present energy requirements in North America (10 kilowatts per person), the total potential decrease in life-span would not exceed 0.02 years, or roughly 1/200th of the postulated health benefit in terms of increased life-span attributed to energy production.

Corresponding values for other energy sources are roughly proportional to their carcinogenic risks as given in the last column of Table V. These values would not be altered greatly by inclusion of potential genetic defects (cf.11,15) attributable to these energy sources. It should, however, be emphasized that this calculation excludes other health effects, notably those due to traumatic accidents and to non-cancer diseases,

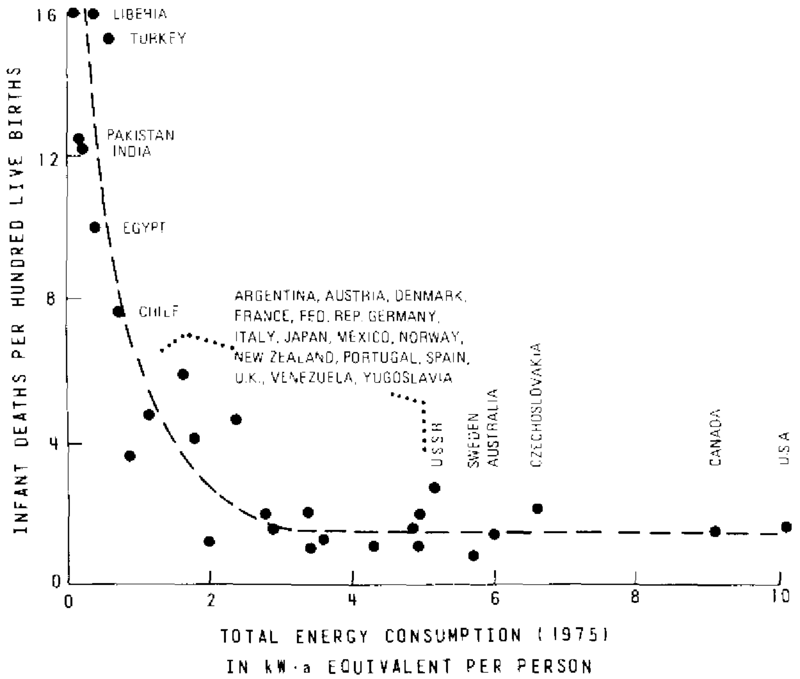

FIG.1. Infant death rates in various countries 1971 to 1975 versus total energy consumption 1975 in kW·a equivalent per person.

associated with energy production; for some sources of energy, these other health effects appear to be more important than those associated with the induction of cancer. Preliminary calculations including all types of deleterious health effects have been reported elsewhere (49). The general conclusion remains similar to that outlined above for nuclear power. For most energy sources, the known benefits in terms of life expectancy greatly outweigh all the potential harm (including carcinogenic and genetic effects) when expressed in the same terms, i.e. life expectancy.

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DISCUSSION

A.A. MOGHISSI: First, I should like to know if you have any data showing SO_2 as a carcinogen. Secondly, I should like to sound a note of caution regarding the use of energy as an indicator of life expectancy and infant mortality. There seems to be an optimum level. Excess generation causes unnecessary exposure while lack of generation causes other problems. Sweden uses about half the amount of energy per capita that the USA consumes, yet has better life expectancy and infant mortality rates.

N.E. GENTNER: Reports by Higgins [40] and by Ford and Bialik [41] indicate that excess lung cancers in humans due to air pollution show the best correlation with SO_2 levels and less correlation with levels of other criteria pollutants. The actual mechanisms involved are uncertain. The fate of SO_2 in the atmosphere has been discussed in the Proceedings of a previous conference.[1] Bisulphite from dissolved SO_2 is a well-known mutagen of low potency which probably affects DNA in much the same manner as does nitrite.[2] According to current theories, bisulphite is therefore a potential carcinogen. Animal studies have failed to demonstrate this suspected carcinogenic action when SO_2 is tested by itself.[3] However, SO_2 does act as a co-carcinogen, which increases the carcinogenic action of benzo(a)pyrene (B(a)P) in animals (data quoted in review by Coffin and Stokinger cited above). Although the mechanisms by which SO_2 exerts a co-carcinogenic action are not yet fully elucidated, these effects, along with those of the numerous other chemical agents in combustion products, need to be considered in any attempt at quantitative estimation of the overall carcinogenic potential of combustion products from various sources. On the basis of the available evidence, it seemed reasonable to use the geometric means of emissions of SO_2 and of B(a)P, a known carcinogen (Table II), as an approximate index of contributions made by coal-fired power stations to the total potential increase in fatal cancers caused by all chemical combustion products. It is to be hoped that this approach can be further refined as more data become available in future.

Dr. Moghissi's remarks on the relationship between energy production and life expectancy do not conflict with the views in Section 3(b) of our paper, where the health risks involved are compared with the health benefits that accrue to people living in a technologically advanced society.

J. SEELIGER: In the Federal Republic of Germany it is the opinion of most physicians and toxicologists that very little is known about the carcinogenic

[1] HAMILTON, L.D., Proc. Conf. Health Effects of Energy Production (GENTNER, N.E., UNRAU, P., Eds), Atomic Energy of Canada Ltd., Chalk River (1980) 149.

[2] SHAPIRO, R., Mutat. Res. **39** (1977) 149; HAYATSU, H., Prog. Nucleic Acid Res. **16** (1976) 75.

[3] COFFIN, D.L., STOKINGER, H.E., Air Pollution (STERN, A.C., Ed.) Vol.2, Academic Press (1977) 231–360.

potential of hydrocarbons emitted by coal-fired power stations. Could you explain the medical basis of your estimates?

N.E. GENTNER: The risk estimates given in our paper refer specifically to continuous inhalation of all the chemical combustion products in air. Two types of hydrocarbons are included in this complex mixture of chemicals. The polycyclic aromatic hydrocarbons include compounds such as B(a)P, which are known to be carcinogenic in animals; levels of B(a)P have been used as a relative index of carcinogenic potential of combustion products in general for humans (see Refs [20] and [21]). Other aromatic and aliphatic hydrocarbons in combustion products act as tumour-promoting and/or co-carcinogenic agents in animal studies. Further information on these agents can be found in Refs [18], [19] and [33], for example.

B. SØRENSEN: It seems to me that in your paper you have failed to specify a number of points. In the case of coal, it was not stated whether fluidized combustion and SO_2 scrubbers were used or not; for indoor air, it was not established whether the structures in question were wood or concrete or whether forced ventilation was used or not; thirdly, in the case of renewable energy, you did not specify alternative backup sources such as hydro and hydrogen power. I further find that your statements regarding a relationship between infant mortality and energy use are totally unsupported.

N.E. GENTNER: As indicated in Section 3(a) of our paper, potential risks were calculated using the given assumptions, all of which refer to current conditions. Emissions data compiled by the OECD [34] and other sources indicate that a major portion of the sulphur in coal is currently emitted into the atmosphere as sulphur oxide during combustion. Some data on impact of future coal-conversion technologies are given in Ref.[42]. Data on radon daughter concentrations in indoor air were calculated from Cliff's data for buildings in England [44]; uncertainties involved in these estimates, including their applicability to wooden frame buildings, are discussed in Section 2(d) of our paper. Certainly one could try to estimate potential carcinogenic risks of renewable energy sources using alternatives such as hydro, hydrogen or nuclear power both for manufacture of components and for backup energy sources; the direction that such calculations might take is suggested by Ref.[2] as well as by Inhaber[4]. However, under current conditions, coal-related emissions do in fact represent most closely the actual situation.

Data in Fig.1 on the relation between infant mortality and per capita energy use are taken from Ref.[27], which would seem to be a reputable source. Further data on infant mortality in earlier societies can be found in Ref.[46][5]. It is fairly clear from these data that the social infrastructure supported by an affluent, industrialized, energy-using society enables important gains to be made in, for

[4] INHABER, H., Science **203** (1979) 718.

[5] See also LOVEJOY, C.O., et al., Science **198** (1977) 291.

example, health care, education and nutrition, resulting in decreased infant mortality and increased life expectancy. Directly or indirectly, the health benefits associated with technologically advanced societies are dependent upon cheap and safe sources of energy. Further information on the relationship between infant mortality and various factors associated with economic development can be derived from the analysis carried out by Sagan and Afifi.[6] This latter analysis also appears to support the above statements on the relation between infant mortality and energy use.

K. SUNDARAM: Your cancer risk estimates for chemicals released from fossil fuels require some elaboration; references should be given to the biological data relating B(a)P to the cancer induction rate. Are these calculations based on animal or human data? This should be made clear in the text of your paper or under "b" of the footnote to Table I.

N.E. GENTNER: The relationship between B(a)P levels and cancer induction has been established both in animal experiments and by epidemiological studies on humans. Risk estimates based on B(a)P as an index of total carcinogenic potential of combustion products for humans are described in detail in Refs [15], [20] and [21] as well as other sources.

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[6] SAGAN, L.A., AFIFI, A.A., Research Memorandum RM-78-41, International Institute for Applied Systems Analysis, Laxenburg (1978).

COMPARAISON DES EFFETS SANITAIRES DE DIFFERENTES FILIERES ELECTROGENES EN FRANCE

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Abstract–Résumé

COMPARISON OF HEALTH IMPACT OF DIFFERENT METHODS OF ELECTRICITY GENERATION IN FRANCE.

The authors present the principal results of a study carried out in France to compare risks associated with the operation of facilities based on the three main methods of electricity generation in current use – the coal, oil and nuclear (PWR) cycles. The comparison was made following a detailed analysis of the fuel cycles utilized in each method. Consistency in the assumptions made in defining the fuel cycles was ensured by using scenarios which take into account forecasts of electricity supply and demand in France up to 1990 (300 TW·h net). Three categories of impact are described: (1) occupational hazards (accidents at work, occupational diseases, theoretical radiological risks); (2) risks to the public evaluated on the basis of estimated data for recurring and accidental wastes and pathways to the environment; and (3) risks to the environment evaluated in terms of the amount of contaminant released.

COMPARAISON DES EFFETS SANITAIRES DE DIFFERENTES FILIERES ELECTROGENES EN FRANCE.

On présente ici les principaux résultats d'une étude menée dans le contexte français sur la comparaison des risques induits par le fonctionnement des installations des trois principales filières électrogènes actuellement utilisées: cycle charbon, cycle fuel, cycle nucléaire (réacteurs à eau sous pression). Cette comparaison a été réalisée après l'analyse détaillée des cycles du combustible relatifs à chaque technique. Les hypothèses utilisées dans la définition des chaînes énergétiques ont été rendues cohérentes par référence à des scénarios qui tiennent compte des perspectives de la demande et de l'offre d'électricité envisagées pour la France à l'horizon 1990 (300 TW·h net). Trois catégories d'effets sont détaillées: 1) les risques professionnels (accidents du travail, maladies professionnelles, risques radiologiques théoriques); 2) les risques pour le public appréciés à partir des données prévisionnelles des rejets continus et accidentels et des voies de transfert dans l'environnement; 3) les risques pour l'environnement évalués en terme de quantité de substances polluantes rejetées.

INTRODUCTION

Les recherches sur l'évaluation des risques environnementaux et sanitaires liés aux technologies électrogènes procèdent d'un double objectif: assurer une clarification des données disponibles en les présentant sous une forme systématique qui permette la comparaison, mais aussi, assurer à terme une meilleure allocation des ressources de sûreté et de protection en dégageant des priorités ou en soulignant d'éventuelles incohérences. En France, le CEPN (Centre d'étude sur l'évaluation de la protection dans le domaine nucléaire) a mené une étude sur la comparaison des risques induits pour le public et les travailleurs par la construction et le fonctionnement des installations des trois principales filières électrogènes actuellement utilisées: cycles charbon, fuel et nucléaire (PWR).

Par ailleurs, une extension de cette démarche aux cas des énergies renouvelables a été réalisée: production d'électricité à partir des centrales solaires thermiques et photovoltaïques.

Tout en s'inscrivant dans le contexte des travaux scientifiques internationaux dans ce domaine, cette étude présente une double caractéristique.

D'une part, elle s'applique spécifiquement au cas d'un pays européen dépourvu presque complètement de ressources fossiles (la France) et à ses conditions géographiques et économiques particulières; rappelons brièvement qu'en 1973 les trois-quarts de la consommation totale d'énergie de la France provenaient de l'importation, que les ressources en gaz naturel ne couvraient que 40% de la demande, qu'aucun gisement important de pétrole ne se trouve sur le territoire français et que ses réserves de charbon sont limitées($\approx$ 1/10000 des ressources mondiales) et exploitées dans des conditions plutôt mauvaises.

D'autre part, elle s'appuie, pour opérer l'évaluation des risques, sur les données qui caractérisent la situation économique. On a ainsi utilisé les méthodes d'analyse macro-économique et les tableaux d'échanges industriels employés dans l'élaboration de la comptabilité nationale française pour le calcul des risques professionnels liés à la construction des installations électrogènes [1].

On se limitera uniquement ici à l'évaluation des risques sanitaires et environnementaux pour le public et les travailleurs liés au fonctionnement normal des filières énergétiques fossiles (fuel et charbon) et nucléaire (PWR) en s'efforçant de comparer ces risques dans le cas d'une production annuelle de la même quantité d'énergie électrique nette.

1. DES «SCENARIOS» ENERGETIQUES COMPARATIFS: UN POINT DE DEPART INDISPENSABLE

Afin de comparer les risques des différentes chaînes énergétiques, il est indispensable d'étudier dans le détail les éléments caractéristiques de chaque

TABLEAU I. SCENARIO DE BASE UTILISE (FRANCE – 1990)

Scénario	Nucléaire	Fossiles
Nombre de tranches	30 tranches de 900 MW_e 28 tranches de 1300 MW_e	Equivalents de 87 tranches de 600 MW_e
Production d'énergie nette (TW·h)	304,5	296
Durée moyenne d'utilisation	5200 h	5670 h

technique, c'est-à-dire pour une technologie donnée, de préciser l'origine des approvisionnements, la qualité des minerais, les distances de transport, la localisation géographique des installations, etc.

Dans le cas du nucléaire, cette tâche se révèle assez simple puisque la notion de «cycle du combustible nucléaire» rend bien compte à la fois de l'ensemble des flux de matières et des transformations industrielles subies par l'uranium depuis son extraction jusqu'au retraitement du combustible irradié, pour une production d'énergie électrique donnée.

En ce qui concerne le fuel et le charbon, le problème se pose en d'autres termes: il faut tenir compte de la multidimensionalité des besoins satisfaits par ces deux chaînes énergétiques et opérer une imputation des coûts sociaux de ces deux filières, au prorata du volume ou des productions de matières premières qui sont destinées à l'usage électrogène.

2. LES SCENARIOS NUCLEAIRE ET FOSSILES

Partant des perspectives d'évolution de la demande et de l'offre d'électricité envisagées pour la France à l'horizon 1990 (tendance «365 TW·h» en 1985) on a défini les scénarios nucléaire et fossiles décrits dans le tableau I.

Le scénario nucléaire comprend le programme électronucléaire (PWR) effectivement engagé depuis 1970 tel qu'il a été défini en 1978.

Les scénarios fossiles sont composés des programmes classiques fuel et charbon susceptibles d'assurer la même production nette d'électricité que le programme nucléaire.

La répartition géographique des diverses installations est fonction des contraintes technico-économiques: approvisionnement en combustibles fossiles, éloignement des lieux de consommation, souci d'aménagement du territoire, etc.;

il y aurait ainsi, pour les centrales, 11 sites «classiques», chacun équipé de 8 tranches de 600 MW_e couplées deux par deux et 15 sites «nucléaires» (25% en bord de mer, 10% en estuaire et 65% en bord de rivière)![1]

Il convient en outre de préciser que les scénarios définis ci-dessus sont conformes aux législations de protection et de sûreté actuellement en vigueur en France et qu'aucune prévision d'évolution de ces dernières n'a été prise en compte dans cette étude. De même, on a considéré les technologies dans leur état actuel sans faire de prospective à leur égard.

3. L'INVENTAIRE DES RISQUES

L'identification des différents risques pour la santé de l'homme et l'environnement naturel présente une part largement conventionnelle due aux multiples incertitudes qui demeurent sur les effets réels. Une certaine pratique générale des études d'impact a ainsi abouti à la quantification d'un certain nombre de risques et de nuisances qui servent depuis de référence sans que ces choix de base soient nécessairement nettement justifiés.

Par ailleurs, face à la diversité extrême des risques envisageables, diverses classifications peuvent être proposées, s'exprimant de la façon suivante:

– les risques spécifiques associés à la fois au fonctionnement en situation normale et en régime accidentel;

– l'ensemble des risques communs pouvant être opposés aux précédents, c'est-à-dire les risques industriels généraux de ces types de production: accidents du travail; problèmes de réchauffement des eaux qui sont de même nature, bruit, rejets chimiques liquides, etc.

On peut encore différencier les risques selon un critère spatio-temporel. On a ainsi, d'une part, les effets diffus, globaux, à long terme, susceptibles d'affecter de très vastes populations. Ces risques comportent souvent des incertitudes notables soit sur leur possibilité de réalisation effective (modification globale de climat liée à la concentration croissante en CO_2 par exemple), soit sur la validité des évaluations quantitatives qu'on opère à leur propos (cas des radionucléides gazeux de longue période et des calculs de dose collective auxquels ils donnent lieu). Et on a, d'autre part, les effets localisés, à court terme et à moyen terme, concernant les personnes du public particulièrement exposées du fait de leur proximité (groupes «critiques») et les travailleurs.

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[1] En ce qui concerne les scénarios fossiles, les importations de pétrole sont ventilées de la manière suivante: Proche-Orient (78%), Afrique du Nord (6,5%), Afrique Noire (7,6%), Mexique (2,8%), divers (5,1%). Celles de charbon: Pologne (10%), Afrique du Sud (25%), Australie (25%), Etats-Unis (20%), Canada (13%), divers (7%).

En ce qui concerne le régime accidentel, on peut distinguer trois catégories de risques:

– les risques catastrophiques, largement évoqués à propos du nucléaire, mais qui peuvent survenir également dans le cas des énergies classiques (par exemple, dans le cas des raffineries ou des accidents de transport par tankers pour lesquels on ne dispose pas toutefois de données quantifiées);

– les risques accidentels mineurs plus courants et qui donnent lieu à des conséquences beaucoup moins graves;

– enfin, le domaine des risques résultant d'actions humaines volontaires (sabotage, par exemple) non quantifiées dans cette étude.

Le tableau II résume l'ensemble des risques considérés généralement à propos des trois chaînes énergétiques étudiées.

4. PRINCIPAUX RESULTATS

Avant de procéder à une présentation synthétique des principaux résultats de l'étude, il convient de rappeler la nécessité d'une certaine prudence dans l'interprétation et l'utilisation des données quantitatives.

Elles sont, d'une part, entachées d'incertitudes plus ou moins importantes selon qu'il s'agit de données relatives à une technologie ayant fait l'objet de recherches ou d'études précises[2]; d'autre part, il faudrait distinguer entre les données fondées sur l'observation directe et celles estimées à partir de modèles théoriques; enfin, elles aboutissent indirectement à accentuer l'importance du connu par rapport à l'incertain.

Or le fait de disposer d'une information riche sur certains aspects du risque signifie d'abord que des efforts particuliers de contrôle et de recherche lui ont été consacrés mais pas nécessairement que ce risque soit important en valeur relative. A l'inverse, la difficulté à quantifier un effet ne doit pas faire négliger son importance comme on s'en rend compte aujourd'hui, par exemple, au sujet de la modification du climat par les émissions massives de CO_2.

Deux catégories de risques sont seulement présentées ici:

– les risques professionnels (accidents du travail, maladies professionnelles, risques spécifiques: radiations et pneumoconioses par exemple);

– les risques pour le public tels qu'ils peuvent être appréciés théoriquement à partir des données prévisionnelles de rejets continus et accidentels et des voies de transfert dans l'environnement.

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[2] Cas des modèles de transfert des rejets toxiques dans l'environnement, les relations dose – effet restant sujettes à controverse.

TABLEAU II. TYPOLOGIE DES RISQUES

Type d'impacts / Chaîne énergétique	Charbon	Fuel	Nucléaire (PWR)
Fonctionnement normal localisé, court terme	pneumoconiose mineurs pollution atmosphérique (centrale)	pollution atmosphérique (centrale)	effluents radioactifs de courte période (public), irradiation et contamination professionnelles (mineurs d'uranium)
diffus, court terme	transfert longue distance pollution des eaux de drainage (mines et préparation du minerai)	effluents liquides hydrocarbures	
long terme	CO^2/climat		effluents gazeux longue période (retraitement) ^{85}Kr, ^{14}C, ^{3}H, ^{129}I déchets longue période effets génétiques
Fonctionnement accidentel	terril de charbon: incendie, rupture de digues accidents de mines	transport de pétrole brut (tanker) accident de raffinerie	sûreté nucléaire (centrale)
Autres risques communs	échauffement des eaux: risques professionnels; occupation du sol; rejets liquides chimiques; bruit.		

L'agrégation des données relatives aux effets sur la santé a été réalisée à l'aide d'un indicateur global élaboré de la façon suivante (en équivalent de jours perdus):
- 1 effet sanitaire radio-induit ≈ 1 décès ≈ $6 \cdot 10^3$ jours perdus[3];
- 1 accident du travail avec arrêt ≈28 jours perdus (moyenne pour la France en 1976) [2];
- 1 maladie professionnelle ≈338 jours perdus (France 1976) incluant la mortalité et les jours d'arrêt de travail) [2];

4.1. Les risques professionnels

Les effets sanitaires annuels sur les travailleurs, lors de l'exploitation pour les trois filières énergétiques considérées et pour un cycle normalisé à 5,67 TW·h soit 1 GW_e X 5670 heures de fonctionnement annuel moyen, sont présentés dans le tableau III.

On constate d'une manière générale que les risques de type classique dans la chaîne nucléaire sont plus importants que les risques radiologiques même en prenant la valeur élevée de $6 \cdot 10^3$ jours perdus par effet radio-induit théorique (mortel).

Compte tenu de la structure économique et géographique d'approvisionnement des matières premières, on note que la chaîne nucléaire concentre 90% des risques sur le territoire national tandis que le résultat est inversé pour les chaînes fuel (15%) et charbon (10%).

Les différences entre niveaux de risque collectif des filières peuvent s'expliquer, soit par des différences entre les effectifs mobilisés pour une même production d'énergie nette, soit par des différences de risque individuel moyen.

4.2. Risques sanitaires pour le public

La comparaison des risques pour la santé publique au voisinage des installations repose, d'une part, sur l'estimation des rejets radioactifs des installations du cycle nucléaire et, d'autre part, sur celle des polluants atmosphériques des centrales thermiques classiques. Dans les deux cas, il s'agit de valeurs théoriques calculées à partir de modèles de rejet, de transfert et de relation dose-effet. Il faut remarquer que l'incertitude liée aux effets sanitaires calculés pour la pollution atmosphérique est beaucoup plus importante que celle associée aux effets radio-induits théoriques.

Le tableau IV synthétise les données sur les effets sanitaires sur le public pour les trois filières normalisées à 5,6 TW·h.

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[3] Il s'agit là d'une surestimation notable s'agissant de cancers ou de leucémies intervenant après des temps de latence importants (jusqu'à une vingtaine d'années).

TABLEAU III. IMPACTS SANITAIRES ANNUELS SUR LES TRAVAILLEURS LORS DE L'EXPLOITATION[a]

	Effectifs travailleurs	Accidents mortels[b] [AM]	Maladies profes-sionnelles[c] [MP]	Effets sanitaires radio-induits théoriques [ES]	Journées perdues par interruption de travail [JPIT]	Equivalents de journées perdues[c] [EJP]	Equivalents de journées perdues par travailleur
PWR	563	0,076	0,46	0,17	1036	2667	4,7
Fuel	232	0,024	0,02		194	345	1,5
Charbon	1623	1,1	1–7	non quantifiés[b]	21 700	28 638–30 666	18–19,3

[a] Cycle normalisé de 1 GW_e pendant 5670 heures de fonctionnement annuel.
[b] Effets de l'uranium et du thorium contenus dans le charbon (mineur).
[c] EJP = (AM × 6000) + (ES × 6000) + (MP × 338) + JPIT.

TABLEAU IV. EFFETS SANITAIRES SUR LE PUBLIC EN FONCTIONNEMENT NORMAL (impact normalisé à 1 GW_e X 5670 heures)

Filière	Population au voisinage d'un site[a]		Mortalité théorique[b]	
	0–20 km	0–100 km	0–20 km	0–100 km
PWR	132 000	848 000	0,0005[c]	0,0031[c]
Fuel	163 500	908100	0,8–3,8	3,9–15,6
Charbon	163 500	908 100	0,42–1,7	2,2–9,9

[a] En France, les scénarios considèrent, pour la filière PWR, 15 sites de 4 tranches de 900 MW_e–1300 MW_e, pour les filières fuel et charbon, 11 sites de 4,8 GW_e. Fuel à 3% de teneur en soufre, charbon à 1%, cheminées de 85 m.
[b] Les décès sont déduits des relations dose-effets de la CIPR (pour les effets radiologiques) et de celles de Hamilton, Lave et Seskin (pour la pollution atmosphérique).
[c] Risque calculé pour les seules centrales; les effets imputables aux centrales ne représentent que 10% de la mortalité totale due au cycle PWR.

TABLEAU V. RISQUE ACCIDENTEL POUR LE PUBLIC (d'après hypothèses rapport WASH 1400)[a]

Conséquences	Risque collectif (annuel)	Risque individuel (annuel)
– Morts immédiates	$1,2 \cdot 10^{-3}$	$6 \cdot 10^{-10}$
– Effets à court terme	$8 \cdot 10^{-2}$	$4 \cdot 10^{-8}$
– Effets somatiques à long terme	$2,6 \cdot 10^{-2}$	$8 \cdot 10^{-10}$
– Cancer thyroïdien	$2,6 \cdot 10^{-1}$	$8 \cdot 10^{-9}$
– Effet génétique[b]	$4 \cdot 10^{-3}$	$0,7 \cdot 10^{-10}$

[a] Transposition directe des données américaines du rapport WASH 1400 au cas du programme français (58 réacteurs; population à 25 km autour des sites: $2 \cdot 10^6$ individus; population totale: $50 \cdot 10^6$).
[b] Taux d'apparition annuel pour la première génération après l'accident.

Du point de vue de la sûreté, le cas des accidents dans les centrales nucléaires est à comparer aux accidents dans les raffineries et aux ruptures de terrils dans les installations d'extraction et de préparation du charbon.

Le tableau V donne une estimation grossière des risques accidentels pour le public obtenus en transposant les données américaines du rapport WASH 1400 au cas français dans le cadre d'un cycle PWR normalisé (304,5 TW·h énergie nette).

5. CONCLUSION

Les efforts menés dans le domaine de la comparaison des conséquences sanitaires et écologiques des différentes filières énergétiques ont suscité des réflexions sur les points suivants:

– l'importance du contexte économique et géographique dans l'évaluation des risques liés à une technologie et dans leur interprétation;

– le rôle des incertitudes caractérisant le niveau de connaissance d'un risque donné, qui tendent à introduire des biais de présentation dans le sens d'une majoration des éléments les mieux connus par rapport à ceux qui sont plus incertains.

– l'hétérogénéité des risques considérés qui rendent délicate toute tentative de synthèse.

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[2] Statistiques nationales d'accidents du travail, Caisse Nationale d'Assurance Maladie, Paris (1978).

DISCUSSION

I.M. TORRENS: Your figures for worker-days lost for oil and coal probably depend rather sensitively on the characteristics for the two normalized fuel cycles mentioned in your oral presentation. It is surprising to note that, for example, SO_x emissions are twice as great for the oil-fired plant (plus refinery) as for the equivalent coal-fired plant. Could you give some more details on the characteristics of the two cycles and other relevant assumptions?

C. MACCIA: The sulphur content in oil is 3% but is only 1% in coal. Both power plants were built without desulphuration systems and thus we found that the SO_x emissions differed by a factor of 2 as a result of the differences in the sulphur content and thermal efficiency of the two fuels. It should be stressed that, apart from the technical factors involved, the origins of the fuel should be included among the sensitive parameters.

P.D. MOSKOWITZ: Could you comment on the uncertainty introduced by assuming an average of 28 working days lost for different occupational sectors with significantly different morbidity experiences?

C. MACCIA: The statistics we used did in fact allow us to determine the number of working days lost in the different industrial sectors. The average figure was simply given for information purposes. Admittedly, some confusion might arise because the value of 6000 working days lost for one death, which appears immediately before the figure in question in the paper, is actually used in the risk index.

P.D. MOSKOWITZ: Could you explain the significant differences between the labour and mortality estimates for the coal and oil fuel cycles in your paper?

C. MACCIA: The difference is simply due to the very high accident rate in mining operations in which most of the coal-cycle workers are involved.

A COMPARATIVE ASSESSMENT OF THE HEALTH IMPACTS OF COAL-FIRED, PEAT-FIRED AND NUCLEAR POWER PLANTS

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Abstract

A COMPARATIVE ASSESSMENT OF THE HEALTH IMPACTS OF COAL-FIRED, PEAT-FIRED AND NUCLEAR POWER PLANTS.

New large condensing power plants must be built in Finland to satisfy the increasing demand for electric power. Possible choices are coal- or peat-fired power plants and nuclear power plants. A study of the health impacts of alternative plants has been made. The total fuel cycle from extraction to waste disposal has been considered, although some phases of fuel cycles of coal and nuclear power are performed in the fuel-supplier countries. Health impacts on the general public from atmospheric emissions of contaminants are assessed for one typical plant site. Risks associated with accidents in fuel transport are assessed on the basis of the present transport situation. A research project in which stack emissions of oil-, coal- and peat-fired power plants are analysed is under way.

1. INTRODUCTION

Alternative building policies of large condensing power plants in Finland include coal- or peat-fired and nuclear power plants. Alternative building programs can be evaluated by estimating their costs and economic impacts or by assessing their health impacts. In a research project completed in 1980 both aspects were considered. In this paper, the latter part of the project is reviewed.

2. ALTERNATIVE BUILDING POLICIES

The total electrical energy consumption in Finland was 40 TWh in 1980. According to forecasts it is expected to increase to 70 to 90 TWh in 2000. The base load is supplied with hydropower, coal and oil-fired condensing plants and nuclear power. Combined heat and power plants and industrial back pressure plants use coal, oil and peat as fuel. To satisfy the growing demand, new condensing power plants as well as combined heat and power plants have to be built.

In 1980 the Technical Research Centre of Finland (VTT) performed a study of the costs and economic impacts of alternative power-plant building programs. The following building policies of new condensing plants were considered:

1) coal power
2) nuclear power
3) coal and nuclear power
4) coal and peat power
5) coal, nuclear and peat power

Plant capacities were fixed so that only 1000 MW(e) nuclear, 500 MW(e) coal and 200 MW(e) peat power plants were considered. Seven alternative building programs were evaluated. Direct and indirect costs of the programs were assessed as well as their impacts on employment and national economy.

3. BASIC ASSUMPTIONS

The economic evaluation was supplemented by a study of the health and other environmental effects of the alternative plant types [1]. The aim of the study was to provide a review of present domestic and foreign knowledge in the field and to compare the health impacts associated with alternative fuel cycles. Previous studies by VTT [2,3] as well as extensive Swedish [4,5] and other foreign studies were utilized.

Health effects can be divided into two categories as regards their dependence on site-specific data. Site-dependent is the dispersion of air contaminants and the resulting collective doses. Less site-dependent are the risks associated with fuel and waste transportation. Site-independent are industrial accidents and occupational diseases in fuel extraction and processing as well as in plant operation.

Because no plant sites were specified in the building programs, only one site was used to assess the site-dependent risks. This site lies on the south coast of Finland, west of the Helsinki metropolitan area. The site is feasible for a coal and possibly for a nuclear plant but not for a peat plant. New nuclear plants would most probably be built adjacent to the existing ones whereas peat plants would be sited in central or east Finland where the fuel peat bogs are. The population density around future nuclear and peat plants would be lower than around the site used in this study.

Population distribution around the site is given in Table I.

TABLE I. CUMULATIVE POPULATION DISTRIBUTION USED IN CALCULATING COLLECTIVE DOSES

circle radius	total population
20 km	25 000
50 km	475 000
100 km	1 150 000
250 km	3 000 000

TABLE II. POLLUTANT EMISSIONS (t/TWh(e))

pollutant	coal plant	peat plant
fly ash	190	220-1400
SO_2	5400	400-4700
BaP	0.035-0.35	1.2

TABLE III. COLLECTIVE POLLUTION BURDENS (man·μg^{-3})

	coal	peat
TSP	9 500	11 000- 70 000
SO_2	540 000	40 000-470 000
BaP	0.35-35	120

TABLE IV. EXCESS FATALITIES PER PLANT YEAR (1000 MW(e))

	coal	peat
SO_2 & TSP	0.6-2.2	0.2-2
BaP	0.006-0.6	2.2

4. HEALTH IMPACTS OF RELEASES IN THE ENVIRONMENT

The following pollution indices were used for coal and peat fired plants

- total sulphur as SO_2
- total suspended particulate (TSP)
- benzo(a)pyrene (BaP)

Emission data of these pollutants for conventional plants were taken from [4] and are given in Table II.

Low sulphur coal with 0.85 % S is used without sulphur removal. Particulate removal efficiency is taken to be 99.5 %. The large variation in fly ash and SO_2 emissions for the peat plant reflects the large variation in the fuel properties. E.g. the sulphur content of peat varies between 0.05 % and 1 %. Some oil is used as auxiliary fuel in the peat plant. Particulate removal efficiency of the peat plant is assumed to be 98 %.

There is a considerable uncertainty in the emission data of BaP. The upper and lower estimates for coal differ by two orders of magnitude in Table II. In 1980 there was no experimental data available for peat plants. The number 1.2 t/TWh is a guess by the authors of [4]. This number, however, was withdrawn from the final version of [4].

Dispersion calculations performed by the Finnish Meteorological Institute for a coal plant with 150-m stack at the site were used to estimate the collective pollution burdens from the emissions in Table II. The plant is assumed to run 6000 h annually with full load. The calculated pollution burdens are given in Table III.

To assess the excess mortality linear dose-effect relationships without thresholds were assumed. For SO_2 and particulates two correlations suggested by Lave [6] were used, for BaP we used the correlations by Carnow and Meier [7]. The calculated numbers of excess fatalities are given in Table IV.

The collective dose from radionuclides in fly ash of a coal plant was assessed adopting the results of a British study [8] to Finnish conditions. We noted the following differences:

- concentration of radioactivity in Polish and Soviet coals is about one third of the value assumed in [8]
- Finland has lower population density
- lower percentage of land area is used in farming in Finland

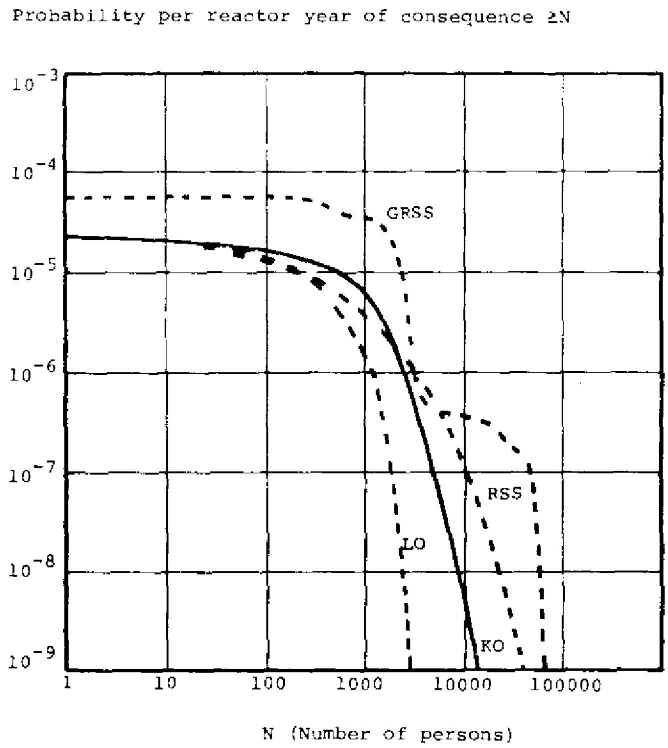

FIG. 1. Complementary cumulative probability distribution for the number of cancer deaths due to reactor accidents.
KO: *Plant site used in this study.*
LO: *Site with existing reactors in Finland.*
RSS: *WASH-1400* [11].
GRSS: *German Reactor Safety Study* [12].

In [8] the collective dose was 5.7 man Sv/(GW(e)a). Taking into account the above differences we conclude that the dose in Finland probably does not exceed 1 man Sv/(GW(e)a).

In the assessment of health impacts in normal operation of a nuclear reactor, statistical data on the releases of a number of US nuclear plants in 1974–1975 were used. A weighted average of BWR median (weight 0.9) and maximum (weight 0.1) releases gave a collective dose of 0.2 man Sv/a. The collective dose was calculated within a radius of 250 km encompassing a total population of three millions. The method of calculation is described in [3].

TABLE V. EXPECTATIONS OF FATALITIES PER PLANT YEAR DUE TO RADIOACTIVE RELEASES (1000 MW(e))

	coal	nuclear power
normal operation		
late effects	0.02	0.004
accidents		
early deaths	-	0.001
late effects	-	0.02

Health risks of radioactive releases were assessed by employing the dose-effect relationship suggested by the ICRP [9]. Collective whole-body dose of 10^4 man Sv is assumed to cause on the average 125 cancer deaths and 80 cases of serious hereditary changes in the descendants (all generations) of the exposed population. Thus, altogether 200 cases of late radiation effects per 10^4 man·Sv are assumed to occur.

Health risks of reactor accidents were assessed utilizing the release categories of [11] with demographic and meteorological data for typical sites in Finland. The assessment includes both the early and the late fatalities. In Figure 1 the cumulative probability distribution of cancer deaths calculated for two sites in Finland are compared with US [11] and FRG [12] results. The expectations of health effects due to radioactive releases from coal and nuclear plants are presented in Table V.

An analysis of the spent fuel storage at the nuclear plant site proved that probability and consequences of severe fuel storage accidents are considerably smaller than for reactor accidents. Spent fuel transportation was assessed to give only a minor contribution (about 2×10^{-3} man·Sv) to the collective dose. According to [10] the final disposal of spent fuel causes a collective dose commitment of about 0.3 man·Sv/(GW(e)a) during those 500 years in which the exposure is highest.

Based on the information in [13,14], radon emissions from uranium mill tailings waste and coal fly ash pile were both estimated to cause a relatively high collective dose commitment. The respective dose commitments were of the same order of magnitude.

TABLE VI. FATALITY RATES IN TRANSPORT IN FINLAND

mode	accidents	rate
ship	accidents aboard	1.4×10^{-2} man-years
rail	level crossing accidents, person run over by a train	2×10^{-6} train-km
road	road traffic accidents	5×10^{-8} lorry-km

TABLE VII. FATALITIES IN TRANSPORT PER PLANT YEAR
(1000 MW(e))

	ship	rail	road
coal	0.08	-	0.01
peat	-	1.9	0.9
nuclear power	-	0.001	-

TABLE VIII. OCCUPATIONAL FATALITIES IN ENERGY PRODUCTION PER PLANT YEAR
(1000 MW(e))

	coal	peat	nuclear
occupational diseases	(2)	not calc.	0.15(0.23)
industrial accidents	0.05(1.6)	0.05	0.01(0.2)

5. OTHER HEALTH IMPACTS

Finnish data on transportation accidents have been reviewed to assess the risks of transporting fuel and waste. The evaluated fatality rates are given in Table VI.

It can be observed that the risks in sea transportation are entirely occupational whereas in land transportation they are primarily public.

The transportation risk assessment was based on the 1980 transportation situation. Coal is transported by ship from supplier countries. About 15 % of the fly ash collected in coal plants is transported to cement factories. Fuel to peat plants is transported (alternatively) by train (average trip lenght 150 km) or by lorry (75 km). Nuclear fuel is transported by ship or by train.

The relatively large number of fatalities in peat transportation is a consequence of the large number of train or lorry loads needed to operate a power plant. The resulting large transportation costs favour sites near the fuel peat bogs. Transportation of fuel peat to such sites would involve smaller risks than those in Table VII. The risk of nuclear fuel transportation in Table VII is almost exclusively caused by conventional traffic accidents.

In [5], data on deaths of miners working in West German coal mines is reviewed. This data was considered representative of the working conditions in the countries which supply coal to Finland. Data on industrial accidents and occupational diseases in the nuclear fuel cycle is taken mainly from [15]. The dose 15 man Sv/(GW(e)a) was employed for the occupational exposure both at a power plant and at a reprocessing plant.

Fuel peat is extracted by the milling method in which about 1-cm-thick top layer is removed from the bog. In certain weather conditions peat extraction is accompanied by copious dust, which occurs in other phases of fuel peat handling, too. In a recent health survey of Finnish peat industry workers with working experience ranging from 1 to 30 years (average 5 years) no cases of occupational diseases of the respiratory tract were discovered [16]. The rate of industrial accidents in peat extraction is low.

Dust explosions have occurred in Finnish peat-fired power plants at the rate of 0.4/plant year [17]. So far these explosions have caused only one death. It is tentatively assumed that the rate of fatal accidents in peat plants is the same as in coal plants.

TABLE IX. FATALITIES (INCLUDING LATE EFFECTS OF IONIZING RADIATION) PER PLANT YEAR (1000 MW(e))

	Coal	Peat	Nuclear
Normal operation			
workers	(2)	not calc.	0.15(0.2)
public	0.6-2.2	0.2-2.2	0.004(0.3)
Accidents			
workers	0.13(1.6)	0.05	0.01(0.2)
public	0.01	0.9-1.9	0.02
Waste			
public	not calc.	not calc.	0.006(0.1)
Total	0.7-2.3(3.6)	1.2-4.2	0.2(0.8)

The estimated occupational fatalities are given in Table VIII. In this table occupational deaths associated with those phases of the fuel cycle which are not performed in Finland are given in parentheses.

6. SUMMARY OF RESULTS

All the results of the previously mentioned health risk assessments are summarized in Table IX. The risk numbers are fatalities per plant year for a 1000 MW(e) power plant. Late effects of ionizing radiation are included in the fatality numbers. The fatalities in fuel supplier countries are given in parentheses.

There is a considerable uncertainty in all the risk figures, expecially the public ones, given in Table IX. The ranges of the assessed public risk figures reflect variations in fuel composition, uncertainties of emission data and the different dose-effect correlations used. However, we have not tried to estimate how the basic uncertainties of the dose-effect relationships (primary agent, functional form etc.) affect the ranges.

The largest contributors to the health risk in Finland are, in the case of coal: sulphur and particle emissions; in the case of peat: polyaromatic hydrocarbon emissions and transportation accidents; and in the case of nuclear power: occupational diseases. For coal and nuclear power plants more fatalities are assessed to occur in the fuel supplier countries than in Finland.

7. MEASUREMENT OF ATMOSPHERIC EMISSIONS FROM POWER PLANTS

At present eight peat fired heat power or back pressure plants operate in Finland. Six plants are in the planning or the construction stage. Until now no experimental data on atmospheric emissions of peat plants has been available. It has been conjectured that PAH emissions from a peat plant are considerably larger than from a coal plant (c.f. Table II).

We have at VTT a research project under way in which stack emissions from Finnish oil, coal and peat-fired plants are analyzed. The concentrations of the following components are measured

- particulates
- SO_2, NO_x, CO, CO_2
- trace elements (including heavy metals, U, Th)
- Po-210, Cs-137, Ra-226
- about 30 PAH compounds

The fuels as well as fly and bottom ash are analyzed for trace elements and radionuclides. The measurements are performed in cooperation with Swedish research institutes which also perform immission measurements around peat-fired plants. Trace elements and PAH are analyzed in water and vegetal samples taken near the plants.

Because PAH emissions are expected to depend upon the combustion process, plants with differing combustion methods were chosen. The projects will be completed in 1982. According to some preliminary results it seems that the guessed BaP emission rate in Table II overestimates the actual rate by one or two orders of magnitude.

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HEALTH IMPACTS OF GEOTHERMAL ENERGY*

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Abstract

HEALTH IMPACTS OF GEOTHERMAL ENERGY.

Geothermal resources are used to produce electrical energy and to supply heat for non-electric applications like residential heating and crop drying. The utilization of geothermal energy consists of the extraction of hot water or steam from an underground reservoir followed by different methods of surface processing along with the disposal of liquid, gaseous, and even solid wastes. The focus of this paper is on electric power production using geothermal resources greater than 150°C because this form of geothermal energy utilization has the most serious health-related consequences. Based on measurements and experience at existing geothermal power plants, atmospheric emissions of non-condensing gases such as hydrogen sulphide and benzene pose the greatest hazards to public health. Surface and ground waters contaminated by discharges of spent geothermal fluids constitute another health hazard. In this paper it is shown that hydrogen sulphide emissions from most geothermal power plants are apt to cause odour annoyances among members of the exposed public - some of whom can detect this gas at concentrations as low as 0.002 ppmv. A risk-assessment model is used to estimate the lifetime risk of incurring leukaemia from atmospheric benzene caused by 2000 MW(e) of geothermal development in California's Imperial Valley. Also assessed is the risk of skin cancer due to the ingestion of river water in New Zealand that is contaminated by waste geothermal fluids containing arsenic. Finally, data on the occurrence of occupational disease in the geothermal industry is briefly summarized.

1. INTRODUCTION

Geothermal energy is a general term that refers to the stored heat of the earth, which can be recovered using current or yet-to-be-developed technologies. At the present time, the only available current technologies are those that extract heat from hydrothermal convection systems; these are systems where water or steam transfers heat from deep parts of the subsurface system to near-surface locations where the fluid may be tapped. Hydrothermal convection systems are subcategorized as vapor-dominated (steam) or hot-water systems. Vapor-dominated systems are the easiest to utilize

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for the production of electricity as the tapped steam can be routed directly to a low-pressure turbine, and the condensate may be used as cooling water.

Hot-water systems exist with a broad range of temperatures. High-temperature (>150°C) systems are usable for the generation of electricity with existing technology; the process is somewhat more complex than for vapor-dominated systems because the hot water must be flashed to produce steam. Intermediate-temperature (~90 to 150°C) systems are usable for other purposes such as space heating and other nonelectric uses.

Other types of geothermal energy are classified as conduction-dominated, igneous-related and geopressured [1]. Conduction-dominated refers to the very low-grade heat that is present everywhere on earth; it cannot be extracted economically at the present time. Igneous-related, often called "hot dry rock," refers to the heat from recent magmatic intrusions that is stored in dry rocks. This is not considered to be a near-term technology, but is an active area of research and development. Geopressured refers to certain geologic zones consisting of highly porous sands saturated with saline water at very high pressure and temperature, and typically at great depths. The value of this resource is greatly augmented by the natural gas that is dissolved in the fluid. The extraction of geopressured energy is not considered to be a demonstrated technology in the sense of heat extraction, but the increasingly valued natural gas content may accelerate development.

1.1. Current status of development

Although geothermal energy is sometimes regarded as a future or advanced energy technology, it has been used for nonelectric applications for centuries and was first used to generate electricity at the turn of the century in Larderello, Italy. Today, the geothermal resource area with the largest amount of development in the world is located at The Geysers, California. Here, the first 11-MW(e) unit was placed in operation in 1960, and today fifteen units with a net capacity of 908 MW(e) are in operation. Hot-water resources have been developed much more slowly in the US, and only 10 MW(e) are currently on-line. Other countries, however, are utilizing hot-water resources effectively; two of the larger power plants are at Wairakei, New Zealand, 150 MW(e),[1] and Cerro Prieto, Mexico, 150 MW(e). During 1979 the total generating capacity of geothermal power plants in eleven countries was

[1] The installed capacity is 192.6 MW(e), but the output in 1974 could be maintained at only 150 MW(e) [2].

TABLE I. Installed electric generating capacity of geothermal power plants in the world during 1979 [2]

Country	Installed capacity (MW(e))
China	1
El Salvador	60
Iceland	32
Italy	420.6
Japan	165
Mexico	150
New Zealand	202.6
Philippines	59.2
Turkey	0.5
U S S R	5
U S A[a]	663
Total	1758.9

[a] An additional 255 MW(e) are now on-line.

about 1800 MW(e) as shown in Table I. Nonelectric uses, in contrast, probably amounted to over 6000 MW(th) [3]. The focus of this paper is on electric power production because we believe that this method of utilization has the more serious health-related consequences.

1.2. Environmental and health-related concerns

There are several environmental issues associated with the extraction of hot water or steam from geothermal reservoirs and the subsequent processing to produce electricity. These include the release of noncondensing gases, the safe disposal of large volumes of spent geothermal fluids, land subsidence caused by the withdrawal of fluids, enhanced seismicity from fluid injection or reservoir cooling, and the production of noise. Based upon measurements and experience at existing power plants, the most serious concerns relate to emissions of gases that are not condensed at operating temperatures and pressures. The composition of these gases varies widely from reservoir to reservoir; however, the major component is typically carbon dioxide, and significant amounts of hydrogen sulfide are nearly always present along with trace amounts of benzene, radon and mercury. Exposure to atmospheric hydrogen sulfide and benzene poses potential hazards to public and occupational health. Public water supplies contaminated by discharges of spent geothermal fluids represent another potential health problem. In addition, exposure to hydrogen sulfide and to toxic chemicals used in hydrogen sulfide abatement systems have been identified as occupational health hazards [4].

We cannot consider all of these concerns in this paper, and will focus only on a few of the more significant. At a later date, we will produce a comprehensive Health and Environmental Effects Document under the sponsorship of the US Department of Energy.

2. PUBLIC HEALTH

The effects of geothermal energy production on public health can range from occasional discomfort or annoyance to premature death. In this section we focus on the public health risks associated with exposure to benzene, arsenic and hydrogen sulfide emitted from geothermal facilities. Long-term exposure to benzene or arsenic can result in cancer, while exposure to hydrogen sulfide can cause personal discomfort. The potential significance of these different health effects is assessed here through the use of case studies involving the development of geothermal resources in the United States and elsewhere.

2.1. Hydrogen sulfide and annoyance

Hydrogen sulfide is a toxic gas [5]; it can cause respiratory paralysis at concentrations above 1000 parts per million by volume (ppmv) and the threshold for serious eye injury is 50 to 100 ppmv. It also has a very offensive odor, and the human nose can detect concentrations as low as 0.002 ppmv. The median odor perception threshold is estimated to be 0.005 ppmv [4]. Furthermore, there are suggestions that chronic exposures to low levels of hydrogen sulfide may produce other health effects, primarily of a neurasthenic nature. For example, residents downwind of power units situated in The Geysers resource area in California have complained of headaches, nausea, sinus congestion, abrupt awakening, etc., when ambient concentrations were roughly 0.1 ppmv [4]. However, there is no evidence that would suggest that hydrogen sulfide is carcinogenic, mutagenic or teratogenic [6].

From a public health standpoint, odor annoyance is the primary effect of hydrogen sulfide emissions. More severe effects are unlikely because atmospheric dispersion of the gas

TABLE II. Uncontrolled emission rates of hydrogen sulfide at several geothermal resource areas

Geothermal resource area	Emission rate (g/MW(e)·h)	Reference
Salton Sea, California[a]	160	[7]
East Mesa, California[a]	55	[7]
The Geysers, California	1 800	[8]
Cerro Prieto, Mexico	32 000	[9]
Wairakei, New Zealand	570	[9]
Ahuachapan, El Salvador	1 580	[2]
Otake, Japan	542	[2]
Matsukawa, Japan	5 050 - 20 800	[2]
Larderello, Italy	14 300	[2]

[a] Located in California's Imperial Valley.

after release from a power plant will normally result in non-toxic ground-level concentrations. Emission rates of hydrogen sulfide from several geothermal facilities in the world are shown in Table II. Even facilities with emission rates below 55 g/MW(e)·h, the lowest presented in Table II, can cause ambient concentrations of hydrogen sulfide that are above the odor detection level of individuals. Ermak et al. [7] have shown that a 100-MW(e) power plant in the Imperial Valley, for example, would have to have an emission rate of 30 g/MW(e)·h in order to prevent hourly concentrations of hydrogen sulfide from exceeding 0.03 ppmv beyond 1 km from the plant. The hourly average concentration of 0.03 ppmv is the California air quality standard for hydrogen sulfide. That standard, however, is above the odor detection level of most people, and therefore emission reductions would be necessary to avoid annoying odors. At The Geysers resource area in California, abatement systems have had to be installed on power plants to limit hydrogen sulfide emissions. Ironically, the operation of these systems can result in occupational health problems.

2.2. Benzene exposure and leukemia

Benzene is a hematotoxin that can cause various blood disorders, including anemia, leukopenia and thrombocytopenia. In addition, it has been identified as a leukemogen. The primary sources of information on the relationship between benzene and leukemia have not been animal studies, but rather epidemiological studies of workers exposed to benzene. Studies by Aksoy et al. [10], involving shoeworkers in Turkey, and Infante et al. [11], involving workers in the rubber industry in the United States, provide strong evidence that the chronic inhalation of benzene can lead to leukemia. Benzene has been detected in the gas phase of geothermal fluids from wells at the following locations: Cerro Prieto, Mexico; Larderello, Italy; and The Geysers and East Mesa geothermal resource areas in the United States [12,13]. The presence of benzene and other nonmethane hydrocarbons in those geothermal fluids is probably caused by the thermal metamorphism of sediments containing organic material [14].

An individual's risk of leukemia due to benzene emissions from a geothermal power plant is a function of personal exposure and the relationship between exposure and the probability of cancer. Stated mathematically,

$$R = C \cdot E \cdot \frac{\chi}{Q} \cdot P \tag{1}$$

where

- R = an individual's lifetime cancer risk resulting from exposure to atmospheric benzene;
- C = concentration of benzene in geothermal fluid, g/kg;
- E = extraction rate of geothermal fluid, kg/s;
- χ = ambient concentration of benzene, $\mu g/m^3$;
- Q = unit emission rate of benzene, g/s; and
- P = lifetime probability of incurring cancer due to exposure to benzene in the atmosphere, $m^3/\mu g$.

An accurate prediction of an individual's risk of leukemia is difficult to obtain because of uncertainties regarding estimates of exposure as well as the dose-response function for the carcinogen being assessed. Therefore, it is helpful to quantify the uncertainties in order to gain a better understanding of how they affect the prediction of cancer risk.

Uncertainty is incorporated into the assessment model by assuming that the parameters are log-normally distributed and they are independent. The uncertainties of the individual parameters can then be propagated to obtain an overall estimate of uncertainty for the predicted cancer risk by

$$\mathrm{Var}(\ell n\ R) = \mathrm{Var}(\ell n\ C) + \mathrm{Var}(\ell n\ E) + \mathrm{Var}(\ell n\ \tfrac{\chi}{Q}) + \mathrm{Var}(\ell n\ P) \tag{2}$$

where, e.g.,

$$\mathrm{Var}(\ell n\ R) = \ell n^2 \sigma_g(R)$$

$$\sigma_g(R) = \text{geometric standard deviation of } R$$

To illustrate the use of the risk-assessment methodology, including the treatment of uncertainty, we analyze the cancer risk of one scenario of future geothermal energy production in the Imperial Valley of California. The Imperial Valley contains five identified geothermal resource areas with a total energy potential of almost 7 000 MW(e) for 30 years [1]. For our analysis of the effects of benzene emissions, we have selected a scenario of energy development in the valley that is based on 500 MW(e) of energy production in each of four hot-water resource areas (i.e. Salton Sea, Brawley, Heber, and East Mesa areas). The prediction of cancer risk resulting from benzene emissions at 2 000 MW(e) of energy production is based on measurements of benzene levels in geothermal fluids from one resource area, estimated extraction rates of geothermal fluids for power plants in each of the resource areas, a simulation of the ambient benzene concentrations in the valley using a multiple-source, Gaussian diffusion model, and a linear dose-response relationship between chronic benzene exposure and an individual's lifetime probability of having leukemia.

Table III presents estimates of the parameters used in the benzene risk assessment. The estimate of C is the geometric mean of benzene concentrations in fluids from two wells in the East Mesa resource area [13]. The value of E was calculated as the geometric mean of extraction rates of geothermal fluids calculated for a typical 100-MW(e) power plant in each of the four resource areas. The estimate of the lifetime probability of leukemia per average lifetime exposure to atmospheric benzene, P, is the geometric mean of estimates of P derived from three separate epidemiological studies [15]. A Gaussian diffusion model was used to calculate the isopleths of the annual ground-level concentration of benzene, χ, per unit emission rate, Q, from five 100-MW(e) power plants in each of the resource areas [7]. The range of values for χ/Q in Table III (i.e. 0.3 to 10.0 $\mu g/m^3$ per g/s of benzene) are for the minimum and maximum isopleths of χ/Q.

For the 0.3 isopleth of χ/Q, which encloses most of the valley, the estimate of R is 1.1×10^{-6}; the value of R calculated from χ/Q equal to 10.0 is 3.7×10^{-5}. These estimates of R represent the incremental risk of incurring leukemia. The total risk of leukemia due to the inhalation of environmental benzene is a function of background levels of

TABLE III. Estimates of the parameters used in the benzene risk assessment and their variances

Parameter	Estimated geometric mean	Estimated geometric standard deviation
C	2.9×10^{-4}, g/kg	6
E	1.7×10^{3}, kg/s	1.8
χ/Q	0.3 to 10.0, $\mu g/m^3$ per g/s	2.5
P	$7.6 \times 10^{-6}/\mu g/m^3$	1.8

benzene plus the incremental levels introduced by geothermal development. Background concentrations of benzene in nonurban areas in the U.S. have been measured to range from approximately 1 $\mu g/m^3$ to 3.5 $\mu g/m^3$ [16], with a geometric mean of 1.9 $\mu g/m^3$. The leukemia risk for the ambient level of 1.9 $\mu g/m^3$ is 1.4×10^{-5}.

The geometric standard deviation, σ_g, for each parameter can be determined graphically or analytically; however, in the absence of sufficient data, it must be determined judgmentally. For example, the measurements of the benzene concentrations, C, in the geothermal fluids were extremely limited, and therefore we assigned a value of 6 to the σ_g of C to reflect the large differences that are often associated with the chemical composition of geothermal resources. A σ_g of 1.8 was computed for the three values of P estimated by Albert [15]. The value selected for σ_g of χ/Q is based on a validation study of Gaussian diffusion models used to predict long-term air concentrations of a conservative pollutant [17]. Finally, we analytically calculated the value of σ_g for the temperature-dependent extraction rates of 100-MW(e) flashed-steam power plants in the four resource areas. The resulting estimate of σ_g for R is 8.8. As a result, the upper and lower limits to the estimated 68% confidence interval for the high prediction of incremental leukemia risk (i.e. at χ/Q = 10.0) are 3.2×10^{-4} and 4.2×10^{-6}, respectively.

The lifetime risks of leukemia calculated for the scenario of geothermal development in the Imperial Valley are less than 10^{-4}. The annual incidence of leukemia caused by the benzene emissions will be small because the population at risk in the valley is less than 90 000 people. To improve the accuracy of the risk estimates, additional data are especially needed on the concentration of benzene in the geothermal fluid from all of the resource areas. Furthermore, the risk assessment model could be refined by including a variable that relates ambient concentrations of benzene to the levels inside dwellings. This type of variable is needed to calculate an exposure to atmospheric benzene that includes the effect of lifestyle (e.g. the time normally spent outdoors versus indoors).

2.3. Arsenic exposure and skin cancer

The most important source of public health risks from geothermal energy production is the emission of noncondensable gases. However, the discharge of spent geothermal fluids is also a potential source of negative public health effects -- particularly when spent fluids containing toxic elements, such as arsenic, are discharged to a water body that is or can be used for drinking water. The chronic ingestion of drinking water contaminated with arsenic can lead to various health effects, including skin cancer [18]. The Wairakei geothermal power plant in New Zealand probably provides the worst example of drinking water contamination from the disposal of geothermal fluids. Since 1964 that plant has discharged its waste geothermal fluids into the Waikato River, which is the source of drinking water for a local town. Under average flow conditions, Axtmann [19] calculated an increase of 39 μg/ℓ in the arsenic concentration of the river because of the geothermal discharges, and noted that, under minimum flow conditions, the incremental concentration could be as high as 250 μg/ℓ. We have calculated the concentration of arsenic in the spent Wairakei fluids, and compare it to concentrations of As in fluids from other resources in Table IV.

In order to estimate the incremental risk of developing skin cancer from the arsenic in the Waikato River, we use the simple model of the USEPA [18]:

$$R' = C' \cdot P' \qquad (3)$$

where

R' = an individual's lifetime risk of developing skin cancer from drinking water containing arsenic;

C' = concentration of arsenic in drinking water, μg/ℓ; and

P' = lifetime probability of incurring cancer due to drinking water containing arsenic, ℓ/μg

TABLE IV. Concentration of total arsenic in geothermal fluids from ten geothermal resource areas

Geothermal resource area	Arsenic concentration (mg/ℓ)	Reference
Wairakei, New Zealand	2.7[a]	
Broadlands, New Zealand	3.3	[2]
Ahuachapan, El Salvador	11.3	[2]
Hatchobaru, Japan	3.2	[2]
Cerro Prieto, Mexico	0.5 - 2.2	[20]
Salton Sea, California	11.	[21]
Brawley, California	2.6	[21]
Heber, California	0.1	[21]
East Mesa, California	0.2	[21]
The Geysers, California	<0.004	[22]

[a] Derived from data in Axtmann [19].

The value of P' has been derived by the USEPA [18], and is 4×10^{-4} per μg/ℓ. It is based entirely upon epidemiological studies of Taiwanese exposed to high concentrations of arsenic in drinking water from artesian wells [23,24]. There is information [18] that indicates that trivalent arsenic was the predominant form of arsenic present in the ground water, instead of the less toxic pentavalent form [25] that is usually expected in natural waters. The Wairakei geothermal fluids could also contain a significant amount of arsenic in the trivalent state because anaerobic conditions in the geothermal reservoir probably would limit the oxidation to the pentavalent form. Data supporting this assumption are from a study conducted by Crecelius et al. [20] of arsenic speciation in the geothermal fluid from a well supporting the Cerro Prieto power plant in Mexico. They found that the arsenic in the geothermal fluid, prior to atmospheric exposure, was 78% trivalent arsenic and 22% pentavalent arsenic. Additional information that corroborates the presence of trivalent arsenic in the Wairakei geothermal fluids is from Coulter [26] who reports that trivalent arsenic represents as much as 70% of total arsenic in sediments in a lake fed by the contaminated Waikato River. Assuming that the speciation of arsenic in the Waikato River is similar to that of the arsenic in the drinking water consumed by the Taiwanese and that there are no significant genetic or dietary differences between the populations at risk, the incremental cancer risk of chronic ingestion of geothermal arsenic dissolved in the Waikato River would be about 1.6×10^{-2}. We estimate values of σ_g to be 2.4 and 2 for C' and P', respectively, and therefore 3 for R'.

This calculated risk is surprisingly high, and we suggest that it should be viewed with caution. As mentioned above, the value for P' was taken from the USEPA [18], and is based upon a linear, no-threshold dose-response model. This, in fact, is almost certainly not the case, as arsenic has been shown to be an essential element [27,28] and mammals also have a detoxification mechanism of methylation [18]. Further, studies on humans indicate that "...arsenic levels in water at concentrations of 100 μg/ℓ or less seem not to produce an undue body burden " [29]. Thus, we conclude that a threshold probably does exist for the production of skin cancer, and that 39 μg/ℓ may be below the threshold level.

3. OCCUPATIONAL HEALTH

Problems of occupational health that result from the geothermal industry have been reviewed previously [30,31]. In general, the more serious problems have historically been with exposure to toxic levels of hydrogen sulfide. More recently, however, attempts to abate the emission of hydrogen sulfide at generating units at The Geysers have resulted in exposure to abatement chemicals and sludges; and the incidence of occupational disease increased markedly in 1976, and then declined [30]. Rough estimates of the incidence of occupational illness in the US are 20 (illnesses per year per 1000 employees) pre-abatement, and 240 in 1976 during the period of initial experience with abatement systems [30].

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DISCUSSION

K.G. VOHRA *(Chairman)*: You have described the main pollutants responsible for human exposure in geothermal fluids. Could you explain how workers and members of the public are exposed to hydrogen sulphide, benzene and arsenic as a result of the utilization of geothermal energy?

D.W. LAYTON: Hydrogen sulphide and benzene are released to the atmosphere as non-condensing gases, and therefore the exposure route is by inhalation. Arsenic, by contrast, remains dissolved in a geothermal fluid (hot water), and if the spent geothermal fluid from a power plant is discharged to surface water used for drinking then the exposure route is by ingestion.

RISK ESTIMATES OF IMPACTS FROM EMERGING TAR-SAND TECHNOLOGIES*

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Abstract

RISK ESTIMATES OF IMPACTS FROM EMERGING TAR-SAND TECHNOLOGIES.

The North American continent has the largest tar-sand resources in the world with approximately 1.3×10^{12} barrels of oil-equivalent in Canada and 3.6×10^{10} barrels of oil-equivalent in the USA. Major tar-sand deposits also exist in Venezuela, Malagasy, Albania, Romania and the USSR. Petroleum from these deposits can significantly increase crude oil supplies. However, no single oil-recovery process is likely to be applicable to all tar-sand deposits, which differ considerably in their geophysical and chemical properties. The authors have estimated the risk of occurrence of significant unfavourable environmental, health and safety impacts associated with tar-sand technologies. These estimates were made from information related to typical emerging surface (above ground) and in-situ (underground) tar-sand oil-recovery processes. Both types of processes are being developed for use on tar-sand deposits in the USA and may also be applicable to deposits in other countries. First, the levels of pollutant emissions affecting land, air and water were determined from data related to current US field experiments involving surface processes (including retort and solvent extraction methods), and in-situ techniques (including combustion and steam-injection methods). Next, these data were extrapolated to determine pollutant levels expected from conceptual commercial facilities producing 20 000 barrels per day. These estimates predict the nature and magnitude of environmental, health and safety impacts. The likelihood of occurrence of these impacts was then assessed. Experience from other industries, including information concerning health and ecosystem damage from air pollutants, measurements of groundwater transport of organic pollutants, and the effectiveness of environmental control technologies, was used to make this assessment, from which it was concluded that certain adverse effects are more likely to occur than others. These effects are discussed in the paper and ordered for surface and in-situ technologies according to their likelihood of occurrence.

* Work performed under the auspices of the US Dept. of Energy by the Lawrence Livermore National Laboratory under Contract W-7405-eng-48. Reference to a company or product name does not imply approval or recommendation of the product by the University of California or the US Dept. of Energy to the exclusion of others that may be suitable.

TABLE I. MAJOR WORLD-WIDE TAR-SAND DEPOSITS

Country	Estimated barrels of oil equivalent	Reference
Canada	1.3×10^{12}	[2,5]
Venezuela[a]	7.0×10^{11}	[1,2,6]
USSR	1.4×10^{11}	[7]
USA[b]	3.0×10^{10}	[8]
Malagasy	2.0×10^{9}	[7]

[a] Heavy-oil reservoirs may be included in this estimate.
[b] 1981 revised estimate according to US Department of Energy Office of Fossil Energy is 3.6×10^{10} bbls of oil equivalent.

1. TAR SAND: DEFINITION, PROPERTIES AND LOCATION

Tar sands are deposits of consolidated or unconsolidated clastic sediments (e.g. sandstone, limestone, diatomite) that have pore spaces partially or completely saturated with a heavy, viscous petroleum known as bitumen. Like heavy-oil, tar-sand bitumen is a member of the petroleum family of organic substances that cannot be economically recovered by the relatively simple techniques used for extracting typical light-crude oil.

Specific characteristics frequently used to classify and identify tar-sand bitumen are (1) an API Gravity (a scale devised by the American Petroleum Institute to measure the density of petroleum) generally $\leq 12°$ [1-4], and (2) a viscosity of 10^5 to 10^6 centipoise (cP) [1-4]. In addition, tar-sand bitumen is generally soluble in strong solvent members of the petroleum family (e.g. benzene and toluene), and the viscosity decreases with increasing temperature.

Substantial tar-sand resources are located in Canada, Venezuela, the Soviet Union, Malagasy and the United States as illustrated in Table I. Other major tar-sand deposits may be found in Albania, Trinidad, and Rumania [7].

In the United States, Utah is believed to have over 80% of the domestic tar-sand resource. Other states containing tar-sand deposits with over 1 million barrels of oil are California, Kentucky, New Mexico and Texas [8]. Smaller deposits are located in Alabama, Kansas, Missouri and Ohio [1]. The geophysical and chemical properties of these tar sands differ considerably, and no single extraction process is likely to be applicable to all deposits [1,2,9].

TABLE II. PROPOSED COMMERCIAL DEVELOPMENT OF TAR-SAND RESOURCES IN THE USA BY 1990 [10]

Company	Proposed extraction procedure	Estimated production capacity (bbl/d)	Location
Getty Oil Company	Surface	20 000	McKittrick, CA
Standard Oil of Ohio	Surface	25 000	Vernal, UT
Conoco Oil Company	*In situ*	25 000	Uvalde, TX

2. EMERGING TAR-SAND TECHNOLOGIES IN THE UNITED STATES

Surface (above ground) and *in situ* (underground) tar-sand technologies are now being evaluated in the United States for commercial application. The surface extraction processes being field-tested involve surface-mining the tar-sand ore and then extracting the oil from the ore by above-ground retorting or solvent systems. The *in situ* systems being developed are combustion and steam-injection procedures. Extraction processes may need to be followed by upgrading procedures to make the crude petroleum an acceptable refinery feedstock.

2.1. Commercialization

Although no commercial tar-sand oil-extraction facilities exist in the United States at this time, both *in situ* and surface tar-sand technologies are perceived to be candidates for commercialization. Table II indicates plans for commercial development that have already been announced by Getty Oil Company, Standard Oil Company of Ohio, and Conoco Oil Company. Smaller-scale commercial development is being considered by other companies. A reasonable estimate of tar-sand oil production in the US by 1990 would be 80 000 to 120 000 barrels of oil per day (bbl/d).

3. RISK ESTIMATES

We extrapolated data from current US field experiments to estimate the levels of pollutant emissions from conceptual 20 000 bbl/d surface retort and solvent operations and *in situ* combustion and steam-injection systems. These estimates were

TABLE III. LIKELIHOOD OF OCCURRENCE OF EFFECTS FROM THE OPERATION OF COMMERCIAL TAR-SAND OIL-RECOVERY PROCEDURES

Possible effects	Effect according to process: Surface extraction	*In situ* recovery
Airborne effluents:		
Public health	2	2
Ecosystems	2	2
Surface-water contamination:		
Public health	1	1
Ecosystems	1	1
Ground-water contamination:		
Public health	2	2
Land subsidence	NA	2
Induced seismicity	NA	1
Land-surface contamination:		
Public health	2	1
Ecosystems	2	1
Worker health	3	2

Key: 3 - Effect likely to occur based upon experience in other industries.

2 - Effect may occur but will be localized and/or controllable.

1 - Effect possible but unlikely to occur.

NA - Not applicable.

then used to predict the nature and magnitude of the potential environmental, health and safety impacts. The likelihood-of-occurrence of the impacts was then assessed based on experience from other energy industries involving health and ecosystem damage from air pollutants, measurements of ground-water transport of organic pollutants, and the effectiveness of environmental control technologies. The possible effects and the subjectively determined values indicating their likelihood-of-occurrence are presented in Table III.

Uncertainties associated with the subjectively determined numerical values introduced in Table III are related to (1) the early stage of development of tar-sand technologies in the US; (2) the efficacy of changes in process design vs "retrofits" for achieving environmental, public and occupational safety; (3) the preparedness of commercial tar-sand facilities to mitigate effectively hazards unique to the technology; and (4) the influence of regulatory and economic factors.

3.1. Atmospheric effects

Controlled emissions of nitrogen dioxide (NO_2), sulfur dioxide (SO_2), carbon monoxide (CO), hydrocarbons (HC), and total suspended particulates (TSP) from 20 000-bbl/d retort or solvent processes are not expected to exceed US National Ambient Air Quality Standards (NAAQS) designed to protect public health. However, SO_2 and TSP emissions may cause US 'Prevention of Significant Deterioration' increments to be exceeded in those regions already meeting NAAQS.

Estimates of uncontrolled emissions of several pollutants from conceptual 20 000-bbl/d in situ combustion and steam-injection processes revealed that a few are likely to exceed US legislated and recommended air-pollutant standards. Therefore in situ systems are certain to require air-pollution-control devices and strategies. Our calculations also indicate that, at least for in situ combustion systems, current air-pollution-control technologies may not be effective in reducing H_2S emissions to acceptable levels.

An estimate of the magnitude of the public-health impacts attributable to air pollution from commercial tar-sand facilities was obtained using a health-damage function derived for SO_2 emissions from coal-fired power plants [11]. The model from which the health-damage function for SO_2 emissions was derived assumes Gaussian plume dispersion, first-order kinetics of chemical reactions, a linear dose-response function, and a population of about 3×10^6 people within 80 km of an eastern power plant [12]. The calculated damage function is 0 to 120 premature deaths per year of operation (80% confidence interval), or 0 to 0.8 premature deaths per 1000 ton (short) of SO_2 emitted [11]. We have used the latter relationship to calculate premature deaths from the release of SO_2 from tar-sand production technologies by using estimated populations of 300 000 people within 80 km of McKittrick, California, and 30 000 people near the tar-sand deposits in Utah. These two situations are representative of projected sites for tar-sand facilities that will use surface and in situ processes, respectively. The results are shown in Table IV.

TABLE IV. ESTIMATED PREMATURE DEATHS WITHIN 80 km OF TAR-SAND TECHNOLOGIES FROM ATMOSPHERIC EMISSIONS OF SO_2

Process	Estimated number of premature deaths / plant-year
Surface retort system[a]	0 to 0.08[b]
Surface solvent system[a]	0 to 0.06[c]
In situ steam-injection procedure[d]	0 to 0.003[b]
In situ combustion procedure	Insufficient data

[a] Location at McKittrick, California.
[b] Assumes 97% control of SO_2 emissions.
[c] Assumes no control of SO_2 emissions other than the use of low-sulfur fuel.
[d] Location near Vernal, Utah.

Atmospheric pollutants from tar-sand facilities can also cause adverse ecological impacts. A simple linear relationship has been developed to describe the growth response of vegetation after exposure to concentrations of SO_2 above the tolerance threshold [13]. This relationship was employed to estimate the magnitude of the effect upon vegetation by SO_2 emissions released from tar-sand projects. The result for surface technologies was a yield reduction of approximately 20%. The concentration of SO_2 resulting from release from a commercial *in situ* steam-injection process was estimated to be lower than the threshold necessary to impact adversely the growth response of vegetation. We were unable to estimate the effect on vegetation of SO_2 released from *in situ* combustion systems because of insufficient data on SO_2 emission rates.

The estimates of impact on yield are preliminary and indicate the potential growth-response of vegetation in the vicinity of the tar-sand operation. We expect that any adverse impacts that do occur will be localized and the application of control technologies will reduce these impacts to minimum levels.

3.2. Surface-water contamination

Adverse impacts on public health and ecosystems from direct contamination of surface water are probably unlikely to occur from the operation of commercial surface and *in situ* systems. This is because the fluids will generally be physically contained in either vessels and pipelines or the formation itself. It is expected that practices already commonplace in the chemical industry will effectively reduce the likelihood of major spills and leaks, and prevent any that do occur from reaching surface waters. These practices include monitoring, maintenance, and spill-prevention programs. However, the risk of surface-water contamination will increase substantially if planned recycling, injection, and underground disposal procedures are not feasible.

3.3. Ground-water contamination

Surface processes can cause ground-water contamination from the leaching of substances contained in tar-sand storage piles, spent-ore storage piles, and in the operating and backfilled mine areas. Product waters that were analyzed from in situ combustion and steam-injection field tests contained significant concentrations of metals, nutrients and organic compounds [14]. This indicates that adverse ground-water-quality impacts could develop, and emphasizes the need for developers to monitor and model the transport of such pollutants at specific sites. For instance, monitoring studies have demonstrated that following undergound coal-gasification, vapor-transported organic materials were conducted to ground water 250 ft (76 m) away from the combustion zone [15]. However, the slow movement of ground water, combined with attenuation of pollutants by soil, means that effects may occur but will be localized and/or controllable.

3.4. Land subsidence and induced seismicity

Land subsidence and induced seismicity are two potential effects associated exclusively with in situ operations.

The impact of any subsidence produced by commercial in situ tar-sand operations will depend on the land use surrounding the site. For the most part, subsidence represents a localized effect that can be controlled effectively by proper site-selection and operating practices.

Seismicity can result from the high-pressure injection of fluids and resulting lubrication of fault planes in a tar-sand formation. It is also conceivable that thermal stresses in the formation may cause seismic events. Unless unusual conditions exist in the formation, the potential environmental effects from induced seismicity should be insignificant.

3.5. Land-surface contamination

Land-surface contamination is more likely to be a major problem from surface systems due to the amount of area needed for mining and operating a retort or solvent process. The release of hazardous substances onto the ground from surface extraction processes should, however, be minimized by mitigating measures such as spill-prevention programs, monitoring procedures, and maintenance. Careful reclamation and restoration of the mine area will reduce the impacts associated with mining activities.

The possibility of land contamination from in situ processes is generally related to leaks or accidental spills from storage vessels and pipes. However drilling and pumping activities may also contribute to the problem. The impacts from these sources can be controlled using current oil-field procedures and, while possible, are unlikely to occur.

3.6. Worker health

The risk of morbidity and injury to workers operating commercial surface or in situ tar-sand processes could be significant. Information from other industries such as oil refineries, foreign shale-oil production, coking, and coal conversion indicates that exposure of workers to chemical compounds similar to those expected from tar-sand oil extraction can lead to toxic, mutagenic or carcinogenic effects.

A preliminary estimate of the cancer risk for workers at a tar-sand facility was computed using carcinogenic risk estimates for coke and oil-refinery workers [16]. The maximum risk of excess cancer-deaths per plant-year for tar-sand technologies based on this calculation is approximately 0.03. This is probably a very conservative figure as the exposure in a modern tar-sand facility will probably be much lower than past exposure in the reference industries because of the availability and increasing use of protective devices (e.g. respirators and protective clothing).

According to Dr. G. Neeson of Syncrude, Inc., Alberta, Canada, their studies to evaluate the carcinogenicity of products have led to the institution of precautionary protective measures to protect the health and safety of the workers from possible carcinogenic substances. Other problems detected at the tar-sand facility in Canada involved exposure of workers to an H_2S concentration of 1 ppm and to the odor of mercaptan compounds. Efforts are also underway to reduce potential occupational health impacts related to these emissions. The nominal similarities of several properties of bitumen from Canada and Utah [17] suggest that problems similar to those detected by Syncrude could develop at a surface tar-sand facility in the US.

Additional occupational health risks may result from repeated long-term exposures to subacute concentrations of toxic chemicals. For instance, hydrogen sulfide (H_2S), identified as an emission from in situ combustion field tests, was implicated as the chemical responsible for chronic poisoning of workers in the Swedish shale-oil industry during the 1940s [18]. This chronic poisoning produced symptoms such as mental depression, weakness, sweating and frequent chills. Futhermore, some residents of communities near units of The Geysers Geothermal Power Plant in California have appeared at hearings of the California Public Utilities Commission and complained of headache, nausea and sinus congestion that were attributed to the presence of H_2S (in concentrations probably as low as 0.1 ppmv) [19].

Worker health problems are more likely to occur at US surface extraction facilities than at in situ projects. This difference arises because surface extraction procedures not only will produce products that may represent occupational hazards, but also will require the use of reagents that may jeopardize workers' health and safety. Environmental control measures routinely used for heavy-oil extraction would most likely be effective for in situ tar-sand procedures because the methodologies are similar.

4. RECOMMENDATIONS AND CONCLUSIONS

Tar-sand petroleum-extraction technologies are in an early stage of development in the United States. Until more data are collected and carefully analyzed, we recommend that development of tar-sand technologies proceed toward commercialization with caution. An institutional commitment for strong safety programs is also needed. While no facility or project can be made completely free from hazard, we believe that a tar-sand industry can be made a reasonably safe place to work, and environmentally acceptable.

From our assessment of risk estimates for tar-sand technologies (see Table III) we draw the following conclusions:

- Surface Extraction Technologies
  - Occupational health problems, including those related to worker exposure to carcinogenic and other hazardous substances, are likely to occur.
  - Public health and/or ecosystem effects from airborne effluents, ground-water contamination and land-surface contamination may occur but will be localized and/or controllable.
  - Public health and/or ecosystem effects from surface-water contamination are possible, but unlikely to occur.
  - Land subsidence and induced seismic effects are not applicable to surface extraction techniques.

- In Situ Processes
  - Occupational health problems, including those related to worker exposure to carcinogenic and other hazardous substances, may occur but will be localized and/or controllable.
  - Public health effects from airborne effluents, (particularly SO_2 and H_2S) and ground-water contamination may occur but will be localized and/or controllable.
  - Ecosystem effects from airborne effluents and land subsidence may occur but will be localized and/or controllable.
  - Public health and/or ecosystem effects from surface-water contamination and land-surface contamination are possible, but unlikely to occur.
  - Induced seismic effects are possible, but unlikely to occur.

Early consideration of these potential problems is essential to the environmentally acceptable development of tar-sand technologies.

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DISCUSSION

J. DELFINER: Worker health effect was rated as being the most significant one because of the probable use of additional reagents in the future. What are those additional reagents?

J.I. DANIELS: For surface solvent systems and hot water processes it is not absolutely certain what the solvents will be, but likely candidates are benzene, toluene and mixed petroleum compounds. Retort processes will produce organic compounds and reduced sulphur compounds that may be toxic to humans. In addition, the technology is unproven for US bitumen at present, and precise knowledge concerning reagents is not available.

ANALYSE CRITIQUE DES ETUDES COMPARATIVES SUR LES RISQUES SANITAIRES ASSOCIES A LA PRODUCTION D'ELECTRICITE

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Abstract–Résumé

REVIEW OF COMPARATIVE STUDIES OF HEALTH HAZARDS ASSOCIATED WITH ELECTRIC POWER GENERATION.

Twenty comparative studies on health hazards associated with different methods of energy production were reviewed in order to highlight the methodological problems which are characteristic of this type of study and attempt to bring the results obtained closer together by using a more consistent methodological approach. The main conceptual difficulties lie in the delineation of the boundaries of a fuel cycle, the choice of technologies, dose-effect relationships and health hazard estimates, and in finding equivalent values in the case of real and hypothetical risks and of present and future risks. Practical limitations in predicting the number and gravity of illnesses and accidents both in workers and the population were also reviewed. The health effects associated with the coal, oil, natural gas and uranium cycles were estimated with the following assumptions: exclusion of risks associated with non-fuel resources, use of advanced technologies, use of linear non-threshold dose-effect relationships, exclusion of hypothetical accidents and exclusion or discounting to present worth of long-term risks (>500 years). It can be concluded from these calculations that the health hazards from all the energy sources considered are relatively low and therefore acceptable, natural gas being the safest source followed by uranium, oil and coal in that order. The uncertainty factors are discussed and methods to improve the results are proposed. It is also shown that the radiological risks to workers in the PHWR cycle are approximately half those of the LWR cycle.

ANALYSE CRITIQUE DES ETUDES COMPARATIVES SUR LES RISQUES SANITAIRES ASSOCIES A LA PRODUCTION D'ELECTRICITE.

Vingt études comparatives sur les risques sanitaires associés aux différents modes de production d'énergie ont été analysées dans le but de mettre en évidence les problèmes méthodologiques caractéristiques de ce genre d'études et pour essayer de diminuer les divergences entre les résultats trouvés, en utilisant une approche systématique plus cohérente. Les principales difficultés conceptuelles se rapportent à la détermination des frontières d'une chaîne énergétique, au choix des technologies, des relations doses-effets et des indices sanitaires, et à l'équivalence entre risques réels et hypothétiques et entre risques présents et futurs. Les limitations pratiques dans la prévision du nombre et de la gravité des maladies et des accidents, autant chez les travailleurs que pour la population, ont également été passées en revue. Des estimations sur les effets sanitaires associés aux chaînes du charbon, du fuel, du gaz naturel et de l'uranium ont été effectuées, basées sur les hypothèses suivantes: exclusion des risques associés aux ressources non énergétiques, utilisation de technologies nouvelles, adoption de relations doses-effets linéaires sans seuil, exclusion des accidents hypothétiques et exclusion ou actualisation des risques à long terme (>500 ans). D'après ces estimations, on peut conclure que les risques sanitaires de toutes les sources d'énergie considérées sont relativement faibles et par conséquent acceptables, et que l'ordre croissant de sécurité va du charbon au pétrole, à l'uranium et au gaz naturel. Les incertitudes qui subsistent sont discutées et l'on propose des moyens pour affiner les résultats obtenus. On montre également que les risques radiologiques chez les travailleurs dans la chaîne PHWR sont environ de moitié inférieurs à ceux de la chaîne LWR.

1. INTRODUCTION

Il semble évident que les sociétés les plus avancées industriellement s'inquiètent de plus en plus des conséquences potentiellement nuisibles découlant des technologies qu'elles ont développées. Etudes, débats et réglementations s'étendent maintenant à des secteurs aussi divers que l'alimentation, le transport, l'habitation, l'industrie chimique et la production d'énergie. Si l'on s'accorde à reconnaître que les principaux bienfaits de la technologie moderne sont la libération de l'homme de ses soucis de base, l'augmentation de sa longévité et l'augmentation de son standard de vie, on craint cependant, à divers degrés, la dégradation de l'environnement, des menaces sournoises à la santé et, en général, la diminution possible de la qualité de vie qui sera léguée à nos descendants.

C'est dans ce très vaste cadre que s'inscrivent les études sur les risques sanitaires associés aux différents modes de production d'énergie. Ces études visent deux objectifs fondamentaux: l'identification des activités nécessitant des mesures correctives et la classification des sources énergétiques selon leur degré de sûreté. L'étude comparative des risques est cependant délicate car de nombreux problèmes méthodologiques doivent d'abord être surmontés. Deux analyses critiques indépendantes [1,2] ont récemment mis en évidence des écarts considérables entre les valeurs trouvées dans la littérature et ont insisté sur la nécessité d'une approche systémique plus cohérente.

Dans le cadre de nos travaux, vingt études comparatives [3-22] ont été analysées. Les sources énergétiques considérées ont été le charbon, le fuel, le gaz naturel et l'uranium. Les chaînes énergétiques ont été examinées dans leur totalité, depuis l'extraction du combustible jusqu'au traitement des déchets. Les effets sanitaires ont été groupés en neuf catégories: maladies professionnelles (mortelles et non mortelles), accidents de travail (blessures mortelles et non mortelles), quatre catégories similaires pour la population en général et les effets génétiques. Ces indicateurs de risques ont été ensuite normalisés par rapport à la production d'une énergie électrique égale à 1 Gigawatt·année.

La compilation détaillée des résultats numériques recensés a été résumée ailleurs [23]. L'essentiel de cette communication portera sur les difficultés conceptuelles et sur la nécessité d'une approche cohérente. On montrera, en particulier, que si l'on utilise des hypothèses de travail raisonnables, les écarts entre les valeurs extrêmes trouvées dans la littérature diminuent considérablement. On indiquera également quelques domaines controversés qui nécessitent plus d'études et de réflexion. Finalement, une comparaison entre les chaînes LWR et PHWR sera amorcée.

2. DIFFICULTÉS CONCEPTUELLES

Certains auteurs, dont Holdren et al. [24], ont déjà discuté des difficultés rencontrées dans l'analyse comparative des risques. A partir de ces études et de notre propre analyse, nous résumons ici ces difficultés et indiquons les hypothèses de travail que nous avons retenues pour faire nos propres estimations.

2.1 Le problème des frontières

La détermination des frontières d'une chaîne énergétique présente un double choix: l'inclusion ou l'exclusion des risques associés à la production et à l'utilisation de matériaux non spécifiquement associés à la chaîne, et l'inclusion ou l'exclusion des phases construction et démantèlement des installations appartenant à la chaîne.

La comptabilisation de tous les risques permet, en principe, de déterminer le risque total mais, en pratique, elle ne peut se faire qu'à l'aide de modèles économiques du type entrées-sorties assez grossiers; la comptabilisation des risques spécifiques à la chaîne ne donne que des risques partiels mais cette approche est beaucoup plus simple et plus précise. Comme il semble que pour les sources énergétiques non-renouvelables, les risques "non-énergétiques" sont effectivement négligeables par rapport aux risques "énergétiques" - ou tout au moins peu différents entre eux d'une chaîne à l'autre - nous favorisons l'approche "risques spécifiques".

2.2 Le choix entre diverses technologies

Les progrès technologiques et la sévérité accrue des réglementations ont contribué à rendre les nouvelles installations plus sécuritaires pour le public et les conditions de travail moins dangereuses pour les travailleurs. Le problème réside dans le choix entre un système de production d'électricité basé sur les installations actuelles et un système de référence basé sur la technologie la plus avancée. Dans le premier cas, on calcule un risque moyen, reflet de la situation actuelle; dans le deuxième cas, on détermine le risque marginal associé à un développement énergétique ultérieur. Nous favorisons le deuxième choix.

2.3 Choix entre différentes relations doses-effets

Qu'il s'agisse des effets des radiations ou des effets des substances chimiques, les effets à faible dose d'exposition et, en particulier, l'existence ou la non-existence d'un seuil de tolérance, restent hypothétiques. Tant que le problème ne sera pas résolu, nous préférons utiliser la même approche dans les deux cas, soit l'utilisation d'une relation linéaire doses-effets, sans seuil.

2.4 Equivalence entre risques réels et risques hypothétiques

La difficulté consiste à décider si la définition habituelle du risque - produit de la fréquence d'un événement par les conséquences de celui-ci - s'étend jusqu'aux limites extrêmement faibles ($\sim 10^{-7}$ par année) que l'on rencontre dans l'analyse des accidents hypothétiques pouvant entraîner des conséquences catastrophiques.

Selon que l'on dispose ou non de données statistiques directes pour identifier ces événements rares et évaluer leurs risques, il convient de classer les accidents en réels ou hypothétiques. Il est suggéré ici de considérer les risques des accidents hypothétiques dans une catégorie séparée, non nécessairement additive avec les autres catégories de risques. Trois raisons militent en faveur d'une telle approche: l'incertitude associée à la grandeur de ces risques, le manque de cohérence à ce sujet dans les études comparatives et la perception plus aiguë de ces risques par le public. On pourrait ajouter cependant que la plupart des auteurs estiment que les risques associés aux accidents hypothétiques sont plus faibles que ceux associés aux accidents réels qui, eux, sont plus faibles que ceux qui résultent du fonctionnement normal.

2.5 Equivalence entre risques actuels et risques futurs

Dans ce cas, il s'agit de décider si les risques légués aux générations futures sont de même nature que ceux qui affectent notre génération et si ces risques sont additifs. Cette difficulté provient du fait que les chaînes énergétiques produisent ou libèrent des substances toxiques et radioactives qui, à cause de leur très longue durée de vie, peuvent affecter des milliers de générations.

Même si l'on admet volontiers que du point de vue éthique, la valeur d'une vie ne varie pas d'une génération à l'autre, nous nous rangeons du côté de ceux qui estiment que pour les risques associés à des produits très faiblement concentrés dans la biosphère, l'horizon temporel de nos responsabilités envers les générations futures ne devrait pas dépasser quelques dizaines de générations, parce que ces responsabilités ne sont pas illimitées, parce que les risques individuels en question sont extrêmement faibles, et parce que l'addition d'effets individuels infinitésimaux sur des milliards d'individus et sur des millions d'années présente un caractère extrêmement artificiel. En pratique, cette approche consiste à ignorer ou à actualiser, à un taux de quelques pour-cent, les risques dépassant l'horizon temporel de l'ordre de 500 ans.

2.6 Choix des indices sanitaires

Comme indices sanitaires, nous avons retenu les sept catégories indiquées dans l'Introduction. Les maladies et les blessures peuvent à leur tour être classées selon leur gravité, en sa basant par exemple sur le nombre de journées de travail (ou de vie normale) perdues. Certains auteurs utilisent ce concept pour additionner tous les effets somatiques, y compris la mort; dans ce dernier cas, on évalue la différence de temps entre l'espérance de vie de la personne décédée et l'âge de cette personne au moment de sa mort. Malgré l'attrait d'une méthode qui permettrait d'exprimer l'ensemble des risques sanitaires par un seul indice [25], nous ne l'avons pas adoptée à cause du délicat problème d'équivalence entre effets différents. Par contre, l'addition d'une nouvelle catégorie d'effets sanitaires - l'anxiété mentale - pourrait s'avérer utile dans l'avenir.

3. LIMITATIONS DANS LA PRÉCISION DES ANALYSES DE RISQUES

Les risques sanitaires provenant des maladies sont difficiles à estimer avec précision; dans certains cas les incertitudes sont tellement élevées que certains auteurs hésitent même à les quantifier. Pour les travailleurs, il s'agit de prévoir les conséquences des améliorations des conditions de travail. Pour la population en général, il s'agit essentiellement du manque de précision dans les modèles de dispersion des produits nocifs dans la biosphère et de la méconnaissance de la géno-toxicité des éléments chimiques libérés le long des chaînes énergétiques.

Les risques sanitaires associés aux accidents sont mieux connus et les incertitudes sont moins grandes. Les limitations qui subsistent sont essentiellement dues aux causes suivantes:

(a) le manque de données statistiques spécifiques précises, par exemple sur les accidents de transport du charbon;

(b) la grande dispersion dans les données statistiques, pour une activité donnée, à l'intérieur d'un pays ou d'un pays à l'autre;

(c) la difficulté d'extrapoler à partir des tendances historiques;

(d) la non-uniformité dans la comptabilisation des blessures légères.

4. RISQUES PROVENANT DES MALADIES

4.1 Chaîne du charbon

Pour la chaîne du charbon, les risques proviennent, pour les travailleurs, de l'étape extraction, et pour la population, de l'étape production d'électricité. Dans le premier cas, il s'agit de la pneumoconiose et de la silicose des mineurs; dans le deuxième cas, des affections pulmonaires provenant essentiellement des produits sulfatés.

(a) Mortalités chez les travailleurs

Les estimations recensées varient entre ~0 et 8.7 [10] mortalités par Gigawatt·année. Il nous a semblé que l'étude de la plus exhaustive était celle de Morris et al. [18]; en se basant sur leurs estimations, nous avons pris comme valeur médiane 0.1 avec un facteur de dispersion de 5, soit, en notation condensée, 0.1 (5).

(b) Morbidités chez les travailleurs

Les estimations recensées varient entre 0.55 [6] et 64 [14]; nous avons choisi une valeur relativement optimiste: 1.5 (2).

(c) Mortalités dans la population

Comme on peut l'observer dans le tableau I, les valeurs varient de 0 à 405; les quelques points d'interrogations indiquent le refus des auteurs à faire des prédictions. En se fiant de nouveau sur l'analyse de Morris et al. [18] l'intervalle est réduit à 0-13 pour les effets des sulfates; à cet intervalle, il faut ajouter les effets des substances aromatiques polycycliques, estimés de 0 à 5.3. Les meilleures estimations semblent se situer autour de 3 si l'on utilise des modèles linéaires et autour de 0.2 pour des modèles à seuil. Notre estimation est de 3 (10).

(d) Morbidités dans la population

Les cas de morbidités se réfèrent aux maladies respiratoires chroniques chez les adultes, aux maladies respiratoires au niveau inférieur chez les enfants, à l'aggravation des

TABLEAU I. NOMBRE ANNUEL DE DÉCÈS, DANS LA POPULATION, DUS AUX MALADIES ASSOCIÉES À LA PRODUCTION DE 1 GW·a D'ÉNERGIE ÉLECTRIQUE. CHAÎNE DU CHARBON

Etape / Auteur	Conversion	Traitement des déchets[a]	Total
WASH-1224 [3]	?	–	?
Hamilton [4]	4-133	?-13.3	4(?)-146.3
Smith et al. [6]	0.2-36	0-13.0	0.2-49
Comar et Sagan [7]	0.076-133	1.3-13.3	1.38-146.3
Morris [10]	0-320	–	0-320
Inhaber [12]	0.1-140	1.4-14	1.5-154
I.E.A. [13]	1.23-3.67	–	1.23-3.67
A.M.A. [14]	0.076-392	1.3-13.3	1.5-405.3
U.K. [15]	?		?
Ramsay [16]	0-32	–	0-32
Schurr et al. [17]	0-10.7	–	0-10.7
Morris et al. [18]	0-18.6	–	0-18.6
Inhaber [19]	32-95	1.4-14	33.4-109
Belhoste et al. [20]	0.34	–	0.34
CONAES [21]	?	–	?
Hamilton [22]	9	–	9

[a] Déchets provenant de l'étape préparation.

symptômes cardio-pulmonaires chez les gens âgés et au nombre d'attaques d'asthme. Les estimations des différents auteurs varient de 59 [20] à 570,000 [19]. Notre estimation est de 10^3 (20).

4.2 Chaîne du fuel

Les risques sanitaires associés à la chaîne du fuel sont pratiquement inexistants pour les travailleurs et similaires à ceux de la chaîne du charbon pour la population.

4.3 Chaîne du gaz naturel

On a estimé que les risques du gaz naturel sont négligeables pour toutes les catégories d'indices sanitaires considérées.

4.4 Chaîne de l'uranium

Les effets sanitaires associés à la chaîne de l'uranium sont essentiellement de nature radiologique. Les risques calculés ont une bonne précision; les écarts entre les valeurs trouvées proviennent en partie de la pente choisie pour représenter la relation doses-effets, en partie de la technologie choisie pour réduire les doses d'exposition et en partie de la méthode utilisée pour calculer les engagements de doses.

(a) Mortalités chez les travailleurs

La plage des valeurs trouvées se situe entre 0.02 [14] et 0.81 [19]. Si l'on exclue les valeurs extrêmes, qui ne correspondent plus à nos connaissances d'aujourd'hui, on arrive rapidement à la conclusion que le risque radiologique se situe autour de 0.20 mortalités par Gigawatt·année, avec un facteur de dispersion autour de 1.50, soit 0.20 (1.5).

(b) Morbidités chez les travailleurs

Aux maladies d'origine radiologique, il faut ajouter les cas de silicose chez les mineurs. Notre meilleure estimation est de 0.60 (1.5).

(c) Mortalités dans la population en général

Les estimés recensés varient entre un total de 0.001 [6] et de 0.53 [19]. Les estimés les plus élevées comprennent des valeurs pessimistes sur la contribution des déchets radioactifs et une contribution non-négligeable du radon dégagé des terrils d'uranium. Une analyse détaillée des valeurs trouvées montre que les risques de mortalité se situent dans l'intervalle 10^{-3} à 10^{-2} par Gigawatt·année si l'on néglige la contribution de l'étape de retraitement du combustible et l'engagement de dose dû au radon. Le retraitement ramène les valeurs dans l'inter-

TABLEAU II. COMPARAISON ENTRE LES EFFETS SANITAIRES ASSOCIÉS AUX DIFFÉRENTES CHAÎNES ÉNERGÉTIQUES. RÉSULTATS NORMALISÉS À LA PRODUCTION DE 1 GW·a D'ÉNERGIE ÉLECTRIQUE (ACCIDENTS EXCLUS). MEILLEURES ESTIMATIONS ET FACTEURS D'INCERTITUDES.

	Travailleurs		Population		Total	
Chaîne	Décès	Maladies	Décès	Maladies	Décès	Maladies
Charbon	0.1(5)	1.5(2)	3(10)	10^3(20)	3(10)	10^3(20)
Fuel	~ 0	~ 0	3(10)	10^3(20)	3(10)	10^3(20)
Gaz naturel	~ 0	~ 0	~ 0	~ 0	~ 0	~ 0
Uranium	0.2(1.5)	0.6(1.5)	0.2(2.0)	0.2(2.0)	0.4(1.5)	0.8(1.5)

valle 10^{-2} à 10^{-1}, tandis que la contribution du radon dépend fortement du taux d'actualisation utilisé. Notre choix est de 0.2 (2.0).

(d) Morbidités dans la population

Nous avons choisi la même valeur de 0.2 (2.0) ce qui est, dans l'ensemble, plutôt pessimiste.

(e) Effets génétiques

De nombreuses études ont été faites pour déterminer les effets génétiques graves. Certaines de ces études ne considèrent que la partie significative du point de vue génétique des doses de radiation reçues. Etant donné les incertitudes sur la relation doses-effets, nous avons considéré, en première approximation, que ces effets étaient numériquement égaux à ceux des cas antérieurs.

4.5 Sommaire

Le tableau II contient nos meilleures estimations sur le nombre de décès prématurés et sur le nombre de maladies associées au fonctionnement d'une centrale électrique produisant une énergie de 1 Gigawatt·année. Les facteurs de dispersion ont été ajoutés entre paranthèses; ces facteurs, estimés de façon subjective, englobent des variations dans les estimations dues au choix des technologies, à l'estimation de l'efficacité des mesures de protection et à l'incertitude des calculs sur les effets sanitaires.

5. RISQUES PROVENANT DES ACCIDENTS

5.1 Chaîne du charbon

Dans la chaîne du charbon, les dangers d'accidents sont concentrés dans l'étape de l'extraction, où les victimes sont des travailleurs, et dans l'étape du transport où les victimes sont à la fois des travailleurs et des membres de la population. Nos estimations sont basées sur les résultats des publications les plus récentes.

(a) Accidents mortels chez les travailleurs

Les valeurs recensées varient de 0.19 [13] à 7.5 [6] et reflètent les conditions très différentes de travail. Notre meilleure estimation est de 1.4 (1.5).

(b) Accidents non mortels chez les travailleurs

Les estimations trouvées varient de 15 [13] à 1406 [20]; l'écart est plus considérable que dans le cas antérieur à cause de la non-uniformité dans les méthodes de comptabilisation des accidents non-mortels. Notre estimation est de 60 (1.5).

(c) Accidents mortels dans la population

La plage des valeurs trouvées s'étend de ~0 [11] à 5.3 [11]. Nous avons choisi 1.0 (1.5).

(d) Accidents non mortels dans la population

Dans ce cas, l'intervalle trouvé est de 0.82 [18] à 48 [5]. Notre estimation est de 1.8 (2.0).

5.2 Chaîne du fuel

La chaîne du fuel est jugée moins dangereuse que la chaîne du charbon à cause de l'automatisation élevée de l'industrie, de la productivité plus élevée des travailleurs et des normes de protection plus sévères. Les seuls domaines d'inquiétudes restent, pour les travailleurs, les dangers associés à l'extraction et, pour la population, les incendies et les explosions dans les installations de raffinage. Les possibilités de déflagrations catastrophiques existent mais les risques associés à ces accidents hypothétiques restent dans le domaine de la conjecture et n'ont pas été rapportés ici.

(a) Accidents mortels chez les travailleurs

Les valeurs trouvées varient de 0.11 [5] à 2.0 [19]. Notre choix est de 0.35 (1.5).

(b) Accidents non mortels chez les travailleurs

Dans ce cas, les estimations varient de 5.2 [6] à 146 [19]. Notre estimation est de 30 (1.5).

(c) Risques à la population

Ces risques, difficiles à quantifier, sont jugés négligeables.

5.3 Chaîne du gaz naturel

Il existe relativement peu d'études indépendantes sur les risques associés à la chaîne du gaz naturel. Toutes ces études montrent que les risques actuels sont faibles. L'emploi croissant du gaz naturel liquéfié, ou du gaz synthétique provenant du charbon, peut modifier les estimations courantes. Les risques catastrophiques ne seront pas discutés ici

(a) Accidents mortels chez les travailleurs

Les valeurs trouvées dans la littérature varient de 0.08 [5] à 0.48 [19]. La valeur choisie est de 0.20 (1.5).

(b) Accidents non mortels chez les travailleurs

Dans ce cas, l'intervalle des estimations varie de 5.3 [14] à 41 [19]. Notre estimation est de 15 (2).

(c) Risques à la population

Comme avec la chaîne du pétrole, ces risques sont jugés négligeables. Un seul auteur [19] a fait des estimations à ce sujet: 0.009 pour les accidents mortels et 0.05 pour les accidents non mortels; nous avons adopté ces valeurs avec un facteur d'incertitude de 1.5.

5.4 Chaîne de l'uranium

A cause des dangers inhérents à l'exploitation de l'énergie nucléaire, la chaîne de l'uranium à fait l'objet, dès son introduction commerciale, d'une réglementation et d'un contrôle très sévères. C'est ainsi que la population a été très bien protégée - les seules victimes a déplorer proviennent des accidents de transport - et que les dangers, pour les travailleurs, restent essentiellement confinés à l'étape de l'extraction. Comme pour les autres chaînes, les risques hypothétiques ne seront pas considérés ici.

TABLEAU III. NOMBRE ANNUEL DE DÉCÈS, CHEZ LES TRAVAILLEURS, PROVENANT DES ACCIDENTS ASSOCIÉS À LA PRODUCTION DE 1 GW·a D'ÉNERGIE ÉLECTRIQUE. CHAÎNE DE L'URANIUM

Auteur \ Etape	Extraction	Préparation[a]	Transport	Conversion	Total
WASH-1224 [3]	0.12	0.007	0.003	0.013	0.13
Hamilton [4]	0.27	0.07	-	0.013	0.35
Hittman [5]	0.16[b]	←	←	0.015	0.175
Smith et al. [6]	0.12-0.27	0.007	0.003-0.012	0.013-0.017	0.14-0.31
Comar et Sagan [7]	0.07-0.27	0.004-0.27	0.003	0.013	0.09-0.55
Pochin [8]	0.3	0.12	0.003	0.27[c]	0.69[c]
Morris [10]	0.12-0.27	0.005	-	0.013	0.13-0.28
W.H.O. [11]	0.1	0.13	< 0.01	0.27[c]	0.55[c]
Inhaber [12]	0.08-0.57[b]	←	0.003-0.012	0.013-0.017	0.10-0.60
A.M.A. [14]	0.07-0.27	0.004-0.27	0.003-0.007	0.013	0.08-0.55
U.K. [15]	0.1	-	-	0.15[b]	0.25
Ramsay [16]					0.09-0.31[c]
Schurr et al. [17]					0.08-0.27[c]
Inhaber [19]	0.12-0.57[b]	←	0.003-0.012	0.013-0.017	0.25-0.75[c]
Belhoste et al. [20]	0.043	0.129	0.009	0.018	0.19
CONAES [21]	0.26	0.001	0.013	0.013	0.27
Hamilton [22]	0.477	0.006	0.015	0.021	0.52

a Comprend les étapes raffinage, enrichissement, fabrication et retraitement du combustible.

b Comprend également les étapes suivantes, indiquées par des flèches.

c Comprend les accidents ayant eu lieu durant la construction de la centrale.

(a) Accidents mortels chez les travailleurs

Le tableau III indique, de façon très détaillée les valeurs recensées. Nous avons retenu la valeur 0.20 (1.5).

(b) Accidents non mortels chez les travailleurs

Dans ce cas, les valeurs varient de 5.0 [7] à 45.7 [20]. Nous avons choisi 15 (2.0).

(c) Risques à la population

TABLEAU IV. COMPARAISON ENTRE LES EFFETS DES ACCIDENTS ASSOCIÉS AUX DIFFÉRENTES CHAÎNES ÉNERGÉTIQUES. RÉSULTATS NORMALISÉS À LA PRODUCTION DE 1 GW·a D'ÉNERGIE ÉLECTRIQUE. MEILLEURES ESTIMATIONS ET FACTEURS D'INCERTITUDES.

	Travailleurs		Population		Total	
Chaîne	Décès	Blessés	Décès	Blessés	Décès	Blessés
Charbon	1.40 (1.5)	60 (1.5)	1.0 (1.5)	1.8 (2.0)	2.40 (1.5)	62 (1.5)
Fuel	0.35 (1.5)	30 (1.5)	~ 0	~ 0	0.35 (1.5)	30 (1.5)
Gaz naturel	0.20 (1.5)	15 (2)	0.009 (1.5)	0.05 (1.5)	0.21 (1.5)	15 (2)
Uranium	0.20 (1.5)	15 (2)	0.012 (1.5)	0.11 (2.0)	0.21 (1.5)	15 (2)

Une seule étude indépendante [3] donne 0.012 accidents mortels et 0.11 accidents non mortels. Nous avons accepté ces valeurs avec des facteurs d'incertitude de 1.5 et 2.0, respectivement.

5.5 Sommaire

Dans le tableau IV nous avons rassemblé, pour chaque chaîne, nos meilleures estimations sur les risques provenant des accidents réels associés au fonctionnement d'une centrale électrique produisant 1 Gigawatt·année d'électricité.

6. COMPARAISON PARTIELLE DES FILIÈRES LWR ET PHWR

Plusieures études s'effectuent présentement au Canada pour estimer les risques sanitaires de la filière nucléaire CANDU qui utilise l'uranium naturel et l'eau lourde. Nous donnerons ici quelques résultats partiels qui permettent de comparer les risques radiologiques subits par les travailleurs dans les chaînes PHWR et LWR. Comme la chaîne PHWR présente moins d'étapes que la chaîne LWR, on peut s'attendre, a priori, que les risques de la chaîne canadienne soient plus faibles.

Les étapes extraction et préparation sont responsables, ensemble, d'une dose collective de 0.58 rem et 0.6 WLM, par tonne d'uranium, provenant de l'irradiation externe et de l'irradiation du radon et de ses sous-produits, respectivement [26]. Tenant compte qu'il faut extraire quelque 175 tonnes d'uranium pour produire 1 GW.a d'électricité, les doses collectives normalisées sont d'environ 100 rem et 100 WLM respectivement. Avec des facteurs de conversion en mortalités radiologiques de 10^{-4}/rem et de $3x10^{-4}$/WLM, respectivement, on trouve un total de 0.04 victimes par Gigawatt.année.

On obtient pratiquement la même valeur si l'on se réfère aux effets des doses collectives dans les centrales nucléaires: les valeurs moyennes des trois dernières années se situent autour de 500 rem pour la centrale de Pickering et de 200 rem pour celle de Bruce, par Gigawatt·année [27].

Les effets associés au transport et à la disposition des déchets sont jugés négligeables (<0.005); le démantèlement contribue à environ 0.004 victimes radiologiques par Gigawatt. année [28].

Le total des victimes radiologiques peut donc être estimé à 0.09 (1.2)/GW·a pour les travailleurs dans la chaîne PHWR soit environ la moitié de la valeur correspondante pour la chaîne LWR.

7. CONCLUSIONS GÉNÉRALES

Malgré les difficultés et les limitations inhérentes à l'analyse des risques associés aux chaînes énergétiques, on peut tirer quelques conclusions générales:

- avec des moyens adéquats de protection, toutes les sources énergétiques considérées sont socialement acceptables du point de vue sanitaire car les risques les plus graves associés à la production d'électricité, sont du même ordre que les risques correspondants associés à la consommation de cette électricité;
- les risques spécifiques associés à la chaîne de l'uranium ne sont pas plus grands et probablement beaucoup plus faibles que les risques correspondants associés à la chaîne du charbon;
- des recherches supplémentaires sont nécessaires pour diminuer la plage des incertitudes qui subsistent (en particulier sur les effets des produits de combustion), pour nous assurer que des effets importants n'ont pas été négligés et pour identifier et protéger les groupes d'individus potentiellement plus sensibles aux effets des produits dégagés le long des chaînes énergétiques;

- les risques des accidents hypothétiques à conséquences catastrophiques devront faire l'objet d'études séparées et systématiques;
- des estimations plus réalistes sur les risques sanitaires des chaînes énergétiques doivent se baser sur des scénarios de développement énergétiques précis qui tiennent compte de façon adéquate des réalités géographiques et socio-économiques locales;
- si l'on désire pousser plus loin l'analyse des systèmes énergétiques, il est indispensable d'avoir recours aux modèles économiques du type entrées-sorties afin de retracer toutes les activités industrielles reliées à la production d'énergie; c'est seulement par ce moyen qu'une comparaison rigoureuse entre énergies renouvelables et non renouvelables pourra être entamée;
- une attention particulière devra être portée au concept de la responsabilité vis-à-vis les générations futures et sur les limites de cette responsabilité.

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DISCUSSION

F. GIRARDI: May I congratulate you on this review and particularly on your attempt to normalize individual studies on the basis of common ground rules. Such studies are urgently needed.

I would like to know if the nuclear energy scenarios described refer to a one-through policy such as that favoured by the United States of America, or to a uranium recycling policy such as that in vogue in the European countries.

W. PASKIEVICI: Thank you. The differences in radiological hazards between the two cycles are not very great, as can be seen from the CONAES Report, and are certainly within the uncertainty factors indicated. The main reason for this is that the hazards associated with mining more uranium ore almost balance those avoided by not reprocessing fuel.

Session VIII

NEW RISK ASSESSMENT DESIGNS FOR COMPARISON OF THE HEALTH EFFECTS OF DIFFERENT ENERGY SOURCES

Chairman

C. RICHMOND

United States of America

ОСНОВНЫЕ ФАКТОРЫ РИСКА ДЛЯ ЗДОРОВЬЯ ПРИ ПОЛУЧЕНИИ ЭНЕРГИИ В ЯДЕРНОМ И УГОЛЬНОМ ТОПЛИВНЫХ ЦИКЛАХ

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Запрошенный доклад

Abstract–Аннотация

THE MAIN HEALTH HAZARDS FROM POWER PRODUCTION WITH THE NUCLEAR AND COAL FUEL CYCLES.

A comparative estimate is made of the health detriment for workers and the public from power production with the nuclear and coal fuel cycles. The following stages in the nuclear fuel cycle are considered: mining of uranium ore, fuel fabrication, power production, waste disposal, fuel reprocessing, reactor decommissioning and research projects. The stages in the coal fuel cycle examined are: coal-mining, transport and power production. In estimating the health detriment, account was taken of possible late effects of radiation (cancer) in workers and the public, accidents and occupational diseases affecting workers, and the incidence of cancer among the public as a result of the action of chemical carcinogens. It is shown that the total health detriment from the nuclear fuel cycle is considerably lower than that from the coal fuel cycle.

ОСНОВНЫЕ ФАКТОРЫ РИСКА ДЛЯ ЗДОРОВЬЯ ПРИ ПОЛУЧЕНИИ ЭНЕРГИИ В ЯДЕРНОМ И УГОЛЬНОМ ТОПЛИВНЫХ ЦИКЛАХ.

Приведена сравнительная оценка ущерба для здоровья персонала и населения при получении энергии в ядерном (ЯТЦ) и угольном (УТЦ) топливных циклах. В ЯТЦ учтены этапы: добыча урановой руды, изготовление топлива, получение энергии, захоронение отходов, переработка топлива, демонтаж реактора, исследовательские разработки. В УТЦ рассмотрены этапы: добыча угля, транспортировка, получение энергии. При оценке ущерба здоровью принимались во внимание возможные отдаленные последствия облучения (рак) у персонала и населения, несчастные случаи и профессиональные заболевания у персонала, возникновение рака у населения вследствие воздействия химических канцерогенов. Показано, что суммарный ущерб здоровью в ЯТЦ существенно ниже, чем в УТЦ.

1. ЗАДАЧИ И МЕТОДЫ СРАВНИТЕЛЬНОЙ ОЦЕНКИ ГИГИЕНИЧЕСКОЙ ЗНАЧИМОСТИ ЯДЕРНОЙ И ТРАДИЦИОННЫХ ВИДОВ ЭНЕРГЕТИКИ

Важность проблем, связанных с необходимостью прогнозирования воздействия энергетики на окружающую среду и здоровье человека, ныне является общепризнанной. Именно энергетика является в настоящее время и, по-видимому, останется в ближайшем будущем одним из главных источников загрязнения окружающей среды. Весьма значительна роль загрязнений и всего процесса производства энергии с точки зрения непосредственного влияния на здоровье человека. Достаточно сказать, что, по мнению ряда специалистов, до 90% существующего уровня заболеваемости злокачественными опухолями обусловлено факторами окружающей среды [1-3]. Значительная часть встречающихся в атмосфере загрязнителей, в том числе выбрасываемых энергетическими предприятиями, обладает мутагенным и гонадотропным действием, то есть способна отражаться на здоровье потомства. Таким образом, не будет преувеличением сказать, что пути развития энергетики самым непосредственным образом связаны с "судьбой" природы и здоровья людей.

В последние годы стал очевидным глобальный, планетарный характер воздействия энергетики на природу и здоровье. Это было показано для долгоживущих радионуклидов с так называемым "глобальным характером распространения". Однако и альтернативные типы энергетики и, прежде всего, энергетика, основанная на органическом топливе, как это стало ясно в последнее время, приобрела характер важного фактора глобального воздействия на окружающую среду.

Все сказанное выше объясняет возникший в последние годы интерес к оценке не только экономической стоимости и конкурентоспособности различных типов производства энергии, но также интерес к оценке сравнительной опасности разных видов энергетики для окружающей среды и здоровья человека. В общем виде задача сводится к тому, чтобы определить "цену" производства единицы энергии в ядерной и других видах энергетики в сопоставимых показателях ущерба для окружающей среды и здоровья человека с тем, чтобы облегчить принятие оптимальных решений по развитию и эксплуатации разных видов энергетики.

Решение задачи по эколого-гигиенической оценке разных видов энергетики сопряжено со многими трудностями и, по-видимому, потребует еще немало времени и усилий для проработки ее во всех деталях. Одна из основных трудностей заключается в том, что эффекты, вызываемые в организме человека за счет воздействия разных видов энергетики, зачастую неоднозначны по своей природе и потому трудно сопоставимы. Многие виды воздействий от ряда факторов вообще количественно не изучены (эффекты сернистого ангидрида, окислов азота, пыли; связь между многими токсическими, в частности, канцерогенными веществами и заболеваемостью населения).

По указанной проблеме в последние несколько лет появился целый ряд работ. Однако большинство из них дает сравнительные оценки лишь влиянию выбросов атомных электростанций (АЭС) и теплоэлектростанций (ТЭС) на традиционном топливе, да и то лишь по отдельным компонентам выброса, а не по всему "букету" [4-10]. Лишь в единичных работах последнего времени предпринимается попытка дать оценку всем выбросам АЭС и ТЭС, а также других предприятий топливного цикла [11-16]. К сожалению, эти оценки не полны из-за учета только части компонентов вредных воздействий и в ряде важных моментов не убедительны. В частности, они не позволяют получить количественных оценок вреда от химических компонентов выбросов.

Ниже, при сравнении воздействия ядерного (ЯТЦ) и угольного (УТЦ) топливно-энергетических циклов, будет более подробно охарактеризована имеющаяся информация и неопределенности, с которыми приходится сталкиваться при гигиенической оценке разных видов энергетики. На основании комплексного подхода дается возможно более полная и реалистическая оценка ущерба для населения и персонала от ЯТЦ и УТЦ.

Важным моментом, зачастую упускаемым из вида при определении "цены" производства энергии, является оценка ущерба для здоровья не только населения, но и персонала, обеспечивающего топливно-энергетический цикл. Число людей, занятых в энергетике, составляет существенную часть населения промышленно развитых стран. Так, в СССР численность рабочих и инженерно-технического персонала, занятого лишь добычей угля, составляет около 1 млн. человек [17, 18]. В то же время персонал, занятый на ряде этапов энергетических циклов, подвергается наиболее выраженным воздействиям. Естественно, что при определении в показателях здоровья "цены", которой общество вынуждено расплачиваться за единицу произведенной энергии, нельзя обойти эффекты, которые имеют место у персонала. Ниже будут рассмотрены главные из них – риск несчастных случаев и профессиональные заболевания на всех этапах циклов, где они встречаются и имеют значение.

Выбор для сравнения двух типов энергетики – ЯТЦ и УТЦ – обусловлен тем, что в ближайшие десятилетия они могут стать преобладающими способами производства энергии [19, 20]. Прогнозирование и планирование развития и рациональное использование каждого из них в значительной степени зависит от объективной количественной оценки влияния этих отраслей энергетики на здоровье человека.

Ниже проводится соответствующее сравнение на основе имеющейся к настоящему времени отечественной и зарубежной информации. Следует отметить, что во всех случаях, когда речь идет о ядерной энергетике и в оценке данных нет определенности, в расчет взяты наиболее осторожные, то есть наименее оптимистичные данные.

Вопросы оценки ущерба от вредных факторов, действию которых подвергается персонал и население в процессе получения энергии, разработаны недостаточно. Одна из первых и наиболее серьезных работ в этой области – публикация №27 МКРЗ [21]. Однако в этой работе рассматриваются лишь некоторые виды ущерба, встречающиеся в ЯТЦ. Ряд важных вредных факторов, в частности, пылевых и химических, в этой работе не оценивается. В нашем сообщении предпринята попытка дать оценку возможного ущерба от ЯТЦ и УТЦ с учетом последствий воздействия всех основных факторов риска.

Оценка основных видов возможного ущерба для здоровья проводилась по трем показателям.

1. Число случаев преждевременной смерти:

   а) от последствий облучения – за счет рака у населения и персонала;

   б) от профессиональных заболеваний, не связанных с облучением (за счет пневмокониозов и хронических пылевых бронхитов) – у горнорабочих и персонала обогатительных фабрик;

   в) от несчастных случаев у персонала на разных стадиях топливного цикла;

   г) от рака, вызванного у населения химическими канцерогенами, содержащимися в выбросах угольных ТЭС.

2. Потери продолжительности жизни, человеко-лет.

При этом используются следующие данные для расчета сокращения продолжительности жизни человека за счет разных факторов:

а) у персонала за счет смерти от рака – 10 лет [21]; за счет вылеченного рака – 5 лет; за счет профзаболеваний (пневмокониозы, пылевой бронхит) с учетом относительно короткого срока развития и медленно прогрессирующего течения – 5 лет [22]; за счет несчастных случаев – 30 лет [21]; за счет всех травм и случаев инвалидности, приходящихся на один смертельный несчастный случай, – дополнительно 30 лет;

б) у населения – на случай смерти от рака, вызванного облучением или химическими канцерогенами, – 30 лет [21]; за счет вылеченного рака и вызванной им инвалидности – 10 лет. Вылеченные случаи рака легких, вызванного химическими канцерогенами, в расчет не принимались.

3. Потеря трудоспособности, человеко-лет:

а) у персонала – за счет вылеченного и приведшего к смерти рака – 5 лет; за счет несчастных случаев со смертельным исходом – 20 лет; за счет всех несмертельных несчастных случаев (травм), сопутствующих одному смертельному, – дополнительно 10 лет [21], из-за профзаболевания – 5 лет.

б) у населения – за счет смерти от "лучевого" и "химического" рака – 20 лет, за счет вылеченного лучевого рака – 5 лет.

Принято, что число заболевших раком вследствие облучения вдвое превышает число умерших (т.е. 50% вылечивается).

Ущерб от облучения оценивался по линейной беспороговой гипотезе, зависимость "доза – эффект" принималась в соответствии с последними, наиболее широко принятыми представлениями [27, 28]. Оценка ущерба от пылевых и химических факторов, дозовые нагрузки на различных этапах ЯТЦ и УТЦ, приходящиеся на персонал и население, степень риска от несчастных случаев приведены в соответствии с работами [21, 27, 29].

2. ВОЗДЕЙСТВИЕ НА ЗДОРОВЬЕ ЧЕЛОВЕКА ЯТЦ

Основными факторами, способными приносить ущерб в результате функционирования ядерного топливно-энергетического цикла, являются:

– облучение персонала и населения;

– несчастные случаи с персоналом;

– профессиональные заболевания, не связанные с ионизирующей радиацией.

Необходимость учета всех указанных видов ущерба для определения "цены", которую общество платит за энергию, может быть проиллюстрирована большим числом примеров. Так, известно, что пневмокониозы у шахтеров, добывающих уголь, ввиду тяжести их течения и большой распространенности, по мнению компетентных организаций, представляют собой социальную проблему [23-25]. В то же время, согласно материалам, обобщенным в Публикации №27 МКРЗ [21], в Англии в среднем по всем отраслям промышленности в год от несмертельных несчастных случаев теряется 20 млн.

человеко-дней и лишь 0,7 млн. человеко-дней – от профзаболеваний. В большинстве отраслей промышленности (кроме угольной) профессиональные болезни дают гораздо меньший вклад в общий ущерб здоровью, чем производственные травмы. Весьма важно, что различные отрасли производства резко отличаются по частоте, характеру и последствиям травм и профессиональных болезней. Так, в ФРГ в горнодобывающей промышленности на 160 случаев смерти от несчастных случаев зарегистрировано 75 случаев смерти от профзаболеваний. Последний показатель в 10 раз выше, чем в машиностроении и химической промышленности [26]. Таким образом, имеющиеся работы по сравнению опасности ядерной энергетики, не учитывающие основных последствий производственной деятельности и ограничивающиеся лишь эффектами облучения, не позволяют вынести объективных оценок; образно говоря, они характеризуют лишь надводную часть айсберга.

В табл.1 приведены данные по общему ущербу для здоровья населения и персонала от ЯТЦ в целом.

Данные табл.1 позволяют заключить, что при определении "цены" энергии в показателях ущерба для здоровья, облучение играет относительно существенную роль по показателю числа случаев преждевременной смерти (например, половина всех случаев в ЯТЦ, при этом основное значение имеет облучение персонала). Примерно равный вклад в показатель сокращения продолжительности жизни и потери трудоспособности дают облучение и несчастные случаи на производстве (главным образом, на этапе добычи руды). Профессиональные заболевания нерадиационной этиологии обуславливают примерно одну шестую часть всех случаев преждевременной смерти и лишь несколько процентов потерь продолжительности жизни и трудоспособности.

3. ВОЗДЕЙСТВИЕ НА ЗДОРОВЬЕ ПЕРСОНАЛА И НАСЕЛЕНИЯ УТЦ

При получении энергии в угольном топливном цикле спектр воздействующих на персонал и население факторов, а также масштабы воздействия значительно шире и многообразнее. Действие ряда факторов до сих пор изучено недостаточно и количественная оценка их эффектов затруднена или невозможна. Многие факторы, например, наличие органических соединений, образующихся при сжигании угля и особенно при его превращении в жидкое или газообразное топливо, остаются даже неидентифицированными. С другой стороны, в отличие от ядерной энергетики, угольная – имеет значительный "стаж" и потому при оценке ряда ее эффектов существует больше определенности.

Как и в ЯТЦ, в угольном цикле основное воздействие на население обусловлено выбросами в атмосферу на этапе использования, то есть сжигания топлива. Остальные этапы цикла обуславливают воздействие главным образом на персонал, занятый добычей, переработкой и транспортировкой топлива.

Одним из относительно малозначащих факторов воздействия УТЦ на персонал и население является радиационный. Воздействие облучения на персонал происходит, главным образом, при добыче угля в шахтах, а на население – за счет выбросов продуктов сгорания угольными ТЭС.

ТАБЛИЦА I. ВОЗМОЖНЫЙ УЩЕРБ ДЛЯ ЗДОРОВЬЯ ПЕРСОНАЛА И НАСЕЛЕНИЯ В ТЕЧЕНИЕ ГОДА ОТ ВСЕГО ЯДЕРНОГО ТОПЛИВНОГО ЦИКЛА В ПЕРЕСЧТЕ НА 1000 МВт (эл.)

Вид ущерба	Причина ущерба					
	Облучение*			*Несчастные случаи на* производстве	Профзаболевания *нерадиационной* этиологии	*Все причины*
	персонал	население	всего			
Число *случаев преждевременной смерти*	0,40	0,23	0,63	0,25	0,15	1,0
Сокращение продолжительности жизни, человеко-лет	6,0	9,2	15,2	15,0	0,75	31
Потери трудоспособности, человеко-лет	4,0	5,8	9,8	7,5	0,75	18

* Без учета ущерба за счет генетических эффектов облучения.

Ущерб от насчастных случаев на ряде этапов угольного топливного цикла по частоте несчастных случаев не выделяется среди других видов промышленной деятельности, например таких, как этапы переработки угля или сжигания его на ТЭС. Основной вклад в ущерб данного вида дают два этапа – добыча и транспортировка. Число несчастных случаев со смертельным исходом среди шахтеров промышленно развитых стран в среднем составляет 1 случай в год на 1000 работающих [21].

Общие потери от несчастных случаев на этапах добычи и транспортировки топлива в УТЦ, приходящиеся на год работы номинальной ТЭС мощностью 1000 МВт (эл.), по обобщенным данным [14, 29] составляют:

случаев со смертельным исходом	3,4
сокращение продолжительности жизни, человеко-лет	
– за счет смертельных исходов	102
– за счет травм и инвалидности	102
ВСЕГО:	204
потери трудоспособности, человеко-лет	102

К наиболее важным этапам УТЦ относятся: добыча угля и его переработка – для здоровья персонала, сжигание угля на ТЭС – для здоровья населения. Другие этапы УТЦ по сравнению с указанными с точки зрения общественного здоровья имеют пренебрежимо малое значение.

Основной вредный фактор при подземной и, в меньшей мере, при поверхностной добыче угля, как и при добыче урановой руды, – пыль.

Несмотря на то, что антракоз – один из наиболее благоприятно протекающих пневмокониозов, ему присущи основные качества этой группы заболеваний – возникновение спустя 15-20 лет после начала воздействия высоких концентраций пыли, прогрессирование заболевания даже после прекращения работы и контакта с пылью, частые осложнения, среди которых важное место занимает туберкулез, а также воспаления легких, хронические бронхиты, бронхо-эктатическая болезнь, эмфизема легких. Несмотря на известные достижения в области изучения пневмокониозов и борьбы с ними, эти заболевания, в частности, антракоз и силико-антракоз, занимают важное место в патологии шахтеров, добывающих уголь, как причина преждевременной смерти.

В среднем по СССР заболеваемость за счет основных форм профессиональных респираторных заболеваний находится у шахтеров на уровне 0,15% или 1,5 случаев на 1000 человек в год. Исходя из необходимости работы в объеме 3500 человеко-лет для обеспечения углем годовой работы ТЭС мощностью 1000 МВт (эл.) в год, ущерб персоналу от профессиональных респираторных заболеваний за год составляет:

$$\frac{1{,}5 \text{ случаев в год} \cdot 3500 \text{ человеко-лет}}{1000 \text{ работающих}} = 5{,}25 \text{ случаев}$$

Оценивая ущерб здоровью населения от выбросов угольных электростанций, напомним, что годовая работа номинальной ТЭС в случае потребления $4 \cdot 10^6$ т угля в год (при зольности угля – 25%, проценте задержки золы на давно построенных станциях – 90%) обуславливает выброс в атмосферу около 10^5 т золы, $(1\text{-}2) \cdot 10^5$ т

ТАБЛИЦА II. ВОЗМОЖНЫЙ УЩЕРБ ЗДОРОВЬЮ ПЕРСОНАЛА И НАСЕЛЕНИЯ ОТ УТЦ, ОТНЕСЕННЫЙ К 1000 МВт(эл.) в год

	Причины ущерба					
	Облучение		Несчастные случаи на производстве (при добыче и транспортировке угля)	Заболевания нерадиационной этиологии		
	персонала в шахтах	населения за счет выбросов ТЭС		персонала при добыче и обработке угля	населения от выбросов ТЭС	*Все причины*
Число случаев преждевременной смерти	0,11	0,055	4,2	6,9	360* (10-600)	370 (20-600)
Сокращение продолжительности жизни, человеко-лет	1,7	2,2	252	35	$1\cdot10^{4}$ $7,2\cdot10^{3}$	$1\cdot10^{4}$
Потери трудоспособности, человеко-лет	1,1	1,3	126	35		$7,4\cdot10^{3}$

* За счет рака легких.

двуокиси серы, десятки тысяч тонн окислов азота, сотни тонн углеводородов, около 100 т фторидов, 10-20 т металлов. Однако в литературе имеются лишь единичные и весьма слабо аргументированные оценки возможного влияния этих выбросов на здоровье населения. Главное внимание специалистов до сих пор привлекал наиболее весомый компонент выбросов — двуокись серы. Однако наряду с ней на людей вблизи ТЭС неизбежно действует сложный комплекс факторов — пылевых частиц золы, металлов, органических соединений и т.д.

Очевидно, достаточно оправдан подход, при котором проводится корреляционный анализ между избыточной заболеваемостью, обусловленной болезнями легких и бронхов, и уровнем загрязнения воздуха всем комплексом выбросов ТЭС. Правда, при этом остается неясной канцерогенная роль микрокомпонентов атмосферных загрязнений — органических канцерогенов, ряда металлов; не выяснена роль в возникновении рака "летящей"угольной золы. Тем не менее, учитывая, что выбросы угольной энергетики весьма богаты всеми этими канцерогенно-опасными компонентами, вся угольная энергетика ответственна примерно за 30% общих атмосферных загрязнений, а основная часть рака легких вызвана загрязнениями атмосферы, представляется обоснованным отнести около 20% спонтанной смертности от рака легких за счет угольной энергетики [1, 3]. При указанном подходе расчет показывает, что работа номинальной ТЭС в течение года вызывает риск дополнительной смертности около 290 случаев, а производство 1000 МВт(эл.) сопряжено с риском 360 дополнительных смертей в год среди населения от рака легких [29].

Имеющиеся экспериментальные данные прямо указывают на способность таких компонентов выбросов ТЭС, как зола и 3,4-бензпирен, вызывать учащение раковой заболеваемости и подтверждают правомерность приведенного выше подхода [30].

Обобщая приведенные выше сведения, касающиеся возможного влияния выбросов ТЭС на здоровье населения, представляется возможным прийти к заключению, что выбросы угольной ТЭС причиняют многообразный вред здоровью, в том числе повышают общую смертность, прежде всего, от респираторных заболеваний, а также смертность от раковых заболеваний, среди которых основное значение имеет рак легких.

Обобщенные данные, характеризующие основной ущерб на всех учитываемых этапах УТЦ, представлены в табл.II.

Материалы табл.II указывают на то, что в случае УТЦ основной ущерб здоровью персонала (по показателю дополнительных смертей) приносят профзаболевания (пневмокониозы), а также несчастные случаи (в шахтах). Обращает на себя внимание, что при более значительном, чем в случае с ЯТЦ, ущербе здоровью на этапе добычи топлива, ущерб в результате выброса угольными ТЭС огромных количеств токсических веществ во много раз выше.

4. СРАВНЕНИЕ ОБЩЕГО УЩЕРБА ЗДОРОЬЮ ЧЕЛОВЕКА ОТ ЯДЕРНОГО И УГОЛЬНОГО ТОПЛИВНЫХ ЦИКЛОВ

В табл.III обобщены оценки общего ущерба здоровью человека от ЯТЦ и УТЦ, полученные в двух предыдущих разделах.

ТАБЛИЦА III. СРАВНИТЕЛЬНАЯ ОЦЕНКА ОБЩЕГО УЩЕРБА ЗДОРОВЬЮ В ТЕЧЕНИЕ ГОДА ОТ ЯДЕРНОГО И УГОЛЬНОГО ТОПЛИВНЫХ ЦИКЛОВ ПРИ ПОЛУЧЕНИИ ЭНЕРГИИ 1000 МВт (эл.)

Вид ущерба	Общий ущерб от всех причин	
	в ЯТЦ	в УТЦ*
Число случаев преждевременной смерти	1,0	370 (20-600)
Общее сокращение продолжительности жизни, человеко-лет	31	$1 \cdot 10^4$
Общие потери трудоспособности, человеко-лет	18	$0{,}74 \cdot 10^4$

* Без учета возможного ущерба здоровью от нераковых заболеваний, вызываемых неканцерогенными компонентами выбросов ТЭС (SO_2, NO_x, ртуть, свинец, кадмий и др.).

Материалы этой таблицы являются ярким свидетельством существенных отличий в "цене", которую общество своим здоровьем платит ныне за получение энергии в ядерном и угольном топливных циклах.

С удовлетворением констатируя сравнительно низкий ущерб для здоровья от ядерной энергетики, а также возможность сохранения жизней и здоровья людей при ее развитии, вместе с тем следует отметить, что и ядерная энергия имеет свою "цену". Поскольку эта цена выражается в ущербе здоровью, а также окружающей среде, следует стремиться снижать "себестоимость" ядерной энергетики по рассматриваемым показателям. Особое внимание заслуживают те вопросы и проблемы, которые могут повести к возрастанию "цены" ядерной энергии в будущем.

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DISCUSSION

B. SØRENSEN: What degree of stack effluent cleaning is achieved in Soviet coal-power plants?

E.I. VOROB'EV: An average of 90% of ash is trapped by filters at coal-fired power plants.

B. SØRENSEN: Do you have any estimates in your country of the probability of a core melt-down accident in a nuclear power plant? If so, do your estimates differ from those of WASH-1400 (about 10^{-4} per reactor-year)?

E.I. VOROB'EV: We ourselves did not estimate the accident probability in our calculations, but the probability of a core melt-down accident in the Soviet Union is extremely low -- about 10^{-9}.

J. DELFINER: Would you recommend that studies of the impacts and risks of decommissioning nuclear power plants be compared with those of decommissioning coal-fired power plants, and that such studies be included in the overall comparison of the two systems?

E.I. VOROB'EV: That is a good point. Our paper did not consider the decommissioning of plants, but the type of study you refer to should be carried out in the future.

ASSESSING SYSTEMWIDE OCCUPATIONAL HEALTH AND SAFETY RISKS OF ENERGY TECHNOLOGIES

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Abstract

ASSESSING SYSTEMWIDE OCCUPATIONAL HEALTH AND SAFETY RISKS OF ENERGY TECHNOLOGIES.

Input-output modelling is now being used to assess systemwide occupational and public health and safety risks of energy technologies. Some of the advantages and disadvantages of this method are presented and some of its important limitations are discussed. Its primary advantage is that it provides a standard method with which to compare technologies on a consistent basis without extensive economic analysis. Among the disadvantages are limited range of applicability, limited spectrum of health impacts, and inability to identify unusual health impacts unique to a new technology.

1. INTRODUCTION

Many renewable energy sources are relatively benign with respect to direct public health risks because their normal operation produces no pollutants. For this reason, comparisons of renewable energy sources with conventional energy sources tend to show the renewables as having no risk at all. Renewable sources of energy are more diffuse than conventional sources, however, and therefore consume more materials and labor per unit of energy delivered. As a result, their occupational health risks can be significantly different, not only in direct fabrication and construction of the devices themselves but also in the materials supply sectors required to support them.

The two methods that have been developed to address this issue are process analysis and input-output analysis. Process

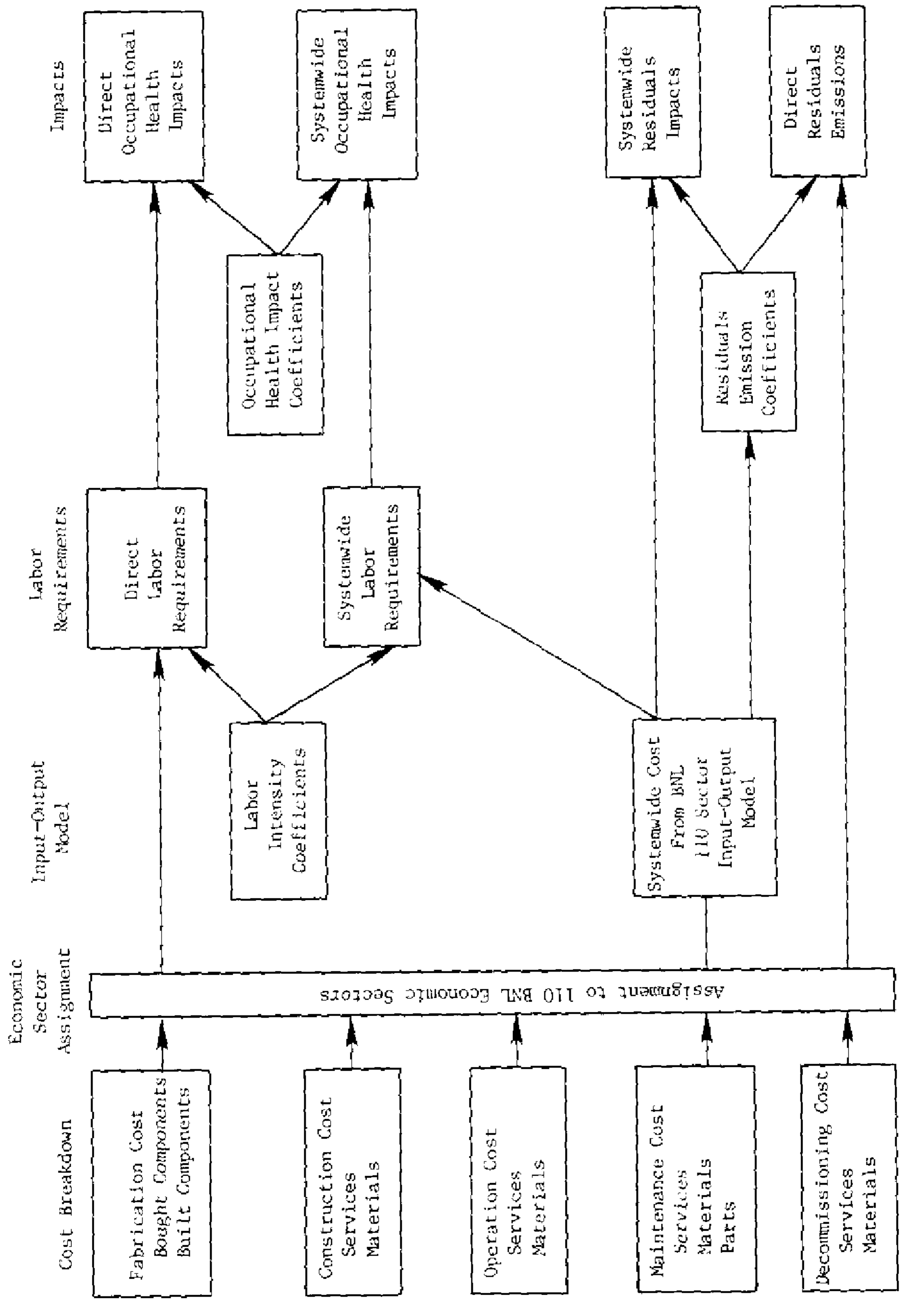

FIG.1. Health and environmental impact accounting method.

analysis is based on identification of important materials required to build and operate energy technologies, tracing flows of these materials back through the supply system to their source, and assessing occupational and public health impacts of each transfer or transformation of the materials.[1-5] The process is analogous to that used for assessing health impacts of conventional fuels by tracing through the fuel supply system.[6]

Input-output analysis for health impact assessment is based on economic input-output models which trace throughout the economic system total dollar flow created by demands by energy technologies for goods and services. Occupational and public health and safety risks are assigned for each industrial sector of the input-output model in proportion to the amount of activity generated therein (Figure 1).[7]

The advantages of input-output analysis over process analysis are straightforward. First, input-output analysis places no boundaries on an assessment. Estimates include all activity in the entire economic system. There is no need to decide which materials are sufficiently important, either economically or with respect to occupational or public health and safety, to be included. Often one cannot determine whether or not a material is important without making the assessment first, so process analysis can require difficult and tedious analyses which add little to the final result. Second, input-output analysis is based on a single set of economic and health impact coefficients, used consistently in all assessments. There are no problems with differential data availability among different technologies, and the level of uncertainty is the same for all technologies. In effect, input-output analysis provides a consistent economic and health scenario within which to compare impacts of different technologies. Third, once a suitable model is built, input-output analysis is easy. It is based on engineering cost estimates broken down by individual technology components. These estimates are readily available even for newly developing energy technologies. Often they are the only data available having the rigor required for quantitative health impacts assessment.

The limitations of input-output analysis for health impact assessment are less straightforward. The input-output method sacrifices ease of interpretation for ease of use. This paper presents some of the problems and discusses how they can be avoided or solved. It also discusses the nature of the various results that can be obtained from the input-output method and their significance for policy analysis.

2. MODEL COEFFICIENTS

The input-output model used at Brookhaven National Laboratory (BNL) has 110 economic sectors, the interindustry coefficients for which are based on 1967 data projected to 1985.[8] Although interindustry coefficients tend to be relatively stable, 18 years is clearly a long time interval over which to project what are essentially technologically based relationships. This is especially true for the near future in which relatively large-scale technological changes can be expected to result from responses to high energy costs.

More recent input-output coefficients are available and we plan to incorporate them into our model. The problem with projection into the future, however, remains.

The health and safety impact coefficients by industry used in the BNL model are limited to those accidents, injuries and fatalities that are reportable under the Occupational Safety and Health Act of 1970.[9] They include total cases, lost workday cases, nonfatal cases without lost workdays, and lost workdays for injuries and illness. These data are broken down by Standard Industrial Classification (SIC) codes to two, three and four digit levels. Because there are few reportable occupational fatalities, total fatalities are given in eight aggregated categories, broken down into 14 causes of death.

Occupational illnesses include any abnormal condition or disorder (not resulting from injury) recognized to be caused by exposure to environmental factors associated with employment. Illness not so recognized and continuing cases from previous years are excluded. Unreported illnesses can include those whose relationships with environmental factors are not understood by the persons responsible for reporting (or by anyone else, for that matter), or those having long induction times for which relationships are obscured by time and worker mobility. With respect to energy technologies, illnesses omitted include occupational cancers from radiation and organic carcinogens in fuels. Some important longer-term exposure-health relationships, such as black-lung disease in coal miners, are recognized and reported, however, so the treatment of this kind of information is not consistent.

Occupational injuries include those caused by work accidents or from a single incident of acute exposure to environmental factors associated with employment. These should be recorded if they result in death, worktime lost, medical treatment other than minor first aid, loss of consciousness, restriction of work or motion, transfer to another job, or termination of employment. These do not include injuries outside the work environment caused by delayed reactions to occupational exposure or stress, etc. Work-related suicides and automobile accidents might be extreme reactions to job stress, for example; ulcers or small accidents arising from inattention are less extreme examples.

The health and safety impact coefficients available for application in input-output models are limited, therefore, to a specific subset of health problems. Most probably this subset represents a large proportion of the total, but the proportion is unknown. The subset does not necessarily represent a large proportion of the health and safety impacts about which the general public is concerned, however. The general perception of many small accidents may be that they are a known and accepted part of work. Persons who routinely work with knives are expected occasionally to get cut. Longer-term problems such as occupational cancers not included in these data are, in contrast, viewed as unacceptable risks to which no-one should be exposed unnecessarily. The relative significance of the subset of accidents and illnesses included in these data is therefore unclear.

There are also significant aggregation problems with the input-output coefficients, the health and safety impact coefficients, and the labor intensity coefficients used to convert dollars of activity in an industry to worker-hours of labor. The problems are similar for all coefficients... industries are grouped in ways that cause combinations of industries, health risks, etc., that are unrealistic when applied to a specific analysis. Since it is not feasible to model all industries, some aggregation problems are inevitable. It is possible to minimize them by careful aggregation specifically for an analysis. The effort required, however, is disproportionately large compared to the expected gain in precision. Therefore, we must usually accept aggregation problems as a penalty for using an existing set of data.

Perhaps the most important potential limitation of the input-output method of risk assessment is that it includes only conventional risks from conventional activities. It does not consider possibilities for new risks from new technologies. Health impacts of exotic toxic metals used in high-technology photovoltaic systems are ignored, for example, as are radiation effects from conventional nuclear technologies. This disadvantage can be turned into an advantage, however, because the method is so much easier than competing methods that more time can be allocated to assessing special problems associated with specific technologies not common to other industrial activities.

Estimates of public health risks using an input-output model must be based on risk per dollar of production in each model sector. This can be based on detailed studies of the industries making up that sector or on estimates of air and water pollution generated per dollar of production, with some appropriate transformation of pollution estimates to estimates of health risk where possible. In most cases, transformation of pollution loadings to public exposure is impossible, so the loadings estimates must be used as indicators of potential public health problems. I know of only one input-output model able to make detailed estimates of environmental residuals...the U.S. Environmental Protection Agency's SEAS model.[10]

Estimates of environmental residuals are not normally expressed per unit of production, and the conversion is difficult, time-consuming, and demanding of data. I assume, therefore, that any input-output models developed in the near future to generate estimates of environmental residuals will rely heavily on those of the SEAS model.

3. SIGNIFICANCE OF MARGINAL ANALYSES

The input-output method of health risk assessment is based on estimates of impacts of fabricating components and building, operating, maintaining and decommissioning a single unit of an energy technology. For ease of comparison, results are reported per unit energy delivered. The unit analysed may not be a single facility. Instead, it is that portion of larger facilities (or that combination of smaller facilities) required to produce the unit delivered. Technology costs are disaggregated to the sectors of the input-output model and entered as final demands.

This adaptation of the normal input-output format involves a number of important implicit assumptions. First, entering the technology as final demands implies that it makes no contribution to the economic system, or that the contribution is so small that its contribution is not significant to the system. Results reflect the impact of addition of a single unit to the existing system. For a small unit of end-use energy, such as 10^{12} Btu's for example, this assumption is clearly valid. These results cannot, however, be extrapolated to large-scale additions of that technology. A system in which a new source of energy is important would have different coefficients from those now in use. Large-scale additions of a new technology would change the economic system by supplying sufficient energy to affect the relative mix of sources of energy available and by creating new industries or expanding existing industries to meet demands for new products. This affects both the interindustry transactions coefficients and, in the BNL input-output model, the energy supply coefficients.

A second implicit assumption of a marginal input-output analysis is that workers are available to provide the necessary labor. Again this assumption is valid for small units of a technology, but would not necessarily be valid for a large increase in production of new energy technologies which would produce correspondingly large interindustry shifts in employment with associated disruption and temporary unemployment. Estimates of occupational health risk would then have to account for these movements to determine where workers on the new technologies had come from, and for unemployment, the health risks of which are relatively high.

To account properly for differences between a marginal addition of a new technology and a system having that technology in significant amounts would require an economic analysis of a specific scenario. For each technology studied, economic modeling would be required to determine how the system would change in response to new demands for goods and services, to determine the changes this response would produce in the coefficients of the input-output model, and to determine how the labor force would respond to changes in the job market. This would be a huge project well beyond the scope of most health impact assessments.

4. NATURE OF RESULTS

The standard estimate of occupational health and safety risks of energy technologies has been total systemwide worker days lost per unit energy delivered. This estimate is intended to represent total occupational risk of the technology to society as a whole, and it is based on the proposition that all risk should be allocated to the driving force that caused the activity creating it. While this estimate may be useful for many analyses, it is not the only appropriate measure. It also has several problems. It ignores, for example, what the workers involved would have been doing had they not been providing the goods and services necessary to build and operate the new technology. Everyone is subject to some risk, at all times. Were the labor force otherwise employed, the workers would have either been unemployed, in which case they are subject to

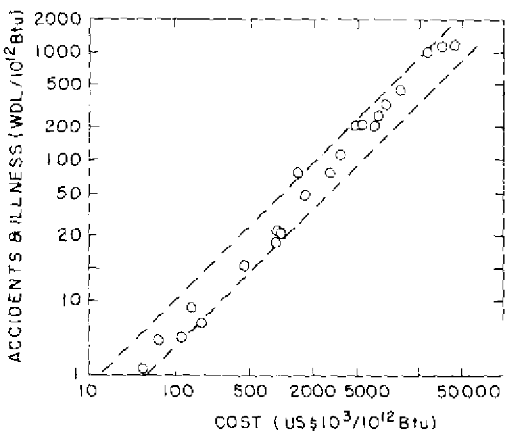

FIG.2. Systemwide occupational health impact as a function of cost. WDL = worker-days lost.

significantly different risks from employed persons, or they would have been employed in some other industry. In addition, because of the nature of the economic system and the input-output model, estimated total systemwide risk tends to be a simple linear function of cost per unit energy delivered (Figure 2). More expensive technologies employ more workers who are exposed to greater total risk. The question of whether or not these workers are better off for having been employed in this manner is not considered.

A better estimate of risk is that faced by the average individual worker per unit effort: days lost or fatalities per 100 worker-years. This estimate captures what matters to individuals, but it may or may not capture the concerns of a policy analyst. They depend on his perception of what is most important to the society. This estimate also ignores what the workers would have been doing were they not involved with the new technology. If all workers employed in producing new technologies were formerly coal miners in deep mines they would clearly be better off. Coal mining is risky.

The estimate of risk that accounts best for all important considerations is net risk per unit effort compared to some base risk. This estimate attempts to account for how workers would have been employed were they not involved with the new technology. Several bases for a net risk estimate are available without the necessity for extensive economic systems analyses. One might assume that one new technology is sufficiently like another for the workers to be considered to have come from an

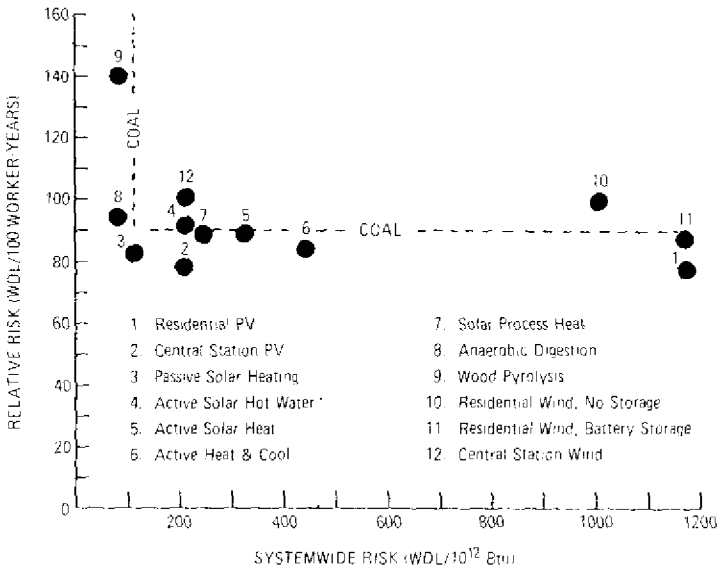

FIG.3. Occupational health and safety risks of renewable energy technologies. WDL = worker-days lost.

average pool of workers. The energy system is, however, undergoing a major change in structure, so this assumption may not be valid. A more reasonable assumption is that a new technology displaces an old one or substitutes for a competing new technology. One might assume, for example, that any energy provided by a renewable energy technology is displacing that which would have been provided by imported oil. An alternative assumption is that use of imported oil is on the decline and that any energy produced by a renewable technology would otherwise be provided by coal.

As an example, Figure 3 and Appendix 1 show total systemwide risk, average relative risk per worker, and net risk compared to coal for 12 renewable energy technologies. Appendix 2 shows the representative coal fuel cycle against which they are compared. It is apparent that major differences in interpretation of the relative risks of energy technologies are possible depending on the risk estimate chosen. Residential photovoltaics have extremely high total systemwide risk, for example, but a relative risk per worker, which is 13 percent lower than that of the representative coal fuel cycle (Table I). It is the least risky of the renewables with respect to individual workers. The high estimate of total systemwide risk

TABLE I. SYSTEMWIDE OCCUPATIONAL HEALTH AND SAFETY RISKS OF SELECTED RENEWABLE ENERGY TECHNOLOGIES

	Net Risk Compared to a Representative Coal-Steam Fuel Cycle	
Technology	Accidents & Illness (% Difference)	Fatalities (% Difference)
Residential Photovoltaics	-13.0	-7.7
Central Station Photovoltaics	-12.0	-34.0
Passive Solar Heating	-7.8	-15.0
Active Solar Hot Water	2.2	38.0
Active Solar Heat Heating	-1.1	-7.7
Active Solar Heat & Cool	-5.6	-15.0
Industrial Process Heat	-2.2	-7.7
Anaerobic Digestion	4.4	54.0
Wood Pyrolysis	56.0	-35.0
Residential Wind, No Storage	11.0	-34.0
Residential Wind, Batteries	-2.2	-35.0
Central Station Wind	11.0	-32.0

arises from high employment required to produce the technology. This could appear to many to be the ideal technology ... one which employs many persons at low-risk jobs.

In contrast, wood pyrolysis has low total systemwide risk and high relative risk per worker. This is in part because most of the cost of wood pyrolysis system is the wood fuel, and logging is among the riskiest of industries.

Note from Table I that a different interpretation would be made were fatalities per unit effort used as an estimate of risk. This difference arises in part out of real differences in relative accident and injury rates compared to fatality rates among industries, and in part from aggregation problems in the fatality data. Statistical aggregation of fatality data is required to obtain meaningful sample sizes, and the aggregation combines industries by general type rather than relative risk. Thus occupations having large differences in relative risk are combined, reducing the sensitivity of the model to differences among technologies and increasing the confidence interval of the results.

To compare health risks of competing technologies, they should be normalized to the same end use. The results presented here have not been so normalized, partly because they cannot be. Wood pyrolysis and sludge digestion, for example, produce gas, and end-use devices to burn that gas are not included in the analysis. They could be added, but little would be gained since gas-based end-use devices have limited substitutability for electrical end-use devices. Gas is used primarily for residential heating. While it is possible to generate heat by electricity, there are better ways to obtain heat for this purpose, so there is limited likelihood in the future that gas and electricity will be direct substitutes one for another. If they cannot reasonably be substituted in some common end use, then they should not properly be compared directly with respect to health impacts. Many of the renewable technologies presented in this report, however, produce electricity, the end use of which is independent of the source. Some (domestic hot water, domestic heating and cooling) produce energy for which electricity is a likely substitute. In such a case, the electrical end use device (water heater, heat pump) and transmission lines should be added to the analysis.

5. CONCLUSIONS

Input-output modeling for health risk assessment has significant advantages over competing methods, but some caution should be used in interpreting its results. These results are not generally applicable to a large range of circumstances. Basically, they reflect the risks associated with a small increase in a specific technology over the existing system. They do not apply to well-developed new technologies providing significant portions of the total national energy supply, and they do not apply to designs based on significant technological advancements over that currently available.

Some care should also be taken that comparisons among competing technologies be made on an appropriate basis. To be compared directly, technologies should be directly substitutable in the same end use.

Units of comparison should be relative risk or net risk to avoid misinterpretation of employment effects. Choice of a basis for net risk assessment depends on decision-makers' perceptions of the role of the technology under assessment.

Appendix 1

Systemwide Occupational Health and Safety Risks of Renewable Energy Technologies

Technology	Labor	Accidents & Illness		Fatalities		Net Risk[a]	
	(100 WY[b]/ 10^{12} Btu)	(WDL[c]/ 10^{12} Btu)	(WDL/ 100 WY)	(Deaths/ 10^{12} Btu)	(Deaths/ 100 WY)	(WDL/ 100 WY)	(Deaths/ 100 WY)
Residential Photovoltaics	15	1200	78	0.17	0.012	-12	-0.0010
Central Station Photovoltaics	2.7	210	79	0.023	0.0086	-11	-0.0044
Passive Solar Heating	1.4	110	83	0.015	0.011	- 7	-0.0020
Active Solar Domestic Hot Water	2.2	210	92	0.041	0.018	2	0.0050
Active Solar Domestic Heating	3.7	330	89	0.044	0.012	- 1	-0.0010
Active Solar Heating & Cooling	5.2	440	85	0.057	0.011	- 5	-0.0020
Solar Industrial Process Heat	2.8	250	88	0.033	0.012	- 2	-0.0010
Anaerobic Digestion	0.83	78	94	0.017	0.020	4	0.0070
Wood Pyrolysis	0.55	78	140	0.0048	0.0085	50	-0.0045
Residential Wind - No Storage	10	1000	100	0.087	0.0086	10	-0.0044
Residential Wind - Batteries	13	1200	88	0.11	0.0084	- 2	-0.0046
Central Station Wind	2.1	210	100	0.018	0.0089	10	-0.0041

[a]Compared to a representative coal fuel cycle having 90 WDL/100 WY and 0.013 Deaths/100 WY.

Source: Rowe, M.D., and P.J. Groncki. Ref. (7).

[b]WY = worker years.

[c]WDL = worker-days lost.

Appendix 2

Systemwide Occupational Health and Safety Risks of a Representative Coal Fuel Cycle

Technology	Labor (100 WY/ 10^{12} Btu)	Accidents & Illness (WDL/ 10^{12} Btu)	Accidents & Illness (WDL/ 100 WY)	Fatalities (Deaths/ 10^{12} Btu)	Fatalities (Deaths/ 100 WY)	Net Risk[a] (WDL/ 100 WY)	Net Risk[a] (Deaths/ 100 WY)
Eastern Strip Mine	0.20	17	87	0.0028	0.014	23	0.0063
Mixed Coal Train	0.24	21	88	0.0033	0.014	24	0.0063
Coal-Fired Power Plant	0.51	48	93	0.0068	0.013	29	0.0053
Wellman-Lord Scrubber	0.25	22	88	0.0032	0.013	24	0.0053
Total	1.20	108	90	0.0160	0.013	26	0.0053

[a]Compared to 1978 industrywide averages of 64 WDL/100 WY and 0.007 Deaths/100 WY.

Source: Rowe, M.D., and P.J. Groncki, Ref. (7).

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DISCUSSION

B. SØRENSEN: I would like to make two points. First, the health impact you calculate for new technologies assumes in effect that these technologies arise in existing facilities. In many cases a new technology would require a new production line, with different options in terms of labour intensity and automation among other things, so there is really no unique impact.

Secondly, since your system-wide effects scale with cost, your comparison of impacts of new energy systems really reflects your cost estimates. I find this questionable (in Denmark, for example, residential wind power is cheaper than central wind power).

M.D. ROWE: With regard to your first point, production of the subcomponents of an energy technology must be assigned to the industries most similar to those expected to exist in the future; these may well be different from those currently involved. For example, having assigned windmill blades to the aircraft industry, I recently learned that wooden blades are superior to metal and fibreglass ones and that these new blades are in fact built by the pleasure-boat industry. One would not, a priori, assign windmill blades to the boat-building industry without careful consideration of the nature of the production process.

As for your second comment, the technologies used in our preliminary assessment are not intended to be generally representative of future processes but are specific technologies described in the literature.

H. INHABER: In your reply to Dr. Sørensen, you implied that the assumptions made in input-output analysis are as great as in process analysis, yet you stated in your paper that one of the drawbacks of process analysis was the large number of assumptions which had to be made. Could you comment on this?

M.D. ROWE: Both methods require many important assumptions. Input-output analysis requires a few large assumptions while process analysis requires many small ones. In my opinion, a few large assumptions are easier to understand and take into account.

H. INHABER: I have two further points. First, assume that you have two solar collectors (or nuclear reactors), identical in every respect except that one costs twice as much as the other. According to your Fig.2, this would mean that the risk associated with one collector would be twice that associated with the other. This does not seem reasonable.

Secondly, you stated that the methods described gave the total loss to society. This is not so: occupational risk is included, but public risk has been omitted.

M.D. ROWE: In reply to your first point, the transaction coefficients of the input-output model are industry averages. A technology costing more is therefore assumed to require more goods and services to produce. Differences attributable to more expensive materials of the same general kind must be accounted for, with the technology components entered into the input-output model carefully specified.

As regards your second remark, the method itself is capable of assessing total health impacts. This paper, however, discusses only the occupational health component.

W. PASKIEVICI: Have you tried to apply both techniques (process and input/output analysis) to the same fuel cycle? If so, what results did you obtain, and are the differences significant?

M.D. ROWE: We have made a preliminary comparison of a coal fuel cycle. The results were fairly similar but the uncertainty of the data was very high. We plan to carry out a more detailed and carefully controlled comparison of a photovoltaic energy system within the next few months.

A.J. DA CONCEIÇÃO SEVERO: In Table I, the number of fatalities under the heading 'active solar hot water' is 38% higher than for a representative coal steam fuel cycle. Could you explain this?

M.D. ROWE: The input-output model contains 12 100 individual transaction coefficients, so it is rather like a 'black box'. Analysis of the structure of its results can explain some such differences. I refer you to the report in Ref.[7] from which Table I was taken.

R.M. BARKHUDAROV: You use the same risk concepts (mortality, reduced lifespan and so on) as other authors. You also present a detailed economic analysis of the technologies involved, and I should like to know whether you also carried out an economic assessment of the resultant health impacts – in other words, did you assess the economic loss due to death or reduced lifespan?

M.D. ROWE: No, we did not, but we plan such assessments in the future.

J. DELFINER: Did you include in your analysis the impacts and costs of setting up medical facilities and training medical personnel to handle accidents associated with nuclear power plants and accidents resulting from the transport of hazardous fuel materials to power plants?

M.D. ROWE: No. These are specific problems which are not well suited to analysis by the input-output method.

C. RICHMOND *(Chairman)*: If I understood you correctly, you said that the input-output methodology was not all 'written down' and recorded because of its complexity. If this is the case, I would urge you to do so in order to make it available for use and review by the scientific community.

M.D. ROWE: I was referring to the projection into the future of changes in input-output inter-industry transaction coefficients. A great deal of experience and intuition is required, and these intuitive projections tend not to be well documented.

OCCUPATIONAL RISK INDUCED BY CONSTRUCTION OF ENERGY-PRODUCTION CHAINS

Methodology and evaluation in France

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Abstract

OCCUPATIONAL RISK INDUCED BY CONSTRUCTION OF ENERGY-PRODUCTION CHAINS: METHODOLOGY AND EVALUATION IN FRANCE.

The Centre d'étude sur l'évaluation de la protection dans le domaine nucléaire has been conducting studies for several years on the comparison of health and social effects in various electricity production chains; the global results of this research are presented in paper IAEA-SM-254/51, by Fagnani et al., at this Symposium. Attention should be paid to a particular area in risk evaluation: the calculation of effects during the construction stage. The method used, the Leontieff macro-economic analysis, should be examined in detail, for it is a powerful tool whose uses go well beyond risk calculation. Moreover, it is at this stage that the richness and ambiguities of occupational risk measurement appear most clearly. As this is a major factor in the evaluation of risks generated by techniques of electricity production, it is important to show how it is calculated and, especially, how comparative evaluations have been interpreted. Three conventional production technologies are examined (PWR, oil and coal) and two solar-powered (a thermal system extrapolated from the Themis plant built in France and a photovoltaic process). The production scenarios are adapted to the French context, with the location of the installations and the origin of the fuel clearly spelled out. The results of this evaluation are striking: with the exception of coal, which is a labour-intensive industry, a very large portion of occupational risk can be traced to the construction phase (from 67% for the PWR system to 97% for the photovoltaic process). Even in total health risk (occupational and public), construction accounts for a preponderant share. In fact, this risk should be compared to average occupational risk in the economy at large, i.e. that to which workers in electricity production chains would be exposed if they worked elsewhere. This is the meaning of the 'reference level' pointed out in the study.

INTRODUCTION

Like other studies, CEPN's comparisons of various electricity production chains during the operating stage (nuclear, coal, oil, thermal solar and photovoltaic) point to the quantitative importance of occupational risk [1]. However, as we are dealing with heavy industrial installations in many cases, relatively little manpower is required for operation in comparison with the construction stage. It seems that the risk related to operation is only the tip of the iceberg if global risk, including construction, is evaluated.

1. PRINCIPLES OF COMPARISON

The risks studied have been normalized to the quantity of energy produced, the unit used being TW · h.

Five electricity production chains – coal, oil, nuclear (PWR), solar thermal and photovoltaic – are compared. The need to take into account the risks of entire production systems leads us to include in the comparison not only the power plant but also the other elements, from the mining of fuel to the disposal of waste. To make a valid comparison, the quantity of electricity produced is not sufficient; it should have the same usefulness or value. The best way to satisfy this criterion is to have each system play the same role in a same-supply-and-demand scenario. Next, as a given technique is not enough to define a real chain, the geographical, logistic, economic and regulatory context is specified in each case by an energy-acquisition scenario, except in the case of the solar system [1]. One must then calculate the impact of the construction necessary between 1980 and 1990 for the system under consideration so as to meet demand by 1990.

Two indicators commonly used in this type of study measure risk: individual occupational risk is quantified by the number of deaths in relation to the number of worker-years. Collective occupational risk is calculated by using an aggregate, the number of 'equivalent working days lost' owing to work accidents and occupational morbidity[1]. Economic analysis also leads us to use other values, in particular manpower and investments.

2. METHOD OF EVALUATION

To calculate impacts due to setting up installations, two methods are possible. The first consists in counting manpower and risks on the worksites themselves. This has so far been the method commonly used to introduce construction into risk evaluation [2–5]. In France, studies of employment due to construction were also available, but results differ significantly. This is because certain surveys look only at the worksites while others also count suppliers, and others even count sub-contractors.

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[1] The equivalent used is 1 death = 6000 working days lost. This value is used both in French and US statistics.

The problem is to define the limits of what constitutes the industrial activities that go into the construction of an installation. Moreover, these limits should be homogeneous for all the equipment. To obtain coherence, a simple method is to adopt the macro-economic approach found in the Leontieff model.

To take into account activities 'induced' in the necessary economy, one must formalize the technical and economic interdependencies that exist in the national system of production. This is the point of departure for the Leontieff model, also known as input-output analysis. Companies manufacturing the same product are grouped into categories called 'branches'. The national accounting system gives production figures for each branch and its purchases in other branches, called 'intermediary consumption', in the form of an input-output. It is then possible to define a_{ij} 'technical coefficient', a_{ij} being the quantity of product i necessary to manufacture one unit of product j. The underlying hypothesis of the method is that these coefficients are constant and represent technical constraints in production.

With the use of matrix A of these coefficients, it is possible to calculate the quantities of each product necessary to satisfy the ultimate demand for a given product. To satisfy demand Y, it is necessary to produce $X = AY$; to produce X, one must produce AX, and so forth. The process is convergent and, in the end, one must produce $[I + A + A^2 + ... + A^n + ...]\,Y$.

To return to the problem under consideration here, investments are broken down into demand for buildings, machinery and so forth (direct impacts : Y); the increase in production generated in each branch by these demands is then deducted (indirect impacts: $[A + A^2 + ... + A^n + ...]\,Y$). Once this principle is adopted, the problem can be broken down into four stages, in the following order:

Stage 1: An inquiry is conducted among enterprises in the electricity-production chains. One must not only discover what investments are needed for construction of all the elements in the chain but also evaluate them in terms of national accounting. The investment for specific equipment (e.g. a PWR nuclear plant) can then be compared with a conventional industrial demand. These data are then aggregated for all the elements in the fuel cycle apportioned to each industrial branch to give the total investment necessary to set up the production system.

Stage 2: The investment is entered into the Leontieff model, which gives the necessary increase in production to satisfy demand.

Stage 3: The manpower is determined by dividing for each branch the growth of the production by the productivity (given by the National Accounting System) [6].

TABLE I. RISK RATE BY FRENCH NATIONAL INSTITUTE OF STATISTICS AND ECONOMIC STUDIES (INSEE) BRANCHES
(France – 1976 – Caisse nationale d'assurance maladie)

INSEE branch	Accidental deaths per 1000 worker-years	Working days lost due to accidents per worker-year
T07 Ferrous metals	0.129	3.106
T10 Glass manufacturing	0.06	2.37
T11 Chemicals	0.125	1.70
T14 Mechanical construction	0.118	2.95
T15 Electrical construction	0.06	1.66
T16 Transport material	0.02	2.47
T17 Shipbuilding	0.20	4.83
T20 Wood industry	0.12	2.89
T24 Building and public works	0.40	5.65
Average for France	0.14	2.19

Stage 4: Occupational risk is deduced by using the risk rate for each industrial activity established by the Caisse nationale d'assurance maladie [6] (Table I).

3. CONSTRUCTION OF ENERGY-PRODUCTION SYSTEMS IN FRANCE DURING THE 1980s

Initially, risk life manpower and the financial investment are 'capitalized' in equipment that is not yet in operation, and are completely independent of production conditions. One approach is therefore to describe the impacts of construction as they will occur during the 1980s, the decade chosen to define scenarios.

Investment structures necessary for the various conventional chains are similar; they are characterized by a high proportion of 'mechanical construction', which is in fact typical of most conventional industrial installations (see Table II). Solar techniques are different, for they rely more heavily on chemicals and glass.

We can jump one stage to look at the manpower employed in construction (Fig. 1). The input-output model is characterized by a multiplier effect, here 1.7; i.e. for 100 jobs directly derived from the investment, 70 are 'induced' in other branches of the national economy. This distinction appears in the figure and can also be seen in the direct employment category. We have also shown the proportion of these jobs within the country. The figure is quite low for coal, as almost all mining and logistic investments are made outside France. The highest proportion of manpower used in France is found in the nuclear chain: during an average year it accounts for 120 000 persons. However, by 1990 the nuclear industry will probably not employ more than 25 000 persons to operate the power plants.

Using accident rates, we can now examine occupational risk. The average annual risk for construction over this decade on a world level is 300 000 equivalent working days lost for the oil scenario and 550 000 for the other two scenarios. These figures should be compared with those of annual operating risk: 60 000 working days lost for oil and 150 000 equivalent working days lost (including induced radiological effects) for the nuclear chain. On the other hand, the annual risk for coal is notably higher during operation (1 700 000 equivalent working days lost) than during the construction stage.

This paper highlights the contrast between a labour-intensive industry (coal) and the two capital-intensive systems (oil and nuclear). The ranking is not the same as for operation. Moreover, risk during the construction stage is nearly proportional to the manpower employed. Individual risk is on the order of 0.16 deaths per 1000 man-years.

4. RISK IN RELATION TO ENERGY PRODUCED

To compare total occupational risk with production, the following formula is used:

$$\text{Total risk} = \frac{\text{Construction risk} + (\text{operating risk}) \times (\text{operating lifetime})}{(\text{annual production}) \times (\text{operating lifetime})}$$

It is thus possible to calculate the risk for each system (see Table III). Here the justification for taking construction into account becomes patently clear: it represents almost the entire risk for three systems and almost two-thirds for the nuclear chain.

The differences obtained are great enough to obtain a significant ranking for oil, nuclear, solar and coal. This is true for average values, but a marginal approach can reverse this ranking. Thus the risk associated with an oil-burning

TABLE II. INVESTMENT STRUCTURE IN ENERGY-CHAIN INSTALLATIONS SET UP BY 1990
(296 TW · h net per annum)

Distribution by branch	Oil	Coal	Nuclear	Solar plants	
				Thermal	Photovoltaic
T07 Ferrous metal	5%	6%	4%	0.8%	
T08 Non-ferrous metal					3%
T10 Glass-making industry				17.1%	29%
T11 Chemicals				25.6%	19%
T14 Mechanical construction	55%	47%	57%	23.6%	
T15 Electrical construction	17%	15%	16%	16.6%	21%
T16 Transport material		2%	4%		
T17 Shipbuilding	3%	12%			
T20 Wood industry		2%			
T24 Building, public works	20%	16%	19%	16.3%	28%
Total cost (in 10^{12} French Francs 1.1.79)	89 000	158 000	154 000		
Cost of kW(e) (in French Francs 1.1.79)				4500	1700

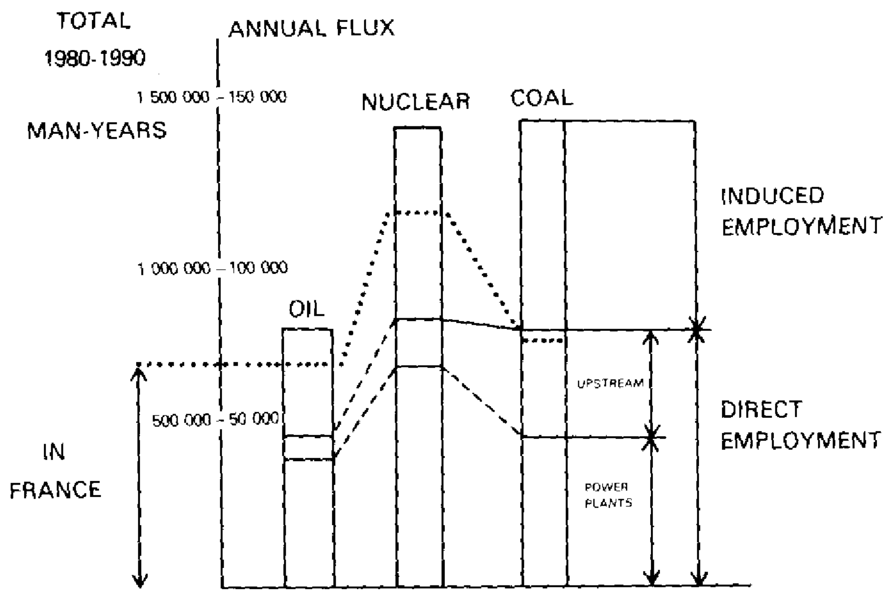

FIG.1. Employment created by construction according to scenarios of conventional energy chains: period of construction 1980–1990 (296 TW·h in France in 1990).

plant using North Sea oil is estimated at 1500 equivalent working days lost for 1 TW · h. At the other extreme, the risk of a coal-burning plant located next to an open-air mine with broad seams can be estimated at 1000 equivalent working days lost for 1 TW · h, i.e. ranking is reversed.

Alongside collective risk, it seems useful to consider individual risk (see Table IV). Here, ranking is not the same: the thermal solar system presents the least risk for a worker. Coal is the most dangerous, while the nuclear chain seems relatively safe. As individual risk decreases for the most recent chains, technical progress might be one explanation. The small proportion of specific risks (theoretical effects of radiation exposure, pneumoconiosis, etc.) is striking when one remembers that the debate on nuclear energy and protection measures adopted have centred on just these risks.

In addition to individual risk, the required manpower accounts for the differences found in collective risk. A reference level, the risk encountered if the workers were working elsewhere (425 equivalent working days lost for oil, 985 for PWR, 1395 for coal, 2950 for solar thermal and 1320 for photovoltaic), is necessary when comparing collective risk. Surprisingly, some chains, such as solar thermal, seem to be risk-saving from this point of view.

TABLE III. COLLECTIVE OCCUPATIONAL RISK INCORPORATED IN PRODUCTION OF 1 TW · h
(Unit: number of equivalent working days lost)

	Oil	PWR	Coal	Solar thermal	Solar photovoltaic
Total risk	432	1326[a]	5930	2620	1490
Share of construction	86%	67%	11%	87%	97%

[a] Without taking into account 55 equivalent working days lost for dismantling.

TABLE IV. INDIVIDUAL OCCUPATIONAL RISK INCORPORATED IN PRODUCTION OF 1 TW · h
(Unit: number of deaths per 1000 man-years)

	Oil	PWR	Coal	Solar thermal	Solar photovoltaic	France (average)
Total	0.16	0.22	0.62	0.13	0.16	0.16
	(0.16)	(0.15)	(0.50)	(0.13)	(0.16)	(0.14)

Note: Numbers in parentheses are common risk.

5. CONCLUSION

Inclusion of construction in risk evaluation is valid, since it constitutes almost the entire risk in certain systems. The transfer of risk from operation to construction is clearly shown in the most capital-intensive chains (solar, oil). One of the results is the small portion of specific risks in the chains. While theoretical calculations give seven radiologically induced effects for workers in the nuclear chain in 1990, there will be 23 deaths on building sites and in workshops during the construction stage. Though this is common for such a major industrial undertaking, and will have no effect on public opinion, it is a large risk, which justifies preventive measures, even more so because construction is undoubtedly one of the areas where the global risk of energy chains can be effectively reduced.

The Leontieff analysis goes well beyond the limits of occupational risk. In this way, collective risk can be relativized in relation to employment created by the chains. Above all, this type of macro-economic analysis evaluates the needed parameters and provides the logical framework in which a cross-comparison between risks and other essential social and economic criteria can be made.

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ASSESSMENT OF ENVIRONMENTAL IMPACT OF COAL-CONVERSION AND OIL-SHALE TECHNOLOGIES

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Abstract

ASSESSMENT OF ENVIRONMENTAL IMPACT OF COAL-CONVERSION AND OIL-SHALE TECHNOLOGIES.

The paper summarizes current efforts of the US Environmental Protection Agency in assessing environmental impact of coal-conversion technologies and oil shale, referred to as synfuel. These efforts include the identification of representative processes for each technology which will be evaluated, e.g. surface or subsurface mining and surface retorting for oil shale; solvent-refined coal (SRC II) for direct conversion of coal; and fixed-bed gasification in conjunction with the Fischer-Tropsch process for indirect conversion. The assessment begins with a description of the technology in terms of release rates of specific pollutants. These release rates or 'source terms' are inadequately known since no full-scale commercial plants exist at present, but they can be estimated from pilot-plant data and process chemistry. The next step is determination of which of the released chemicals can be grouped together for the subsequent stages of the evaluation. A health risk analysis is then conducted in three steps: exposure assessment, health effects assessment, and health risk analysis. Because the data for health risk analysis are known to be less than adequate in some areas, the health risk analysis will trigger research efforts to provide the necessary data. The paper reviews the information available at present in terms of applicability to the health risk analysis. Areas where little or no data are available are identified. Particular emphasis is placed on the terrestrial foodchain as a potential exposure route.

INTRODUCTION

The desire to obtain liquid fuels from coal and to recover liquid fuel from shale is over a century old. Already in 1860 there were oil-shale operations in the U.S., and the first patent to hydrogenate coal was awarded to Bergius in 1919 [1]. At the height of the Second World War, the German production of liquid fuels from the Fischer-Tropsch Process [2] reached 4 x 10^9 kg/year (100 000 bbl/day) and a number of plants in different countries used the Fischer-Tropsch Process and the low-pressure Fischer-Pichler Process [3]. However, the economics of these technologies as compared to petroleum forced a moratorium on all activities until the price increases by petroleum-producing countries changed the condition in favor of alternative fuels to petroleum.

The adverse environmental and health impacts of coal handling and coal chemistry have been well studied during the last two decades. For example, the health hazards of coke-oven emissions are known, as are causal relationships between the presence of coal-derived polycyclic organic matter and adverse health effects. The potential adverse health effects of a number of other chemical compounds and certain operations associated with coal-conversion technologies and oil shale are also known. However, no comprehensive assessment of the risks, both health and environmental, associated with the emergence of synfuels has been carried out. Major reasons for the lack of risk analysis data are significant changes in the technology which have taken place and are continuing to occur. Although the basics of synfuels are often a century old, the new engineering advancements have improved the yields and have decreased potential environmental releases considerably. Because large-scale applications of the new and improved technologies permit options in the choice of control technologies, it is now possible to optimize potential hazards in terms of economic and other input resources. Therefore, risk analysis may provide the basis for genuine economic and labor savings without adversely affecting human health and other environmental factors. EPA's program in risk assessment of synfuels began in 1980 with a series of planning workshops. The program is being conducted in association with the Department of Energy, a number of national laboratories, other U.S. agencies and academic institutions.

ELEMENTS OF RISK ANALYSIS FOR SYNFUELS

Conventional risk analysis [4] has three major elements: pollutants release rates (source terms), multi-media transport (exposure assessment) and effects assessment. In the case of health risk analysis, adverse health effects are expressed as mortality (deaths) and morbidity (illness). Environmental effects are assessed in terms of economic loss (crops, fisheries resources), losses of species, loss of habitat and other ecosystem changes.

The first comprehensive risk assessment was conducted for nuclear power. For this assessment, release rates in terms of gamma-radiation exposure were established for a given nuclear power scenario. In addition, release rates of various radionuclides were estimated for normal and abnormal operations. Exposures were then established using appropriate models, and dose was calculated using known metabolic data for each radionuclide separately. In this assessment, every pollutant was followed from its point of release through every media pathway up to the biological site. Subsequently, the radiation dose from every radionuclide was summed and the combined dose was related to an effect.

In the case of synfuels, the exact numbers of chemical compounds involved in the process and in the product is unknown. However, sufficient chemical analysis has been conducted to know that the number exceeds several thousand. An approach similar to that of nuclear power would have been required to establish multi-media transport of each chemical compound along with dose-response curves for various biological endpoints. Clearly, the evaluation of every compound is an impossible task. The number is too large to be evaluated even if substantial resources would be devoted to it. Obviously a concept similar to the external radiation exposure from gamma-radiation is inappropriate.

Therefore, new approaches had to be worked out to accommodate the very large number of chemical compounds and yet cover all of them. EPA's approach uses a compromise in risk analysis. The chemical compounds are categorized in various units; each unit consisting of either a class of compounds, a single element or compound or a mixture.

The objective of this categorization is to simplify the initial effort which establishes the magnitude of hazard within each category so that the subsequent effort can concentrate on those categories with highest potential adverse impact. Once a category, henceforth referred to as a risk analysis unit (RAU), has been found to be of little or no significance, it will be dropped from further consideration. Conversely, an RAU with a high degree of potential risk will be subdivided into new RAUs and studied. The process will be repeated as necessary.

Obviously, the selection of RAUs will greatly impact the risk analysis process. Examples of RAUs are as follows:

1. Polycyclic aromatic matter.

2. Oxides of nitrogen, carbon and sulfur.

3. Benzene.

4. Monocyclic aromatic hydrocarbons except benzene.

5. Certain trace metals.

6. Arylamines.

7. Saturated straight chain aliphatic hydrocarbons.

8. Phenols.

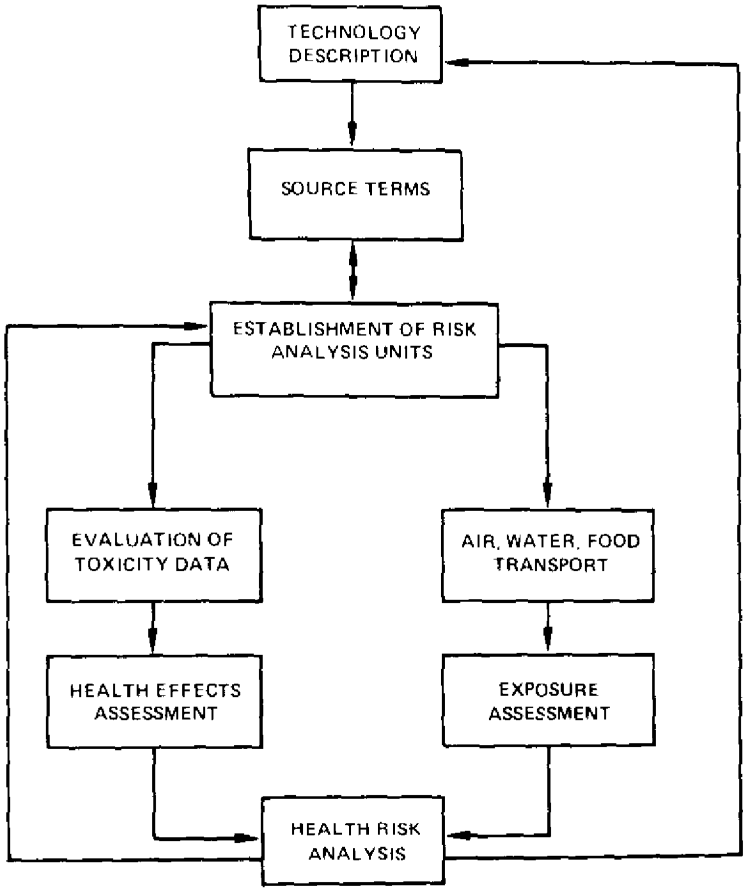

FIG. 1. Schematic diagram of risk assessment.

RISK ANALYSIS PROCESS

As described previously, there are three major elements in risk analysis: the source terms, the multi-media transport and the effects assessment. Figure 1 shows a schematic description of the process. The process can be illustrated using the example of polycyclic organic matter (POM), as an example of an RAU. The source terms of POM for various technologies, although subject to considerable uncertainty, can be estimated. For example, the concentration of POM in coal gasification has been established.

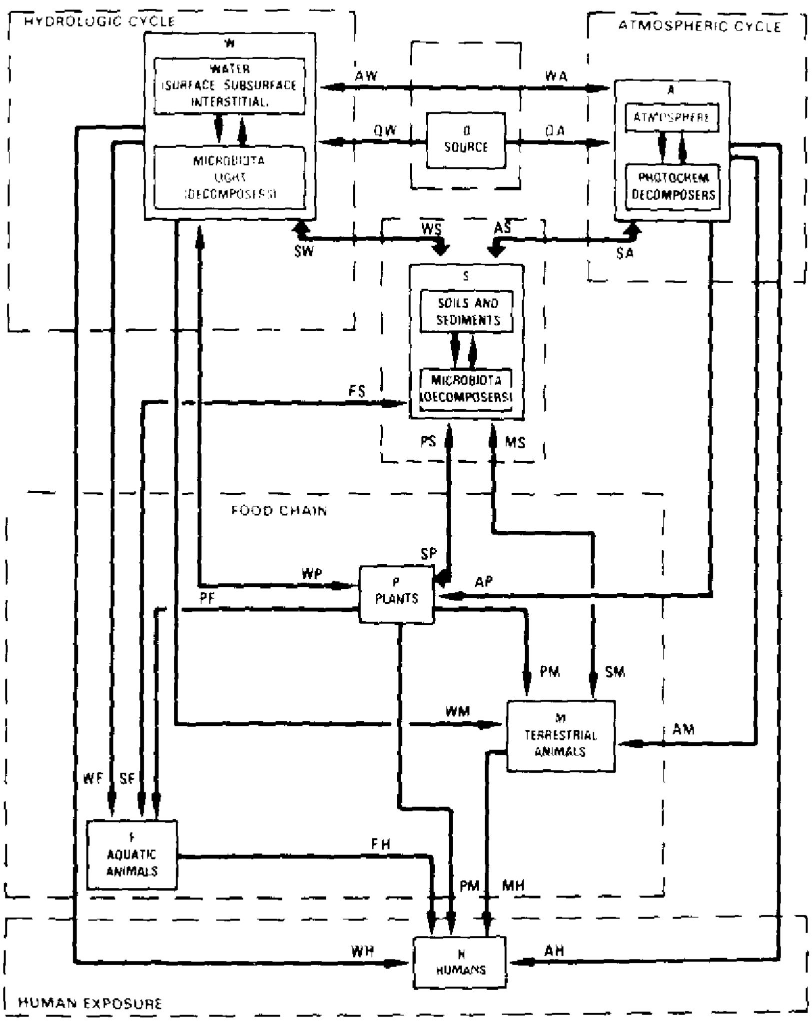

FIG.2. Exposure assessment model.

Similarly, the concentration of POM in diesel fuel or gasoline is not expected to exceed levels above several ppm. EPA's Alternate Fuels Group is in the process of defining these source terms and publishing them in a series of Pollution Control Guidance Documents (PCGD). These documents will include the source term data for the synfuel processes configured both with little or no control technology and with specific control options.

The multi-media transport leading to exposure can be assessed using a model shown in Figure 2. The model is probably the simpliest multi-media model and will be refined depending upon the significance of each route. In the case of POM, the atmospheric transport (AH) will probably dominate the exposure, although in certain specific cases, other routes such as AP-PM (deposition on agricultural plants and uptake by humans) or WP-PM (the corresponding water route) may be important.

It is important in the case of atmospheric transport to assess the lifetime or persistence in the atmosphere. Lane and Katz [5] have studied the photodecomposition of POM. According to these authors, the persistence half-life of Benzo(a)pyrene deviates from 0.3 - 0.6 hours depending upon the concentration of ozone. The presence of light and ozone changes the range of the half-life from 0.08 to 5.3 hours. Half-life values for other members of POM using the same conditions deviate from 2.9 to 52.7 hours without light and 0.9 to 14.1 hours with light. The subject is certainly more complicated because the decomposition almost surely will follow a reaction rate exceeding first order and thus it will be concentration-dependent. A recent review by Turner [6] discusses atmospheric dispersion models and their usefulness for various applications. These models meet the requirements of risk analysis. The report by the National Academy of Sciences [7] discusses various routes of exposure to humans, including the foodchain. According to NAS, foodchain is a minor contributor to the exposure to POMs although the information is too inadequate to draw a final conclusion. However, if additional data supports this contention, the routes of AP-PM and WP-PM would be deemed unimportant.

The biological effects of POM have been studied for over a century. There is sufficient information to assign a risk factor to POM [7]. The quantitative information on POMs appears adequate to allow one to relate release rates of POM to a POM exposure concentration to a hypothetical cancer risk. The final value will be a range rather than a value reflecting the uncertainty in the existing data base.

APPLICATION OF THE RAU CONCEPT

Once the risk analysis for all RAUs is completed, it is easy to determine if changing the composition of the releases or that of various fuels will have little or no effect on evaluation of risk. For example, if the fuel is composed of n RAUs designated as $(RAU)_1$, $(RAU)_2$ ... $(RAU)_n$, and if the concentration of each RAU in the release is given by $C_1, C_2, C_3, \ldots C_n$, the total risk R is estimated from the individual risks, r, to be as follows:

$$R = r_1 C_1 (RAU)_1 + r_2 C_2 (RAU)_2 + \ldots r_n C_n (RAU)_n$$

This summation approach disregards potential synergistic or antagonistic effects of various RAUs. However, given the lack of data for various chemical compounds constituting an RAU, it will be difficult if not impossible to consider synergism and antagonism for all chemical compounds. A sensitivity analysis will be performed if it is suspected that synergism or antagonism may be important to decide if data should be sought through the research program.

SUPPORTING RESEARCH

Risk analysis in association with a sensitivity analysis will have the added benefit of prioritizing the needed research by indicating where the significant data are lacking or where the uncertainty needs to be reduced. Due to the cost of long-term carcinogenicity studies, EPA will not support these studies for synfuels; instead research data from the other agencies will be used. Appropriate oversight groups have been set up to assure the applicability of these data to the risk analysis efforts.

Until the risk assessment defines additional needed research, the supporting research will include a major effort in reproductive effects both teratogenic and genetic and also an effort to validate a new short-term test for leukemogenicity. Foodchain studies are also included since this human exposure pathway for RAUs is least well understood.

As expected, a major emphasis in research is likely to be placed upon human data. A combination of epidemiology, clinical and exposure assessment will be used to obtain human data. For obvious reasons, workers in various industries are of primary interest. The application of RAU concept suggests that human data may be obtained from industries unrelated to synfuels. For example, human data for POM may be obtained from coke-oven workers, and similarly, data on aliphatic hydrocarbon from refinery workers and for amyl amines from the appropriate chemical industry.

CONCLUSIONS

The EPA program of risk analysis for adverse health and environmental effects of synfuels is now under way. The categorization of chemical compounds has made it possible to coordinate a computational process with experimental research efforts to achieve a common goal. The ultimate purpose of the entire effort is to elucidate potential risks of various regulatory options and to help the industry to select the least costly and least hazardous control technology options.

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DISCUSSION

F.A. SEILER: The separation of a mixture of a large number of chemicals into risk assessment units (RAUs) is indeed a practical way of dealing with this problem. However, I am not entirely satisfied with the mathematical formulation of your approach. Since it is totally linear, the absence of synergisms, both between substances in the same RAU and substances in different RAUs, should have been included in your list of conditions for this method. Such terms would be quadratic, i.e. proportional to the products of doses. In addition, you do not allow for the non-linearity of dose-effect relationships. Could you comment?

A.A. MOGHISSI: I agree that the assumption underlying the RAU concept is the additivity of the effect. We disregard the synergism and antagonism between two RAUs because we do not see any practical way of establishing the synergistic or antagonistic effects of the very large number of chemicals involved. Obviously, the non-linearity of the dose-effect for each chemical cannot be established since the dose-effect relationship is unknown. It should be understood that one of the primary objectives of this exercise is to establish an order of priority among toxicants.

K. SUNDARAM: You have included mutagenicity and carcinogenicity in your projections. I understand that these are in-vitro test systems. Could you explain how you propose to resolve the difficulty of transferring positive test results to human risk?

A.A. MOGHISSI: In-vitro systems are used primarily to rank waste streams. Recent experience with diesel has led to some hopeful developments in the use of these tests for risk assessment. However, at the moment we plan to rely on animal and/or human data.

F. GIRARDI: I have two questions. First, could you give some indication of the time scale involved in the process of defining the RAUs and the transfer coefficients? Secondly, is the concept of RAUs applied only to synfuels or is it becoming the general policy of the US Environmental Protection Agency?

A.A. MOGHISSI: The RAUs will be defined within the next month. We hope to have some data on all areas of concern within a year, and those data will be improved in future years.

As for your second question, the RAU concept is at the moment limited to our energy efforts.

J. SINNAEVE: The risk analysis unit approach appears to be very attractive. However, I would like to know how you take into account the toxicological effects of metabolites in mammalians, i.e. metabolites derived from organic chemicals within one RAU. This was not reflected in the mathematical formulation of your overall risk factor for health effects.

A.A. MOGHISSI: The toxicity evaluation takes into account the metabolism of various compounds. A risk factor is assigned to a chemical compound and this obviously includes effects produced by the metabolites of that compound.

HEALTH IMPLICATIONS OF ENERGY PRODUCTION

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Abstract

HEALTH IMPLICATIONS OF ENERGY PRODUCTION.

Recent WHO activities concerning the impact on environmental health of all forms of energy production are reported. The concept of 'health detriment' is defined. Risk estimates from the nuclear fuel cycle, the fossil fuel cycles (coal, oil and natural gas), renewable energy sources and local energy sources mainly used in developing countries are given.

1. HEALTH IMPACT OF ENERGY PRODUCTION

The environmental and health impact of energy production is one of the most important fields in which the international organizations of the UN family have been active.

The World Health Organization's (WHO) programme in environmental health is in the process of establishing criteria for the main environmental pollutants, some of which,such as sulphur dioxide, CO and CO_2, nitrogen dioxide and suspended particulates, are directly connected with energy production. A monitoring system to evaluate the pollution of air and other environmental media is now being developed under WHO supervision in the joint project with the UN Environment Programme (UNEP) and other international organizations.

Rapid increase in energy production,as an essential part of socio-economic development, requires the evaluation of comparative health detriment from various sources of energy (wood, coal, oil, nuclear, etc.) for planning purposes and preventive environmental health action. Approaches to comparative assessment of health detriment from energy production were considered at a WHO meeting jointly organized with experts from the Federal Republic of Germany to assist in planning national energy programmes. These included centralized systems as well as local energy sources for domestic use in small communities in rural areas, particularly in developing countries.

In planning national energy programmes it is important to consider health detriment from different energy sources, including centralized systems for generating electricity and the consumer-produced energy used for domestic cooking and heating together with energy required for small communities in rural areas, particularly in the developing countries.

To assess health detriment from generating electricity by different methods, it seems appropriate to adopt the approach used for such assessments in the nuclear fuel cycle, including all stages of the cycle, from mining radioactive ore to final decommissioning of a nuclear power station. This includes (a) assessment of occupational injury from radiation exposure and conventional injuries, and (b) health detriment suffered by the public from low levels of exposure to radioactive pollutants both from normal operation and from accident conditions.

The concept of *health detriment* applicable to exposure to ionizing radiation must be defined. The deleterious effects of exposure to ionizing radiation include stochastic and non-stochastic effects in the exposed individuals, as well as stochastic effects in later generations, i.e. hereditary damage. The concept of health detriment expresses the whole damage to man from the biological effects that can appear after an exposure in defined conditions of exposure and at a given dose level. The consequences of deleterious effects may be impaired health, suffering and death. The assessment also involves consideration of the latent periods after which effects may develop following an exposure. The ICRP has defined the concept by a simple mathematical equation relating detriment to the probability of the occurrence and of the severity of the effect.

Although the concept of detriment can be applied to both stochastic and non-stochastic effects, it is mainly used to estimate damage from stochastic effects. In this case, the probability is an increasing function of dose, and the severity factor is not dose-dependent. Protection against non-stochastic effects can be achieved by setting dose limits below a threshold value, whereas no threshold is assumed for stochastic effects. It is therefore necessary to select an acceptable level of risk for stochastic effects.

In assessing health detriment in large groups exposed occupationally or as members of the public, the concept of *collective detriment* may be used. This can be applied to exposures from a nuclear power station or from any stage of the nuclear fuel cycle. The total dose equivalent committed to the exposed group as a whole is used as a measure of collective detriment.

In the nuclear fuel cycle, the health effects of uranium mining, fuel processing, conversion to electrical energy in a reactor, and reprocessing waste, represent the most important stages. Similar assessments can be made for fossil fuels (coal, oil and natural gas), solar energy, hydroelectric power and other fuel cycles.

2. THE NUCLEAR FUEL CYCLE

The low doses and dose rates from natural radiation in the environment, and the considerably lower doses that would be given at low dose rates in those occupationally exposed (with the possible exception of uranium miners) at the various stages in the nuclear fuel cycle, are very much lower than those for which epidemiological data exist on the frequency of tumour induction. For radiation protection purposes, and to get an approximate estimate of risk, one assumes that there is a linear relationship between the dose and the probability of a stochastic effect – tumour induction or genetic effect – within the range of doses encountered in the nuclear fuel cycle. If the relationship were in fact more complex, as some animal experiments and theoretical considerations suggest, this linear assumption would produce estimates of risk which are generally characterized as tending to be upper bounds, i.e. overestimates of the actual risk. The lower bounds of the risk from exposure to low-level and low-LET radiation (the type emitted from reactors) could include zero.

Radiation is one of the most studied of environmental pollutants and one that can be measured accurately and relatively easily down to very low levels. The objectives of radiation protection and the definitions of basic concepts such as detriment, absorbed dose, dose equivalent, and effective dose equivalent have all been made by the ICRP[1] and elaborated in the publications of various international organizations.

In considering the general assessment of the nuclear fuel cycle, it must be kept in mind that the actual effects of individual plants will vary, depending on local population density and meteorology. In developing countries the doses defined and the resulting effects will be influenced by the technical backup available, since it may not always be possible to assume satisfactory operating procedures and controls.

The following recommendations on estimation of health detriment from radiation and the nuclear fuel cycle should be made:

(a) Important studies are being carried out on non-stochastic effects by UNSCEAR and ICRP, and on hereditary effects by UNSCEAR. WHO should follow the progress of these studies.

(b) International organizations should encourage appropriate national agencies to collect up-to-date information on the accidents and non-radiation diseases of uranium miners, and should follow developments on the definition of radiation-induced cancer in uranium miners. Regarding potentiation between cigarette-smoking and exposure to Rn-222 decay products, recent data from miner groups and from the Japanese A-bomb survivors indicate that smoking shortens the latent period to the onset of bronchial cancer, but that the combination of smoking and radiation leads to a cancer risk that is not much

[1] INTERNATIONAL COMMISSION ON RADIOLOGICAL PROTECTION, Recommendations of the ICRP, ICRP Publication 26, Ann. ICRP 1 3, Pergamon Press, London (1977).

more than additive. WHO should seek to promote, in this and in other appropriate areas, competent studies of potentiation of the effects of environmental pollutants.

(c) We should continue to maintain a critical overview of the health problems associated with the nuclear fuel cycle, and invite the co-operation of the appropriate organizations in these subjects.

(d) WHO should promote, in association with IAEA, UNSCEAR, OECD/NEA, and collaborating centres, appropriate practice and control measures throughout the nuclear fuel cycle so as to assist national health authorities to implement at all stages the recommended dose limitations.

3. THE FOSSIL FUEL CYCLES

The fossil fuel cycles for centralized generation of electricity include the use of coal, oil and natural gas. It is now well established that fossil fuels, particularly coal and natural gas, will continue to constitute an important component of the energy mix in the coming decades. In previous use of coal for power generation, the main problems were the health effects on coal-miners and the exposure of the public to chemical pollutants released. Assessment of health detriment from fossil fuel cycles is more complex than for the nuclear fuel cycle because of our lack of knowledge of the chemical toxicology of the large variety of pollutants involved.

For the fuel resource and the end use of the fuel, the approach developed to the nuclear fuel cycle can be used. The main difference arises from our lack of knowledge of the dose-effect relationship of the chemical pollutants for stochastic effects. The non-stochastic effects, including health detriments like respiratory and cardiac disorders in critical groups, are well recognized; contributions from stochastic effects are at present undetected and so are likely to be comparatively small.

Approaches to the assessment of health detriment from fossil fuel cycles must include detailed reviews of the problems in coal-mining, drilling operations for oil and natural gas (particularly offshore drilling), air and water pollution, and accidents, including mine and tanker explosions.

Regarding air pollution problems from all types of fossil-fuel stations, no entirely satisfactory approach is yet available to estimate the damage function and evaluate the harm to the health of the general public. Epidemiological studies to determine the damage function of each possible pollutant would involve enormous human effort and cost. The WHO and FRG joint programme nevertheless considered in great detail various pollutants that could be used as index substances for estimation of harm. Amongst these are polycyclic aromatic hydrocarbons (PAH), sulphates and total particulates. Benzo(a)pyrene studies carried out in the past have provided some correlation between the air concentration of this substance and the incidence of lung cancer. Research effort is necessary to understand the mechanisms of toxic effects of different PAH and to identify promotion of their carcinogenic effects. Studies of potentiation and antagonism in the induction of cancer by the PAH are necessary. Detailed studies of the profiles of PAH under different climatic and environmental conditions, particle size distribution and atmospheric conversion would also be of great value.

The monitoring of chemical pollutants must be continued in order to study the trends and improve our understanding of stochastic and non-stochastic health effects. Models based on particulate acid sulphates as the index of detriment have been used in the past for stochastic effects. A risk estimate based on total suspended particulates has been used for assessing for detriment from sulphates, which can be used in the model, including SO_2 emissions from coals of different sulphur content. Such practices provide some control of non-stochastic effects on the public.

The health detriment associated with the radioactive content of coal has also to be assessed for established power stations. The assessments made for different conditions of exposure show that external irradiation due to surface deposition as well as ingestion of contaminated food by the critical groups are important contributors to the collective effective dose-commitment from this source. The exposure depends on the radioactive content of coal, which is highly variable. Fly-ash control can reduce the exposure by large factors.

4. RENEWABLE SOURCES OF ENERGY

Renewable sources of energy are often regarded as risk-free because they do not produce substantial quantities of toxic effluents. However, if an assessment of health detriments is made for all the different stages of these energy cycles, particularly those associated with construction, fabrication and maintenance, the total detriment can indeed be significant.

For the purpose of this paper, renewable energy sources include central electricity generation from hydroelectric, geothermal, solar, wind, wave (including tidal) and biomass. The health impact, expressed in terms of harm per $GW(e) \cdot a$, varies widely among the different renewable sources of energy.

5. LOCAL ENERGY SOURCES, PRIMARILY USED IN DEVELOPING COUNTRIES

The energy scenario of most developing countries is different from that of the economically advanced countries. In the urban areas of developing countries, the commercial energy production is basically similar to that in the developed world. However, in many rural areas there is no centralized production of electricity, and energy for domestic and non-domestic use is obtained from the direct combustion of firewood and farm waste.

Whereas adequate models are available for the assessment of health detriment from most centralized systems, new approaches have to be developed for assessing health detriment from non-commercial energy sources. Important examples of these are:

(1) The use of firewood, agricultural waste and dried animal dung by direct combustion for domestic uses; and

(2) Biogas plants using wet animal and plant waste for the production of combustible gases for the use of small communities.

In some of the developing countries, e.g. India, nearly 40–50% of domestic energy is obtained from the combustion of firewood, agricultural waste and dried animal dung, and a large proportion of the rural population depends on such sources. The combustion of these materials gives rise to a substantial production of pollutants including CO, NO_x, gaseous and particulate organics, NH_3, H_2S and HCl. These wastes released into the air are likely to cause considerable health detriment for which no systematic study yet exists. Biogas plants have also not been assessed for the health detriment associated with their development. Large numbers of small biogas plants are being used in developing countries, and proposals have been made for medium-size biogas plants for the use of rural communities with populations of 500 to 1000. These plants appear attractive since they can lead to the conservation of precious forests and kerosene, and save electricity for other use.

The typical case of rural India has been presented in Dr. Vohra's paper at this Symposium (IAEA-SM-254/102). In the method of energy production described in the paper the fuel cycles are simple, including the collection of solid fuels by single families or small communities and their direct combustion without any control of air pollution. Fairly large concentrations of harmful pollutants are released. Health detriment from this source, although reported through sporadic observations, has not been systematically assessed. There is a need to develop models for such assessment and to compare the detriment with that for centralized energy production.

It is anticipated that such assessments would probably indicate greater health benefits for the centralized alternative. However, inadequate transport facilities make it difficult to provide commercial centralized energy in the rural areas where the greater part of the population of developing countries lives. The non-centralized sources provide as much as 50% of total energy requirements for nearly half the world, i.e. Asia, Africa and Latin America. Their advantage lies in that they are locally available and essentially renewable.

It is, in fact, possible to make efficient use of these resources for both domestic and agricultural use, for example in biogas plants. Some developing countries in tropical regions are indeed already developing programmes for the use of animal and farm waste for the production of combustible gases in such biogas plants with capacities of a few kilowatts. These biogas plants have large apparent advantages, e.g. saving firewood, conserving forests, saving kerosene, and the availability of fertilizers from their waste. Moreover the gas generated by these plants can serve domestic, transport and other uses, and the residual waste can be used as fertilizer.

The health benefits and detriments of these alternative methods of energy production should be reviewed by international organizations.

6. GENERAL CONCLUSIONS

(1) For assessment of the radiation health detriment, the assumption of a linear dose-response relationship is thought to give an overestimated value for the risk from low doses of low-LET radiation of long, protracted exposure. For high-LET radiation the same assumption is thought to give the best estimated value. For chemical substances, the uncertainties are higher. But, for many chemical agents, an over-estimate of the health detriment is furnished by the linear hypothesis, for example for various diseases such as bronchitis, emphysema and lung cancer. WHO should further encourage the assessment of dose-effect relationships for chemical agents and the examination of the possibility of thresholds.

(2) The total detriment is the sum of all deleterious effects caused by the different noxious agents. If we wish to compare the various agents, a similar assessment of the total detriment is necessary. For instance, we have to take into account non-stochastic effects as well as stochastic effects such as the induction of cancer and hereditary effects. WHO should encourage studies to increase our knowledge of the different components of the total detriment.

(3) The problem of potentiation or antagonism between noxious agents is important for assessment of the total detriment, and WHO should encourage studies on potentiation between radiological and chemical agents or between chemical agents. As in the case of a simple additivity, it is desirable to choose a common unit in which to express the various detriments, such as loss of days for workers or members of the public.

(4) The concept of harm commitment from the late effects of noxious agents is complex. The first component is related to the level of pollution of

the environment; the second is connected with the dose received after intake by the exposed persons; the third is dependent on the latent period for the clinical manifestation of the effects. In considering the long-lived radionuclides or stable chemical substances, one must be realistic in order to avoid the difficulties of interpretation of a detriment integrated over infinite time. UNSCEAR takes into account a continuous production of energy during the next 500 years at the same rate as would be reached by the year 2000. We should consider this a reasonable basis for assessing future health detriment.

(5) For comparing the health detriment attributable to energy production from different sources, the detriment is better related to the practice (e.g. expressed as per GW(e) · a output) than to the source (e.g. per power station). Thus the estimated collective doses to workers and to the public from the nuclear fuel cycle have been evaluated as 50 to 80 man-sieverts (5000 to 8000 man-rem) per GW(e) · a output. The corresponding total detriment from radiation and other causes has been estimated to be in the region of one death per year in a given population of 1 to 2 million persons deriving their total electricity consumption of 1 GW(e) from nuclear sources.

(6) The corresponding values for other fuel cycles are not so well established, but provisional estimates suggest totals of the order of 10 deaths per year per GW(e) · a derived from coal or oil, which would supply a similar population of 1 to 2 million. In a population of this size there would be from 10 000 to 25 000 deaths per year from all causes. It is evident that deaths attributable to electrical energy production vary substantially according to the fuel source but form only a very small proportion of deaths from all causes, whatever the source. WHO should assist in promoting a better assessment of the total health detriment per GW(e) · a for the various sources of production of energy, or per unit output of energy from other sources including the non-centralized. These considerations could assist countries to formulate health and energy policies for their socio-economic development.

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Italic numerals refer to the first page of a paper by the author concerned
Upright numerals denote comments and questions in discussions

TRANSLITERATION INDEX

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The following conversion table is provided for the convenience of readers

FACTORS FOR CONVERTING SOME OF THE MORE COMMON UNITS TO INTERNATIONAL SYSTEM OF UNITS (SI) EQUIVALENTS

NOTES:

(1) SI base units are the metre (m), kilogram (kg), second (s), ampere (A), kelvin (K), candela (cd) and mole (mol).

(2) ▶ indicates SI derived units and those accepted for use with SI;
▷ indicates additional units accepted for use with SI for a limited time.
[*For further information see the current edition of The International System of Units (SI), published in English by HMSO, London, and National Bureau of Standards, Washington, DC, and International Standards ISO-1000 and the several parts of ISO-31, published by ISO, Geneva.*]

(3) The correct symbol for the unit in column 1 is given in column 2.

(4) * indicates conversion factors given exactly; other factors are given rounded, mostly to 4 significant figures.
≡ indicates a definition of an SI derived unit: [ ] in columns 3+4 enclose factors given for the sake of completeness.

Column 1 *Multiply data given in:*	Column 2	Column 3 *by:*	Column 4 *to obtain data in:*	
Radiation units				
▶ becquerel	1 Bq	(has dimensions of s^{-1})		
disintegrations per second (= dis/s)	1 s^{-1}	≡ 1.00×10^{0}	Bq	*
▷ curie	1 Ci	= 3.70×10^{10}	Bq	*
▷ roentgen	1 R	[= 2.58×10^{-4}	C/kg]	*
▶ gray	1 Gy	[≡ 1.00×10^{0}	J/kg]	*
▷ rad	1 rad	= 1.00×10^{-2}	Gy	*
▶ sievert *(radiation protection only)*	1 Sv	[= 1.00×10^{0}	J/kg]	*
rem *(radiation protection only)*	1 rem	[= 1.00×10^{-2}	J/kg]	*
Mass				
▶ unified atomic mass unit ($\frac{1}{12}$ of the mass of ^{12}C)	1 u	[= $1.660\,57 \times 10^{-27}$	kg, approx.]	
▶ tonne (= metric ton)	1 t	[= 1.00×10^{3}	kg]	*
pound mass (avoirdupois)	1 lbm	= 4.536×10^{-1}	kg	
ounce mass (avoirdupois)	1 ozm	= 2.835×10^{1}	g	
ton (long) (= 2240 lbm)	1 ton	= 1.016×10^{3}	kg	
ton (short) (= 2000 lbm)	1 short ton	= 9.072×10^{2}	kg	
Length				
statute mile	1 mile	= 1.609×10^{0}	km	
nautical mile (international)	1 n mile	= 1.852×10^{0}	km	*
yard	1 yd	= 9.144×10^{-1}	m	*
foot	1 ft	= 3.048×10^{-1}	m	*
inch	1 in	= 2.54×10^{1}	mm	*
mil (= 10^{-3} in)	1 mil	= 2.54×10^{-2}	mm	*
Area				
▷ hectare	1 ha	[= 1.00×10^{4}	m^2]	*
▷ barn *(effective cross-section, nuclear physics)*	1 b	[= 1.00×10^{-28}	m^2]	*
square mile, (statute mile)^2	1 $mile^2$	= 2.590×10^{0}	km^2	
acre	1 acre	= 4.047×10^{3}	m^2	
square yard	1 yd^2	= 8.361×10^{-1}	m^2	
square foot	1 ft^2	= 9.290×10^{-2}	m^2	
square inch	1 in^2	= 6.452×10^{2}	mm^2	
Volume				
▶ litre	1 l *or* 1 ltr	[= 1.00×10^{-3}	m^3]	*
cubic yard	1 yd^3	= 7.646×10^{-1}	m^3	
cubic foot	1 ft^3	= 2.832×10^{-2}	m^3	
cubic inch	1 in^3	= 1.639×10^{4}	mm^3	
gallon (imperial)	1 gal (UK)	= 4.546×10^{-3}	m^3	
gallon (US liquid)	1 gal (US)	= 3.785×10^{-3}	m^3	

This table has been prepared by E.R.A. Beck for use by the Division of Publications of the IAEA. While every effort has been made to ensure accuracy, the Agency cannot be held responsible for errors arising from the use of this table.

Column 1 *Multiply data given in:*	Column 2	Column 3 *by:*	Column 4 *to obtain data in:*	
Velocity, acceleration				
foot per second (= fps)	1 ft/s	$= 3.048 \times 10^{-1}$	m/s	*
foot per minute	1 ft/min	$= 5.08 \times 10^{-3}$	m/s	*
mile per hour (= mph)	1 mile/h	$= \begin{cases} 4.470 \times 10^{-1} \\ 1.609 \times 10^{0} \end{cases}$	m/s km/h	
▷ knot (international)	1 knot	$= 1.852 \times 10^{0}$	km/h	*
free fall, standard, g		$= 9.807 \times 10^{0}$	m/s^2	
foot per second squared	$1\ ft/s^2$	$= 3.048 \times 10^{-1}$	m/s^2	*
Density, volumetric rate				
pound mass per cubic inch	$1\ lbm/in^3$	$= 2.768 \times 10^{4}$	kg/m^3	
pound mass per cubic foot	$1\ lbm/ft^3$	$= 1.602 \times 10^{1}$	kg/m^3	
cubic feet per second	$1\ ft^3/s$	$= 2.832 \times 10^{-2}$	m^3/s	
cubic feet per minute	$1\ ft^3/min$	$= 4.719 \times 10^{-4}$	m^3/s	
Force				
▶ newton	1 N	$[\equiv 1.00 \times 10^{0}$	$m \cdot kg \cdot s^{-2}]$	*
dyne	1 dyn	$= 1.00 \times 10^{-5}$	N	*
kilogram force (= kilopond (kp))	1 kgf	$= 9.807 \times 10^{0}$	N	
poundal	1 pdl	$= 1.383 \times 10^{-1}$	N	
pound force (avoirdupois)	1 lbf	$= 4.448 \times 10^{0}$	N	
ounce force (avoirdupois)	1 ozf	$= 2.780 \times 10^{-1}$	N	
Pressure, stress				
▶ pascal	1 Pa	$[\equiv 1.00 \times 10^{0}$	$N/m^2]$	*
▷ atmosphere[a], standard	1 atm	$= 1.013\,25 \times 10^{5}$	Pa	*
▷ bar	1 bar	$= 1.00 \times 10^{5}$	Pa	*
centimetres of mercury (0°C)	1 cmHg	$= 1.333 \times 10^{3}$	Pa	
dyne per square centimetre	$1\ dyn/cm^2$	$= 1.00 \times 10^{-1}$	Pa	*
feet of water (4°C)	$1\ ftH_2O$	$= 2.989 \times 10^{3}$	Pa	
inches of mercury (0°C)	1 inHg	$= 3.386 \times 10^{3}$	Pa	
inches of water (4°C)	$1\ inH_2O$	$= 2.491 \times 10^{2}$	Pa	
kilogram force per square centimetre	$1\ kgf/cm^2$	$= 9.807 \times 10^{4}$	Pa	
pound force per square foot	$1\ lbf/ft^2$	$= 4.788 \times 10^{1}$	Pa	
pound force per square inch (= psi)[b]	$1\ lbf/in^2$	$= 6.895 \times 10^{3}$	Pa	
torr (0°C) (= mmHg)	1 torr	$= 1.333 \times 10^{2}$	Pa	
Energy, work, quantity of heat				
▶ joule ($\equiv W \cdot s$)	1 J	$[\equiv 1.00 \times 10^{0}$	$N \cdot m]$	*
▶ electronvolt	1 eV	$[= 1.602\,19 \times 10^{-19}$	J, approx.]	
British thermal unit (International Table)	1 Btu	$= 1.055 \times 10^{3}$	J	
calorie (thermochemical)	1 cal	$= 4.184 \times 10^{0}$	J	*
calorie (International Table)	$1\ cal_{IT}$	$= 4.187 \times 10^{0}$	J	
erg	1 erg	$= 1.00 \times 10^{-7}$	J	*
foot-pound force	$1\ ft \cdot lbf$	$= 1.356 \times 10^{0}$	J	
kilowatt-hour	$1\ kW \cdot h$	$= 3.60 \times 10^{6}$	J	*
kiloton explosive yield (PNE) ($\equiv 10^{12}$ g-cal)	1 kt yield	$\simeq 4.2 \times 10^{12}$	J	

[a] atm (g) (= atü): atmospheres gauge
atm abs (= ata): atmospheres absolute

[b] lbf/in^2 (g) (= psig): gauge pressure;
lbf/in^2 abs (= psia): absolute pressure.

Column 1 *Multiply data given in:*	Column 2	Column 3 *by:*	Column 4 *to obtain data in:*	
Power, radiant flux				
▶ watt	1 W	[≡ 1.00×10^0	J/s]	*
British thermal unit (International Table) per second	1 Btu/s	= 1.055×10^3	W	
calorie (International Table) per second	1 cal_{IT}/s	= 4.187×10^0	W	
foot-pound force/second	1 ft·lbf/s	= 1.356×10^0	W	
horsepower (electric)	1 hp	= 7.46×10^2	W	*
horsepower (metric) (= ps)	1 ps	= 7.355×10^2	W	
horsepower (550 ft·lbf/s)	1 hp	= 7.457×10^2	W	
Temperature				
▶ kelvin	K			
▶ degrees Celsius, t where T is the thermodynamic temperature in kelvin and T_0 is defined as 273.15 K	$t = T - T_0$			*
degree Fahrenheit	$t_{°F} - 32$	$\times \left(\frac{5}{9}\right)$ gives	t *(in degrees Celsius)*	*
degree Rankine	$T_{°R}$	$\times \left(\frac{5}{9}\right)$ gives	T *(in kelvin)*	*
temperature difference[c]	$\Delta T_{°R}$ (= $\Delta t_{°F}$)	$\times \left(\frac{5}{9}\right)$ gives	ΔT (= Δt)	*
Thermal conductivity[c]				
1 Btu·in/(ft²·s·°F) *(International Table Btu)*		= 5.192×10^2	$W \cdot m^{-1} \cdot K^{-1}$	
1 Btu/(ft·s·°F) *(International Table Btu)*		= 6.231×10^3	$W \cdot m^{-1} \cdot K^{-1}$	
1 cal_{IT}/(cm·s·°C)		= 4.187×10^2	$W \cdot m^{-1} \cdot K^{-1}$	
Miscellaneous quantities				
litre per mole per centimetre *(molar extinction coefficient or molar absorption coefficient)*	(1M/cm =) 1 $ltr \cdot mol^{-1} \cdot cm^{-1}$	= 1.00×10^{-1}	m^2/mol	*
G-value, traditionally quoted per 100 eV of energy absorbed *(radiation yield of a chemical substance)*	$1 \times 10^{-2}\ eV^{-1}$	= 6.24×10^{16}	J^{-1}	
mass per unit area *(absorber thickness and mean mass range)*	1 g/cm^2	[= 1.00×10^1	kg/m^2]	*

[c] A temperature interval or a Celsius temperature difference can be expressed in degrees Celsius as well as in kelvins.

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